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C. Max Finlayson · Mark Everard
Kenneth Irvine · Robert J. McInnes
Beth A. Middleton · Anne A. van Dam
Nick C. Davidson *Editors*

The Wetland Book

I: Structure and Function, Management, and Methods

Springer

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Editors

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With 369 Figures and 74 Tables

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Foreword: The Wetland Book

The venerable lineage of encyclopedic publishing can be traced back to Pliny the Elder's *Naturalis Historia*, which contained chapters on water and aquatic life. Although our terminology regarding and understanding of the aquatic environment has evolved over the past two millennia, one constant has been the need for a multidisciplinary approach to examining these areas. Using an encyclopedic model, this multidisciplinary book builds on an ancient format and adapts it for a modern audience. In this way, *The Wetland Book* builds on a long tradition of scholarly publishing and presents invaluable information for its modern audience.

Wetlands have been around longer than the traditions associated with academic publishing. Wetland management and wise use have been practiced by indigenous cultures in many forms for millennia, and that ancient knowledge about wetlands was often curated and passed down orally or in traditional systems and forms. In modern times, the pressures and threats to wetlands are vastly different in their scope and magnitude. The forms of governance and administration that respond to these pressures and threats have also changed, particularly in their scale as it has been recognized that management takes place up to the level of countries and river basins, rather than simply at the local level.

Internationally, wetland conservation, management, and wise use are promoted through the Ramsar Convention on Wetlands. The countries that have signed onto the Ramsar Convention have recognized the imperatives to work with stakeholders and decision makers beyond the traditional wetland community and to incorporate wetlands into policy-making in other sectors such as water, energy, agriculture, and health. Indeed, in 2008 at the 10th Conference of the Contracting Parties of the Ramsar Convention, the Changwon Declaration was adopted, which contains key messages for wetland conservation, management, and wise use addressed to planners; policymakers; elected officials; managers in the environmental, land, and resource-use sectors; educators and communicators; economists; and health workers. *The Wetland Book* offers a base of knowledge that is intended to reach a similarly broad audience.

The editors and contributing authors to *The Wetland Book* have long experience and deep understanding of wetland science and management. Many have worked with the Ramsar Scientific and Technical Review Panel (STRP), the Convention's scientific advisory body, over the years. This collection of people provides a repository of knowledge that can help meet the challenge of learning about and understanding the value of protecting and managing wetlands.

Making this knowledge more easily accessible, however, has always been difficult. There are physical limitations to how much we can pick a person's brain, and there are limitations to how much a wetland manager out in the field, perhaps with little technical support, can search for, read, and review scientific and traditional knowledge to find answers to pressing questions. Thus, the encyclopedic style of publication remains a viable format for accessing high levels of expertise, including expertise from distant locations with similar landscape and ecological characteristics. *The Wetland Book* provides an in-depth level of knowledge in the form of a handbook to assist those seeking information on the many facets of wetland management.

Of course, reading *The Wetland Book* will not make an individual an expert in all aspects of wetland science, wise use, and governance, a feat which no one publication can deliver. Instead, a truly useful publication should offer an individual the vocabulary to support further inquiry and to find knowledge that is locally, regionally, nationally, or even internationally applicable. It should also allow a reader to know who to ask and what questions to pursue when she or he needs more knowledge to solve a research question or particular management problem. *The Wetland Book* delivers this foundation through two volumes – Vol. 1: Structure and Function, Management, and Methods and Vol. 2: Distribution, Description, and Conservation.

We highly recommend *The Wetland Book*; it provides an unparalleled source of knowledge about wetlands by building on the ancient form of the encyclopedia, revitalized by new technologies for distribution and access. We are also proud to see that many of those who have contributed to the Ramsar Convention over many years or even decades have also contributed their knowledge and wisdom to *The Wetland Book*. Given our personal association with the convention, we also recognize the incredible contribution that the Convention has made to wetland knowledge and look forward to further contributions.

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Chair, Scientific and Technical Review Panel,
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Preface

The Wetland Book is a hard copy and online production that provides an unparalleled collation of information on wetlands. It is global in scope and contains 462 chapters prepared by leading wetland researchers and managers. The wide disciplinary and geographic scope is a unique feature and differentiates *The Wetland Book* from the existing wetland literature. The editors have compiled *The Wetland Book* from contributions supplied by authors from many countries and disciplines. Combined, these chapters represent a global source of knowledge about wetlands. Given the number of chapters and the scope of the content, it has been published as two separate books.

The bibliographic detail of two books is given below. Book 1 with 292 chapters covers the structure and function of wetlands, as well as the management and the methods used to investigate them.

The Wetland Book I: Structure and Function, Management, and Methods: edited by Finlayson CM, Everard M, Irvine K, McInnes RJ, Middleton BA, van Dam AA and Davidson NC.

Its companion book, published separately, with 170 chapters is:

The Wetland Book II: Distribution, Description, and Conservation: edited by Finlayson CM, Milton GR, Prentice RC and Davidson NC.

The Wetland Book was developed following discussions with wetland experts from the Scientific and Technical Review Panel of the Ramsar Convention on Wetlands and from the Society for Wetland Scientists. These experts pointed to the rapidly expanding literature on wetlands and enthusiastically proposed the development of a comprehensive information resource aimed at supporting the trans- and multidisciplinary research and practice, which is essential to wetland science and management. They were also seeking an information resource that would both complement and extend the existing literature and in particular provide a compendium of knowledge with contributions from authors around the world.

Aware that wetland research was on the rise and that wetland researchers and practitioners often needed to work across disciplines, *The Wetland Book I* has been prepared to serve as a first port of call for those interested in the key concepts in wetland science and management. This approach was taken to allow individuals and multi- and transdisciplinary teams to search for particular terms and subjects, access further details, and read overviews of topics selected by the editors and expert authors. The content provides a global coverage of wetland knowledge with chapters provided by leading wetland experts with information that spans local and regional issues to the wider body of science that is needed to assist practitioners and enable students to come to grips with one of the world's most diverse and important set of ecosystems. This content is especially important as wetland ecosystems in many parts of the world are under increasing pressure due to degradation from human development, which continues at an alarming rate and will require more effective management and restoration. It draws heavily on knowledge compiled through the formal processes of the Ramsar Convention and associated programs and extends upon information contained in the seminal global assessment of wetlands undertaken through the Millennium Ecosystem Assessment.

Book 1 is organized in three parts, Wetland Structure and Function (coordinated by B.A. Middleton); Wetland Management (coordinated by M. Everard and R.J. McInnes); and Wetland Methods (coordinated by A.A. van Dam and K. Irvine). Each part is divided into thematic sections with one or more overview chapers, supported by articles and case studies providing further information on different aspects of the theme. Each section was developed and collated by section leads, namely M. Acreman, M. Alexander, J.A. Boudell, N.C. Davidson, M. Demissie, M. Everard, C.M. Finlayson, P. Gerbeaux, R. Kumar, R. Lucas, R.J. McInnes, B.A. Middleton, D. Moreno Mateos, R. Slootweg, K. Stevens, and C. Stratford. For Wetland Structure and Function, there are chapters covering: succession; biological adaptations; ecological processes and biogeochemistry; importance of hydrology to wetlands; and landscape ecology. For Wetland Management, there are chapters covering: an overview of wetland management; the international framework for wetland conservation and wise use; wetland law and policy; and management of ecosystem services. For Wetland Methods, the chapters cover: wetland delineation and classification; Earth observation approaches and their application to wetland ecosystems; wetland monitoring and assessment; environmental flows; wetland management planning; wetland restoration and creation; environmental impact assessment; strategic environmental assessment; and economic valuation.

We warmly convey our special thanks to those who have acted as section leads and to all the many authors who gave their time and knowledge of wetlands to support this effort as well as their patience while the large number of chapters were collated and edited. We are proud to have worked with them to produce this book. With the benefit of their unstinting efforts and incredibly rich knowledge, *The Wetland Book I* provides a comprehensive source of information for wetland researchers, students, and practitioners. It specifically provides a much needed information resource to support the many efforts to ensure the wise use of wetlands

globally. It has not only drawn on but also extended the expert guidance and advice that the Ramsar Convention's Scientific and Technical Review Panel has provided for governments and wetland experts alike for almost 25 years. In this respect, the foreword provided by the past and present chairs of the panel is particularly appreciated. In providing the foreword, they have reflected on the wealth of knowledge collated by wetland experts who have worked tirelessly to provide government officials with the knowledge base needed to ensure the conservation and wise use of wetlands around the world.

As editors for *The Wetland Book I*, we personally compliment the many contributors and extend a huge vote of gratitude to our section leads for their incredible contributions to the most comprehensive compendium of knowledge about wetlands ever assembled. In particular, we commend their contributions to the wetland literature and acknowledge their unstinting efforts to compile the many chapters and work with the authors to produce *The Wetland Book*. Their knowledge and efforts are matched by their willingness to share the collated knowledge that is now contained in the *The Wetland Book*.

The publisher is thanked for their foresight in developing the concepts that led to *The Wetland Book* and for providing both a hard copy and online version, with the latter being available for future updating. We recommend *The Wetland Book* to all those interested in the growing international knowledge about wetland science and management of these incredibly valuable but threatened ecosystems.

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Max Finlayson is an internationally renowned wetland ecologist with extensive experience internationally in water pollution, agricultural impacts, invasive species, climate change, and human well-being and wetlands. He has participated in global assessments such as those conducted by the Intergovernmental Panel for Climate Change, the Millennium Ecosystem Assessment, and the Global Environment Outlook 4 and 5 (UNEP). Since the early 1990s, he has been a technical adviser to the Ramsar Convention on Wetlands and has written extensively on wetland ecology and management. He has also been actively involved in environmental NGOs and from 2002 to 2007 was president of the governing council of global NGO Wetlands International.

Professor Finlayson has worked extensively on the inventory, assessment, and monitoring of wetlands, in particular in wet tropical, wet-dry tropical, and subtropical climatic regimes covering pollution, invasive species, and climate change. His current research interests/projects include the following:

- Interactions between human well-being and wetland health in the face of anthropogenic change, including global change and the onset of the Anthropocenic era
- Vulnerability and adaptation of wetlands/rivers to climate change, including changing values and trade-offs between uses and users, considering uncertainty and complexity
- Integration of ecologic, economic, and social requirements and trade-offs between users of wetlands with an emphasis on developing policy guidance and institutional changes
- Environment and agriculture interactions and policy responses/outcomes, and collaboration between stakeholders and policymakers

- Wetland restoration and construction, including the use of artificial wetlands for waste water treatment and the generation of multiple values
- Landscape change involving wetlands/rivers and land use (agriculture and mining) and implications for wetland ecosystem services and benefits for local people

He holds the following associated positions

- Scientific Expert on the Scientific and Technical Review Panel, Ramsar Convention on Wetlands, Triennium 2016–2018
- Ramsar Chair for the Wise use of Wetlands, UNESCO-IHE, Delft, The Netherlands (2014–2018)
- Visiting Professor, Institute for Wetland Research, China Academy of Forestry, Beijing, China
- Editor-in-Chief, Marine and Freshwater Research, CSIRO Publishing
- Chair, Environmental Strategy Advisory Panel, Winton Wetlands Restoration (Australia)

Professor Finlayson has contributed to over 300 journal articles, reports, guidelines, proceedings, and book chapters on wetland ecology and management. He has contributed to the development of concepts and methods for wetland inventory, assessment and monitoring, and undertaken many site-based assessments in many countries.

Mark Everard
International Water Security Network
University of the West of England, Bristol, UK

Mark Everard is associate professor of Ecosystem Services at the University of the West of England (UWE, Bristol) in the UK, as well as a consultant, author, and broadcaster.

Mark has extensive involvement in the development and implementation of ecosystem services and the ecosystem approach since the 1980s. He has particular interests in wetland and water systems, including the many important roles they play in socioecological systems and sustainable or other feedback between human and natural elements of these systems.

Mark's work has included extensive international development work, principally in Africa and India, exploring and helping people optimize their interdependencies with wetlands. He has also served as a policy adviser to UK government around ecosystem and environmental issues, as well as to governments in South Africa, India, and Sri Lanka. However, as the formal policy environment is only as strong as its influence and enforcement, Mark has also worked at local and regional scales particularly in developing countries to learn and out-scale social processes that develop sustainable relationships between people and water resources.

Mark's academic involvement has been extensive, including his most recent role at UWE, and he has also been involved in trustee and advisory capacities with many environmental NGOs. Mark is also a prolific communicator, writing many books, academic papers, and magazine articles targeting a range of scientific, technical, and popular audiences, also making regular contributions to TV, radio, and online media.

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Kenneth Irvine, born in Dublin, has worked on a range of lakes and catchments in Europe and Africa, gaining broad experience of the global challenges facing water and habitat quality. After gaining a Ph.D. in 1987 at the University of East Anglia (UK) for a study on shallow lake food webs, he worked as a nature conservation officer for the UK Nature Conservancy Council, before moving to study ecosystem structure and estimating the secondary production of Lake Malawi in Africa. From there, in 1994 he moved to Trinity College Dublin, Ireland, and spent a decade and a half grabbling with the intricacies of policy and ecology to support the implementation of the EU Water Framework Directive. His alter ego continued to work on the African Great Lakes of Malawi and Tanganyika, and the ecology of the Makgadikgadi salt pans of Botswana. In 2011, he moved to UNESCO-IHE Institute of Water Education in the Netherlands to engage more fully in research and teaching to support capacity development. He heads up the Aquatic Ecosystems Group and their work on, mainly, African wetlands, with other recent work on the capacity development within the Danube basin and for Integrated Water Resource Management in India and Southeast Asia. He continues to learn about the complexities and wicked problems of sustainable use of water and ecosystems.

Robert J. McInnes
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Robert McInnes is an independent chartered environmentalist with over 25 years' experience in wetland-related environmental research, consultancy, and conservation. His main areas of interest in wetlands revolve around three inter-related themes: understanding their biodiversity and the ecosystem services provided to human society; the practical restoration and creation of wetlands for multifunctional benefits; and the development and implementation of wetland conservation and wise use policies and strategies.

He works on wetland-related projects within the UK and overseas and has knowledge extending across a range of wetland types. He regularly publishes articles in peer-reviewed journals, books, and conference proceedings. Prior to working independently, Rob was head of Wetland Conservation at the Wildfowl and Wetlands Trust, UK, and has also worked in ecological consultancy and in academia at the universities of Exeter and London.

Rob has been actively involved with the Ramsar Convention's Scientific and Technical Review Panel since 2008 where he has contributed to the Panel's work on urban wetlands, wetland restoration, wetlands and climate change, and wetland ecosystem services. In addition to undertaking projects on behalf of the Ramsar Convention Secretariat, he has worked for intergovernmental organizations, including UNESCO, CBD, and UN HABITAT, major international NGOS, national and local governments, and private clients.

In addition to his project work, Rob is an active member of the Society of Wetland Scientists (SWS), has been a former president of the European Chapter of SWS, is an associate editor of the society's journal *Wetlands*, and in 2011 was awarded the President's Service Award for the significant contributions he has made in promoting the goals of the society.

Beth A. Middleton
U.S. Geological Survey
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Beth Middleton is a research ecologist with the US Geological Survey's Wetland and Aquatic Research Center in Lafayette, USA. Her biogeographical research focuses on the impact of climate and landuse change on wetlands, particularly forested freshwater wetlands. Her most recent studies are on hydrologic remediation and vegetation response, and she applies those findings to natural resource conservation. Her work has contributed to the understanding of world wetland restoration and global climate change and her book *Wetland Restoration, Flood Pulsing, and Disturbance Dynamics* received the Merit Award of the Society of Wetland Scientists. Her dissertation was on monsoonal wetlands in India (Ph.D. Iowa State University) and was the origin of her later research on the implications of shifts in drought cycles on wetland biodiversity. Her writing is extensive with several books and more than 125 research articles.

She is a member of the graduate faculty at the University of Louisiana and an adjunct professor at Louisiana State University. Before moving to USGS, she was a full professor at Southern Illinois University. Currently, she is a member of several climate change advisory committees and management working groups. She has done extensive research on worldwide wetlands including monsoonal wetlands, baldcypress swamps, peatlands, salt marshes, fens, and mangrove swamps. She is a fellow of the National Conservation Leadership Institute. Her Fulbright work was at G.B. Pant University. She has served as a senior visiting professor with the

Chinese Academy of Science working on wetland dynamics in China and is a Sigma Xi Distinguished Lecturer. Also, she has given several high level addresses including the Earth Day talk for the US Consulate in Chennai, India, and a TEDx talk called "Conservation Oblivion" (www.youtube.com/watch?v=8O72jOgTQPw).

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Anne van Dam is associate professor of Environmental Systems Analysis at the UNESCO-IHE Institute for Water Education in Delft, the Netherlands. He holds a Ph.D. in Agricultural and Environmental Sciences (1995) from Wageningen University in the Netherlands. Before joining UNESCO-IHE in 2003, he worked for the International Center for Living Aquatic Resources Management (ICLARM) and for Wageningen University, in various aquaculture and fisheries research projects in the Netherlands, South and Southeast Asia, Africa, and Latin America. His research interests are in the areas of sustainable aquatic resource management, freshwater wetland ecology and governance, nutrient dynamics, and ecosystems modeling. His publications cover subjects like fish growth modeling, periphyton-based fish production systems, integrated agriculture-aquaculture systems, and wetland ecosystem services. During the last 10 years, he had a leadership role in research and capacity development projects in East Africa, with research focusing on the interactions between livelihood activities (e.g., agriculture) and the ecological functioning of papyrus wetlands, collaborating with universities, government agencies, and NGOs in Kenya, Uganda, and Rwanda. He has represented UNESCO-IHE at the Scientific and Technical Review Panel (STRP) of the Ramsar Convention, is an executive editor of the international journal *Aquaculture Reports*, and was guest editor of a special issue of *Wetlands Ecology and Management* on the ecology and livelihoods of papyrus wetlands.

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Nick Davidson was the deputy secretary general of the Ramsar Convention on Wetlands from 2000 to 2014, with overall responsibility for the convention's global development and delivery of scientific, technical, and policy guidance and advice and communications as the Convention Secretariat's senior advisor on these matters. He has long-standing experience in, and a strong commitment to, environmental sustainability supported through the transfer of environmental science into policy-relevance and decision-making at national and international scales. Nick currently works as an independent expert consultant on wetland conservation and wise use.

Nick has over 40 years' experience of research on the ecology, assessment, and conservation of coastal and inland wetlands and the ecophysiology and flyway conservation of migratory waterbirds, with a 1981 Ph.D. from the University of Durham (UK) on this topic, and continues to publish on these issues. Prior to his Ramsar Convention post, he worked for the UK's national government conservation agencies on coastal wetland inventory, assessment, information systems, and communications and as international science coordinator for the global NGO Wetlands International.

He is an adjunct professor at the Institute of Land, Water and Society, Charles Sturt University, Australia; was presented with the Society of Wetland Scientist's (SWS) International Fellow Award 2010 for his long-term contributions to global wetland science and policy; chairs the SWS's Ramsar Section; is an associate editor of the peer-reviewed journal *Marine & Freshwater Research*; is a member of several IUCN Commissions and their task forces (World Commission on Protected Areas (WCPA), Species Survival Commission (SSC), and Commission on Ecosystem Management (CEM)); and is an honorary fellow of the Chartered Institution of Water and Environmental Management (CIWEM).

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Section I

Introduction

Nick C. Davidson

Introduction to the Wetland Book 1: Wetland Structure and Function, Management, and Methods

1

Nick C. Davidson, Beth A. Middleton, Robert J. McInnes, Mark Everard, Kenneth Irvine, Anne A. van Dam, and C. Max Finlayson

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Abstract

The Wetland Book 1 is designed as a 'first port-of-call' reference work for information on the structure and functions of wetlands, current approaches to wetland management, and methods for researching and understanding wetlands. Contributions by experts summarize key concepts, orient the reader to the major issues, and support further research on such issues by individuals and multidisciplinary teams. *The Wetland Book 1* is organized in three parts - *Wetland structure and function; Wetland management*; and *Wetland methods* - each of which is divided into a number of thematic sections. Each section starts with one or more overview chapters, supported by chapters providing further information and case studies on different aspects of the theme.

Keywords

Wetland structure · Wetland function · Wetland policy · Wetland management · Wetland methods

Introduction

The Wetland Book developed from conversations with the Secretariat of the Ramsar Convention on Wetlands and members of the Convention's Scientific & Technical Review Panel (STRP) on the status and knowledge of wetlands globally, and the need to provide a compilation of wetland information to complement existing, but often scattered sources. *The Wetland Book* is produced as an online and hardcopy publication in two parts, *The Wetland Book:* 1 – Structure and Function, Management, and Methods*;* and 2 – Distribution, Description, and Conservation. The books will support the work of students, transdisciplinary researchers, natural resource managers and agency staff, engineers, planners, policy advisors, NGOs, and environmental consultants.

The Wetland Book 1 is designed as a "first port-of-call" reference work for information on the structure and functions of wetlands, current approaches to wetland management, and methods for researching and understanding wetlands. The contributions by experts aim to summarize key concepts, orient the reader to the major issues, and support further research on such issues by individuals and multidisciplinary teams. Each chapter draws upon multiple sources of information including primary and secondary literature, including publications and reports by government agencies

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and nongovernmental organizations, where appropriate. Chapters are not comprehensively referenced in the style of research papers but rather provide a selection of key sources for finding further information on each topic. Cross-references are provided to help navigate between related chapters in *The Wetland Book.*

Book Structure

The Wetland Book 1 is organized into three parts – *Wetland structure and function; Wetland management;* and *Wetland methods*. Each of these is divided into a number of thematic sections. Each section starts with one or more overview chapters, supported by chapters providing further information and case studies on different aspects of the theme.

For *Wetland structure and function* there are chapters covering: succession (10 chapters); biological adaptations (4); ecological processes and biogeochemistry (7); importance of hydrology to wetlands (8); and landscape ecology (9).

For *Wetland management* there are chapters covering: the international framework (45 chapters); wetland law and policy (42); management of provisioning services (17); management of regulating services (27); management of wetlands for cultural, aesthetic, and associated services (14); and the importance of managing wetland supporting services (6).

For *Wetland methods* chapters cover: wetland delineation and classification (14 chapters); earth observation approaches and their application to wetland ecosystems (17); wetland monitoring and assessment (12); environmental flows (11); wetland management planning (10); wetland restoration and creation (8); environmental impact assessment (9); strategic environmental assessment (3); and economic valuation (6).

The main topic coverage of each part of *The Wetland Book 1* is further summarized below.

Wetland Structure and Function

This part of the Book provides basic information of key importance to the understanding of wetlands. It progresses from large scale ecosystem and landscape concepts to those of a progressively finer scale including biogeochemical, biological, and physiological/anatomical considerations of plants.

Wetland Succession A general chapter on succession, a topic which forms the underpinnings of wetland change and function over time is provided (Middleton 2018) and includes a short chapter on historical ideas of hydrosere (i.e., Clementsian) succession or wetland change over time toward drier land. Later thinking led to the modern Gleasonian perspective, examining the role of disturbance and species life history dynamics in succession. The US $10,000 succession challenge describes the prize offered in the 1950s by the eminent ecologist Frank Egler to anyone who could demonstrate Clementsian succession leading to a climax community. Although the idea of successional stages leading to a climax community has only weak evidence, the concept is still often used in teaching succession concepts to introductory students of biology (e.g., at high school and first year college level). The debate over the nature of succession (Clementsian vs. Gleasonian) has relevance to more recent discussions of self-design versus designer approaches in restoration practice. A number of specific examples explain how anthropogenic disturbance drives vegetation change, for example cattle grazing in wetlands. Whereas fire can be a natural disturbance in wetlands, the human impacts of drainage, cutting and climate change are creating unnaturally destructive peat fires in swamp forest in South-east Asia. Differences in the succession process are apparent in coastal settings, and comparisons are made with fresher water settings.

Landscape Ecology Landscape ecology has emerged relatively recently as a new field in wetland ecology, as broad scale drivers have been increasingly acknowledged as influencing wetland processes (Boudell 2018). Chapters describe various elements of wetland landscape ecology including the concepts of connectivity, corridors, fragmentation, and patch. Process concepts are covered in chapters on disturbance, ecosystem function, and gap and patch dynamics. Species reassembly in riparian and other wetland systems differ because of underlying differences in landscape dynamics in those settings. Practical applications of these concepts are given in chapters on riparian zone buffers, source-sink dynamics, wetland restoration implications, and the heterogeneity of wetland processes as molded by landscape setting. The dynamic nature of wetland processes controlled from a variety of scales lends complexity to the nature of population genetics across wetland landscapes.

Ecological Processes and Biogeochemistry General aspects of chemical transformations in wetlands dictate the nature of other processes (Baldwin 2018). The nature of wetland chemical transformations has important influences on various ecosystem processes, including carbon flux, photosynthesis, primary production, and respiration. Wetland processes related to water quality improvement and habitat support provide vital services to humans, and approaches have been developed to evaluate the economic value of wetlands to human communities. Wetland degradation can negatively impact the extent of ecosystem services provided by wetlands. Therefore, various agencies have implemented programs to better develop policy to protect wetlands including Ecosystem Service Partnerships, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), and the Millennium Ecosystem Assessment.

Hydrology for Wetlands: Importance How water dynamics dictate wetland function is described in Rasmussen (2018). These processes play out rather differently in coastal wetlands because of their proximity to the ocean. Due to the complex processes associated with various water inputs and outputs, hydrologic modeling helps frame our basic understanding of wetland hydrology. A specific case of hydrologic dynamics as it relates to treatment performance in constructed wetlands is provided for the Everglades region in Florida, USA.

Biological Adaptations Finer scale considerations of wetlands are summarized, including the anatomical (Seago 2018), morphological (Schaffer 2018), and physiological (Kandalpas 2018) adaptations of plants to life in an aquatic environment. Some aquatic species succeed in environments that terrestrial species would find harsh, by engaging in symbiotic relationships with microbes.

Wetland Management

Information provided on wetland management will be useful to audiences interested in approaches for managing wetlands, including how these approaches have changed over time with the development of international frameworks and newer concepts, such as the developing focus on ecosystem services and the adoption of the 2030 Sustainable Development Goals (SDGs). The importance of legal structures and policy settings is also explored.

Overview of Wetland Management Management of wetlands is complex, needing to address both the pressures imposed on wetlands by wider landscape change as well as more direct *in situ* pressures and measures needed to retain a wetland's natural character and ecosystem services. There is a long history of wetland management, for example to protect favored birds and other species or for fishery, recreational, and other uses. However, in the main, wetland management has not historically been undertaken with conscious efforts to optimize the multiplicity of interconnected ecosystem services providing benefits to diverse sectors of society and underpinning system resilience. Everard (2018a) explores the need for systems scale thinking in wetland management, ensuring that management maximization of single or narrow subsets of benefits does not occur at net cost to wider system functioning and optimization of human well-being across a range of linked ecosystem services.

International Framework for Wetland Conservation and Wise Use Such is the importance of wetlands and their ecosystem services that they are the subject of a wide range of international conventions and agreements and others mechanisms for protection and positive management (reviewed by Gardner 2018). These include biodiversity-related conventions, of which the Ramsar Convention on Wetlands is an international exemplar (Davidson 2018). The multiple values they provide to society also mean that wetlands have been a focus of the Convention on Biological Diversity, as well as making important contributions towards achieving global biodiversity targets.

Wetlands also have significant international ramifications, both as transboundary systems and as networks of habitats vital for migratory species. For this reason, a range of transnational and regional legal frameworks has been established, particularly with respect to integrated management of migratory bird flyways (Davidson and Stroud 2018). Regional legal frameworks have also been developed for the comanagement of significant international and interregional wetlands such as Lake Chad and the Murray-Darling Basin, Australia.

Significant implications for wetland management also arise from global challenges such as climate change and the commitment by UN signatories to the 2030 Sustainable Development Goals (SDGs).

A range of nongovernmental organizations (NGOs), institutions, and programs, across a range of scales, have particular focal interests in wetland conservation and management including, for example, BirdLife International, the International Crane Foundation, the International Peat Society, the Society of Wetland Scientists and Wetlands International. Some institutions have government backing and support for significant wetland rehabilitation programs, such as South Africa's national wetland rehabilitation program called *Working for Wetlands*.

Wetland Law and Policy Wetland management needs and initiatives have given rise to a diversity of wetland law and policies (Gardner and Finlayson 2018), including a wide range of national wetland policies (Bonells 2018). These policies encompass a range of principles including "no net loss," different types of permit schemes, the "avoid-mitigate-compensate" sequence and mitigation banking. Management tools such as Environmental Impact Assessments and Strategic Environmental Assessments support impact assessment and mitigation measures.

Economic values as well as diverse non-market benefits associated with wetland ecosystem services mean that management occurs within economic, ecological, and social contexts. Economic incentives may be significant for the nonregulatory conservation and management of wetlands (Watts and Everard 2018). Financial exchanges may have a role to play in incentivizing land owners to retire land or to manage in ecosystem-sensitive ways; examples include the US Conservation Reserve Program and the Payments for Ecosystem Services (PES) approaches. Establishment of national wetland policies is a strategic priority for signatories to the Ramsar Convention. Consequently, many countries have developed wetland policies and examples are described from Australia, Ghana, Chile, and the USA.

Management of Provisioning Services Management of wetlands for provisioning ecosystem services can encompass a diversity of outputs (Everard 2018b) including, for example, wetland food (such as lake bed cropping), reed products, traditional medicines, and salt production.

Management of Regulating Services Wetland management for regulatory services (McInnes 2018) also encompasses a diversity of benefits including support of insect pollinators, climate regulation, coastal protection, and urban wastewater management.

Management of Cultural Services Cultural services as foci for wetland management (Papayannis and Pritchard 2018) cover a diversity of benefits such as protection of archaeological resources, religious interests, educational resources (including visitor centers), and inspiration for the arts.

Importance of managing wetland supporting services Often overlooked, certainly by traditional economic assessment as they are not directly "consumed" in the economy, management of wetlands for their supporting services is essential for maintaining ecosystem integrity, functioning, and a reliable flow of other services (Everard 2018c). These services include nutrient cycling, biodiversity support, and generation of soils.

Wetland Methods

This part of the Book will be useful to a diverse audience. For those who work with wetlands, it can serve as an entry point to more detailed technical publications on the practical application of these methods. For students and decision makers, it will support a broad understanding of how wetlands are measured, mapped, and valued; how changes in wetlands can be assessed; and how planning for management and protection can be improved.

Wetland Delineation and Classification With the introduction of formal protection of wetlands by means of policies and laws, the need for formal delineation, mapping, and inventory arose. Tiner (2018) describes how this is done in the USA, a country with one of the longest traditions in wetland identification and delineation. Three types of indicators (on hydrophytic vegetation, hydric soil, and wetland hydrology) can be used to characterize wetlands. Aerial photography and, more recently, geographical information systems and satellite imagery play a large role in incorporating these indicators into the maps of wetlands.

Another need arising from formal protection and management procedures is the classification of wetlands (Gerbeaux et al. 2018). A variety of classification systems for wetlands has evolved, based on different characteristics of the wetland such as soil or vegetation type, geomorphology or nutrient status. Several chapters present different classification systems for specific types of wetlands (e.g., coastal wetlands, peatlands, and estuaries) or for specific countries (e.g., Brazil, Canada, India, South Africa, and the USA). A separate chapter describes the Ramsar Convention's classification system.

Earth Observation Methods for Wetlands In this section earth observation approaches, and their application to wetland ecosystems, are covered. An overview is given by Lucas (2018a). Since the 1970s, sensors operating in the visible and infrared regions of the electromagnetic spectrum (e.g., the Landsat satellites) have provided valuable imagery for characterization of wetlands and their dynamics. More recently, sensors in the thermal and especially the microwave region of the spectrum have also become important, especially because the latter (e.g., Synthetic Aperture Radar, SAR) can observe changes in water dynamics regardless of cloud cover or overhanging canopies. Separate chapters provide more in-depth insights into different sensor types (Lucas and Costa 2018) and the electromagnetic spectrum (Lucas 2018b) and into the use of remote sensing for observing water persistence and duration (Milne 2018). Other chapters focus on remote sensing applications for a variety of wetland types (arctic and boreal wetlands, tropical forest wetlands, seagrasses, mangroves, peat swamps, subtropical wetlands, temperate bogs, mires, and fens) and human activities related to wetlands (aquaculture, agriculture).

Wetland Monitoring and Assessment One of the assumptions of protection and management is that information on the state of the wetland, and especially on the characteristics of the ecosystem which are part of management objectives, is available. This leads to the need for monitoring and assessment of wetlands (Stratford 2018a). Different types of assessment (hydrological, biological, functional, integrated, and vulnerability assessment) are discussed. While assessment is often a "snapshot" observation, monitoring (Stratford 2018b) provides more long-term data on the changes of the wetland, often based on very specific monitoring objectives (e.g., related to pressures on the ecosystem or to the effectiveness of management measures).

Environmental Flows The extraction of water from aquifers, rivers, lakes, and other wetlands for human use and the observed impact on ecosystems have led to the realization that there is a trade-off between the quality of ecosystems and the benefits of water for humans. The environmental flows concept is introduced by Acreman and Arthington (2018) and deals not only with the quantity of water available for ecosystems but also with water quality and the timing of flows. This is a complex issue, as it involves the challenge of determining the water requirements of the ecosystems, the practical questions around safeguarding these flows, as well as the policies and management strategies needed for implementing them. Subsequent chapters discuss various frameworks developed over the past decade for quantifying and realizing environmental flow regimes with examples of their implementation, and place the environmental flow concept within the context of integrated water resources management.

Wetland Management Planning As a result of increasing awareness of the importance of wetlands, and supported by the Ramsar Convention, increasing numbers of countries have formulated and adopted wetland conservation policies (see chapters in the "Wetland Law and Policy" part). The impact and effectiveness of such policies depend crucially on how they are translated into actual management on the ground. For this, management planning is extremely important. Alexander (2018) introduces the planning process which starts with the formulation of a clear vision and objectives, and participation of all involved in the wetland site. Other chapters discuss the planning process and which tools can be used. Emphasis is on adaptive management which allows frequent reassessment of objectives and actions to adjust the process to changing circumstances and needs. A separate chapter covers approaches to increase the capacity of countries to implement the management planning process, and several case studies are presented to support this topic.

Restoration and Creation of Wetlands Sometimes it is not possible to protect wetlands from degradation or destruction, and increasingly efforts are being made to restore degraded wetlands and even create new ones. Moreno-Mateos (2018) shows that, although restoration cannot bring back the original ecosystem, it is possible to restore some of the important ecosystem functions and services that the original wetland system provided. Restoration can lead to artificial new wetland ecosystems ("constructed wetlands"), to brand new natural wetlands which are allowed to develop independently ("created wetlands"), or to the recovery of existing wetlands from earlier anthropogenic impact. Chapters discuss the ecosystem aspects of restoration (e.g., water quality, succession and plant communities, carbon and nutrient cycling), policy-related topics, and the economics of wetland restoration.

Environmental Impact Assessment for Wetlands Human development and economic activities create pressures on nature that, through the years, have led to the degradation and destruction of ecosystems all over the world. Concerns over negative impacts on the environment (including wetlands) in the 1960s led to the development of Environmental Impact Assessment (EIA) as a process to influence decision-making about development projects and to mitigate negative effects on not only ecosystems but also on society in a broader sense. Slootweg (2018a) introduces EIA and its development, including how it plays a role in policy and practice for wetland protection and conservation. In the following chapters, the various stages in the EIA process (screening, scoping, assessment and evaluation, reporting, and decision making) are explained further. Separate chapters are devoted to mitigation and to health and social impact assessment, as well as to the vital need for the participation of stakeholders in the EIA process.

Strategic Environmental Assessment for Wetlands While the EIA process is usually applied at the project level, Strategic Environmental Assessment (SEA) has been developed as a set of tools to identify and address the environmental and social impacts of higher-level initiatives such as policies and programs. SEA became important in the 1980s and 1990s, and Slootweg (2018b) describes its importance for wetlands, as demonstrated by the fact that EU member states have had since 2006 a legal obligation to apply SEA to their development plans and programs. Further chapters discuss the background and scope of the SEA process, including what is needed to implement the SEA process effectively for improved and more integrated decision-making for wetlands and their resilience.

Economic Valuation of Wetlands One of the topics that has received a lot of attention during the last decade is the economic valuation of ecosystems. Propelled by initiatives such as the Millennium Ecosystem Assessment and the "*The Economics of Ecosystems and Biodiversity*" (TEEB) economic valuation of ecosystem services has become part of the mainstream in policy and decision-making for wetlands. Kumar (2018) explains the background principles of economic valuation and highlights some of the challenges, e.g., of upscaling of values obtained from limited studies and uncertainty related to the limitations of the methods used for valuation. Subsequent chapters deal with the total economic value of wetlands and provide more details on the different valuation methods. Two separate chapters discuss market-based and policy-based instruments to incorporate the economic value of wetlands in policies for wetland conservation and protection. A final chapter presents a number of examples of wetland valuation studies.

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Section II

Wetland Succession

Beth A. Middleton

Succession in Wetlands

2

Beth A. Middleton

Contents

Abstract

Succession refers to the change in vegetation over time driven by disturbances and the maturation of plant species. In wetlands, these disturbances include water and salinity level changes along other factors that can alter vegetation. The historical view of succession (Clementsian) was that vegetation change represented the linear progression of through stages of vegetation toward a climax state. These stages were thought to be comprised of species that were interlocked with each other. These days the idea that succession is represented by the successive replacement of highly related sets of communities over time has been deemphasized, in favor of the idea that species in the community act more independently of one another (Gleasonian). An important example of this Gleasonian perspective model has been developed for prairie wetlands of North America by van der Valk. In this view, succession proceeds in a cyclic fashion, with flooding and drought driving changes in specific species, so that the individualistic species responses to water regime and other disturbances drive changes in the system (Environmental Sieve Model). The succession of

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many other world wetlands types is thought to occur in a similar way. These recent ideas of succession emphasize that species that are able to regenerate after disturbance via seed banks and propagules, and that the nature of post-disturbance regeneration is the most important determinant of later succession (initial floristics). Notably, the idea that lakes and bogs represent an early state of succession, and that depressions fill in to become dry land (terrestrialization) has little evidence. With climate change, wetlands are likely to have altered successional trajectories, particularly as these ecosystems become exposed to different climatic temperatures, flooding/drought cycles, salinity intrusion and increased CO_2.

Keywords
Clementsian succession · Gleasonian succession · primary succession · secondary succession · relay floristics

Definitions

Allogenic succession is the change in ecosystems related to abiotic influences. Environmental constraints are thought to be the most important delimitors of succession.

Autogenic succession is the change in ecosystems due to biotic influences.

Clementsian succession is the idea that vegetation in communities changes over time through successive stages by allogenic and autogenic processes until a climax stage is reached. Although the idea is often portrayed in textbooks, it is not widely held by ecologists. As an example of an ecosystem that does not have seral stages, baldcypress trees establish after seed germination in the light gaps of tree falls or old fields. Once established, the trees can live for centuries.

The climax stage is the final outcome of succession following the ideas of Clementsian succession.

Preemption is a common type of competition where individuals that occupy a space make it difficult for individuals of other species to invade that space. Preemption is likely the most important type of competition, although resource competition is often the focus of studies.

A functional group is a set of organisms with similar traits, roles, and responses to disturbances in ecosystems. The response of the functional group can be considered as typical of the set of organisms.

Perturbation is very similar to the term “disturbance” for most ecologists. Some ecologists define perturbation as a disturbance that is not natural in the ecosystem, e.g., a human disturbance.

Resilience is the relative ease of ecosystem recovery after disturbance. Some systems are repeatedly impacted by hurricanes, salinity intrusion, or

(*continued*)

Forest change in subarctic environments may be limited by freezing and flooding. Pictured is a peatland in the Sanjiang Plain of northeastern China (Photo by Beth Middleton)

other disturbances but quickly return to their original ecosystem type and function.

Persistence is the existence of an ecosystem type despite repeated disturbances. Gleasonian succession is the idea that vegetation in communities changes over time in response to disturbance. Following this idea, succession is viewed as the response of species to environmental conditions with respect to maturation and fluctuation. Hydrarch succession is the idea that a lake can fill in to become dry land. This old concept is challenged by most ecologists, who more likely favor the idea that left to natural succession, once a wetland, always a wetland. Also, it is more likely that dry land will become a wetland through paludification than that a wetland will become dry land through hydrarch succession.

Primary succession is vegetation establishment on a site that has not had vegetation before e.g. a site with volcanic outfall or a new shoreline.

Secondary succession is vegetation change on a site where vegetation has occurred before e.g. old field succession community or ecosystem change over time.

(*continued*)

Old growth baldcypress in Cat Island National Wildlife Refuge near St. Francisville, Louisiana. Trees of this species are the first to germinate and establish on the edges of swamps in old field settings. Pictured is Evelyn Anemaet (Photo by Beth Middleton)

Chronosequence is the concept of space-for-time substitution, which attempts to demonstrate that zonation patterns lying along a gradient demonstrate succession over time. This approach has been largely invalidated.

The Environmental Sieve Model is the idea that species may be blocked (sieved) from a site because of a lack of tolerance for the environment by various life history stages of a species. In wetlands, the water tolerance of the life history stages of species is important in determining successional outcome.

Facilitation is the idea that certain species can colonize harsh environments, and thereby enable the colonization of other species.

Initial floristics refers to the idea that the species originally establishing in a site after disturbance set the long-term successional pattern. Following this idea, after establishment, vegetation change is thought to be mainly an outcome of the maturation of the established individuals. Generally, the majority of species that come to occupy an ecosystem are present immediately after the disturbance.

(*continued*)

After species establish on a site, it is difficult for others to establish because they are preempted from occupying the space. Pictured is Evelyn Anemaet standing inside of the trunk of an old growth baldcypress tree, Cat Island National Wildlife Refuge, St. Francisville, Louisiana (Photo by Beth Middleton)

Life history attributes are the traits that lead species to respond in particular ways to environment as based on the characteristics of each life stage (i.e., seed set, dispersal, germination, and growth of seedlings and adults).

Paludification is the process by which dry land becomes a peatland following the invasion of hydrophilic mosses. After the invasion of these mosses, the water table rises, and subsequent drainage is impeded by the accumulation of peat.

Pioneer species are those that establish quickly in communities after disturbance.

The reductionist concept of succession places an emphasis on the role of species traits in ecosystem processes e.g. the emphasis on individual life history characteristics in Gleasonian succession.

(*continued*)

Tupelo and baldcypress (*left and right*, respectively) are tree species of the same functional group, which occupy similar habitats and respond similarly to environment. Pictured is a swamp forest in Cat Island National Wildlife Refuge, St. Francisville, Louisiana (Photo by Beth Middleton)

Relay floristics describes succession as a process characterized by a sequence of stages (successional seres), each successively creating the conditions necessary for the following stage. This idea of succession emphasizes autogenic processes, that is, that the organisms themselves are changing the conditions on the site.

A safe site is a place where seed germination and species regeneration can be successful following disturbance. Light gaps in a forested wetland may provide a place for seed germination for some plant species depending on environmental constraints such as flooding and salinity level.

A seral stage is one of a sequence of successional stages thought to prepare the way for subsequent stages in Clementsian succession. Seral stages have been largely discounted in successional theory. Instead, the idea that each species establishes and matures independently of others during succession is currently more favored among ecologists.

(*continued*)

Pictured is a fire in a wetland in Rajasthan, India (Photo by Beth Middleton)

Stability in an ecosystem reflects its ability to return to its original condition after disturbance.

The steady (stable) state is a relatively unchanging vegetation status over time.

The superorganism concept is the idea that a community is an entity with unit characteristics more important to function than the individual species comprising the community, i.e., the sum is larger than the parts. In succession, this idea corresponds to the notion that seral stages are characterized by species that move together as a unit through time and space.

Terrestrialization is the filling-in of a lake to become a bog. In actuality, the process of paludification is more likely to turn dry land into a wetland.

Water regime refers to the water dynamics of a wetland ecosystem (or experiment) including water level, periodicity, and flow.

Introduction

Succession refers to species change over time as related to the maturation and fluctuation of species (van der Valk 1981). In wetlands, these changes are driven by natural disturbances such as water and salinity level fluxes, herbivores, fires,

Resilient ecosystems return to their original state after a specific type of disturbance. Pictured is a mix of *Phragmites* and baldcypress swamp species along the Pocomoke River after Hurricane Sandy. Note that some of the baldcypress trees to *left* appear to have died (Photo by Beth Middleton)

windstorms, hurricanes, and earthquakes. Historically, the view was that natural disturbance interrupted the progression of succession toward a climax state and that each successional (seral) stage was comprised of a set of interlocked species (Clements 1916). This successional progression was viewed as a product of largely autogenic processes on the part of this unit of interlocked species. Over time, these species were thought to fill the open water of the basin with organic materials until the wetland became dry land, which was seen as a largely autogenic process. This Clementsian idea of succession is no longer wholly accepted. In fact, notions of regional climax, terrestrialization, and seral stages of communities have been deemphasized in modern views of succession (Niering 1987).

Drivers of Succession

These days, a widely held idea of succession is that disturbance drives successional changes in plant communities and that the species in the community act independently of one another (Gleason 1926). A succession model following

Primary succession occurs in places where vegetation has not occurred before, for example, on rocks along new shorelines. Pictured is a shoreline of Hudson Bay, near Wemindji, Quebec, Canada (Photo by Beth Middleton)

these Gleasonian ideas was developed for prairie pothole wetlands in North America (van der Valk 1981) and monsoonal wetlands in Australia and India (Finlayson 1991 and Middleton 1999, respectively). The van der Valk model of wetland succession proceeds in a cycle that maintains wetlands over many millenia, with the key drivers of succession related to disturbance (Fig. 1(1, 2)). The cycle proceeds as vegetated freshwater marshes are destroyed by disturbances such as high water and herbivory. After the vegetation is killed, the wetland becomes an open marsh. The open marsh is maintained for many years because the emergent vegetation cannot germinate in flooded conditions, preventing the development of new emergent vegetation. Emergent vegetation reappears in the wetland during drought, which recurs at 5–25 year intervals. During drought, the wetland draws down resulting in the reemergence of mudflats. In these mudflat conditions, emergent (e.g., cattail) and other mudflat species can germinate and reestablish (Fig. 2). When normal rainfall resumes, the emergent species regrow into the water column (regenerating marsh stage). Eventually, high water and herbivores remove the emergent vegetation (open marsh stage), and the cycle begins again (Fig. 3).

Similar succession patterns occur in other freshwater wetlands around the world, with the timing of drought/reflooding patterns key to the timing of vegetation reestablishment. Monsoonal wetlands in India, Africa, Australia, and South America

Zonation patterns in wetlands do not necessarily represent succession over time. Pictured are zones of *Phragmites* and mixed swamp forest along the Pocomoke River in Maryland (Photo by Beth Middleton)

often have annual cycles of drought/reflooding, so that the succession cycle occurs in a shorter time frame (Fig. 4) (Middleton 1999).

Within the context of hydrologic dynamics in relationship to the establishment and maintenance of wetland species, multiple steady states may persist in wetlands such as the Everglades (Zweig and Kitchens 2009). Furthermore, the growth of the runners of certain salt marsh species in bare areas demonstrate the facilitation of the colonization of less tolerant species because the facilitating species make conditions less harsh (e.g., lowers salinity; Crain et al. 2008).

Many species establish immediately after disturbance, so that subsequent changes in vegetation may be because of the development of taller species, which are a part of the initial floristics of a site (Niering 1987). The initial floristics concept suggests that most species are present as seeds or propagules in the soil after disturbance (Niering 1987). Shrubs and trees may not be apparent in the early stages of secondary succession, but later, these taller species develop and dominate the community. A good example of the importance of initial floristics during secondary succession is on Alaskan floodplains. Pioneer species (willow, poplar, alder) and late-forest species (spruce) are both present on these floodplains almost immediately after

Tree species typically cannot establish in permanently flooded conditions e.g. baldcypress seeds only germinate in unflooded environments. Pictured is the impounded Buttonland Swamp in southern Illinois with little tree regeneration (Photo by Beth Middleton)

agricultural abandonment, but the late-forest species become dominant later. Also note that primary succession is similar to secondary succession, except that primary succession occurs on sites where vegetation has not occurred before, e.g., lake margin of newly formed volcanic crater.

Seral Stages or Not

Wetland zonation patterns sometimes are used to demonstrate seral stages of wetlands, but this idea has also been challenged (Niering 1987). Chronosequences or space-for-time substitutions have been invalidated with analyses examining long-term vegetation studies, palynology, and other related approaches. These studies show that the inferred seral stages are not related to vegetation change over time (Johnson and Miyanishi 2008). Changes in the structure of the wetland zones around lakes, bogs, and tidal marshes actually are the product of water level changes related to climate or edaphic factors, including beaver activities (damming and tree cutting) (Niering 1987).

After Hurricane Katrina, many coastal wetlands returned to their prehurricane condition. Pictured is a brackish marsh a few weeks after Hurricane Katrina along the Pearl River in Mississippi (Photo by Beth Middleton)

That lakes or bogs could infill was once used as an example of seral stages in succession (Klinger 1996). For example, it was thought that seral stages including *Sphagnum* and a series of other seral stages could bring the site to an ever-drier state toward an upland forest (terrestrialization). Instead, many studies show that once a bog forms, it is likely to remain as a bog for thousands of years. It is more likely that dry land will become invaded by *Sphagnum* by paludification and become a bog. Therefore, a bog should be viewed as the end point of vegetation change (Klinger 1996).

Disturbance Models and Succession

Disturbances act on the vegetation by removing the dominant vegetation, so that disturbances can maintain biodiversity in wetlands by giving nondominant species a window of opportunity to grow. The species that can occupy a wetland after a disturbance depend on the environmental conditions related to water/salinity regime, soil conditions, and other environmental constraints. Species have very specific abilities to establish after disturbance depending on their life history characteristics. The environmental sieve model suggests that the species composition of a wetland

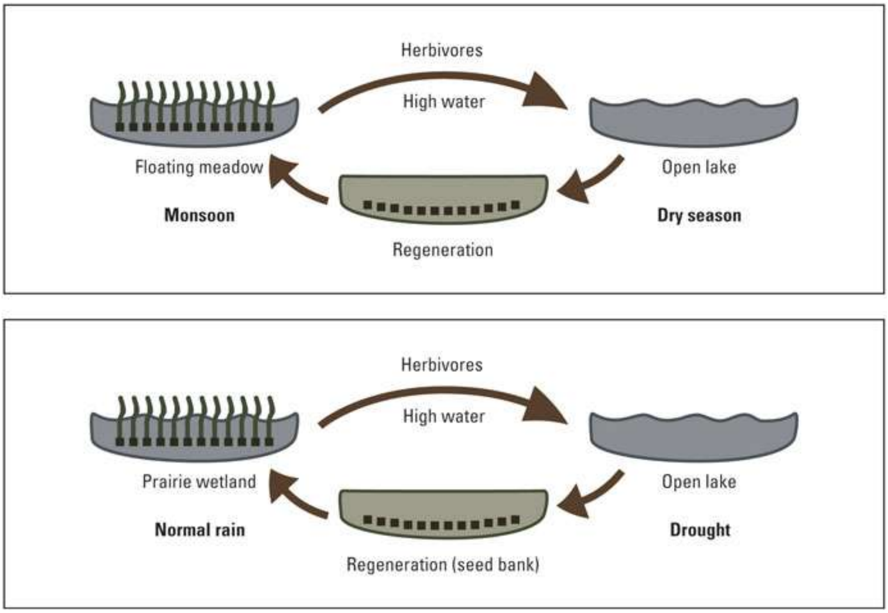

Fig. 1 (**1**) Monsoonal wetlands in northern India flood during the monsoon (October through March) and drawdown during the dry season (April through July), thereby completing a vegetation cycle in 1 year. If the monsoon fails, the wetland stays dry until the next monsoon, unless water floods the floodplain from the river (van der Valk 1981). (**2**) The vegetation cycle in prairie potholes in North America occurs over a 5–25 year interval and is also driven by periodic drought. Some species of monsoonal wetlands and prairie potholes live for decades in the seed bank, which preadapt these species to survive any long-term drought related to climate change (Middleton 1999)

depends on the life history requirements of the species and their adaptation to the specific environments of wetlands. Certain species are periodically "sieved" for a time from a wetland depending on the conditions in the wetland and the tolerances of the species for those conditions (van der Valk 1981).

Models have been developed to predict which species will grow in response to flooding versus drawdown (Figs. 4, and 5(1, 2); van der Valk 1981; Middleton 1999, Finlayson et al. 1991). The main determinants for species development following disturbance depend on the ability of the life history stage of a species (seed, seedling, adult) to grow in various water regimes (flooded versus drawndown) and if the species is annual or perennial (Fig. 4). Species of similar types transition in response to flooding versus drawn down conditions in wetlands, although the particular species may vary regionally. For example, in communities of native rice, *Oryza* is an annual species, which establishes in standing water (AS-II). The particular species may vary by region or continent (e.g., Australia and India: *Oryza australiensis* vs. *Oryza rufipogon*, respectively; Fig. 5(1, 2). (Finlayson 1991 and Middleton 1999, respectively).

Fig. 2 Regeneration of species from the seed bank in a drawndown mudflat in a monsoonal wetland in the Keoladeo National Park, India (Photo by: Beth Middleton)

Fig. 3 Open water in freshwater marsh in the Okavango, Botswana. Large herbivores such as hippopotamus maintain hemimarsh in some African wetlands (Photo by: Beth Middleton)

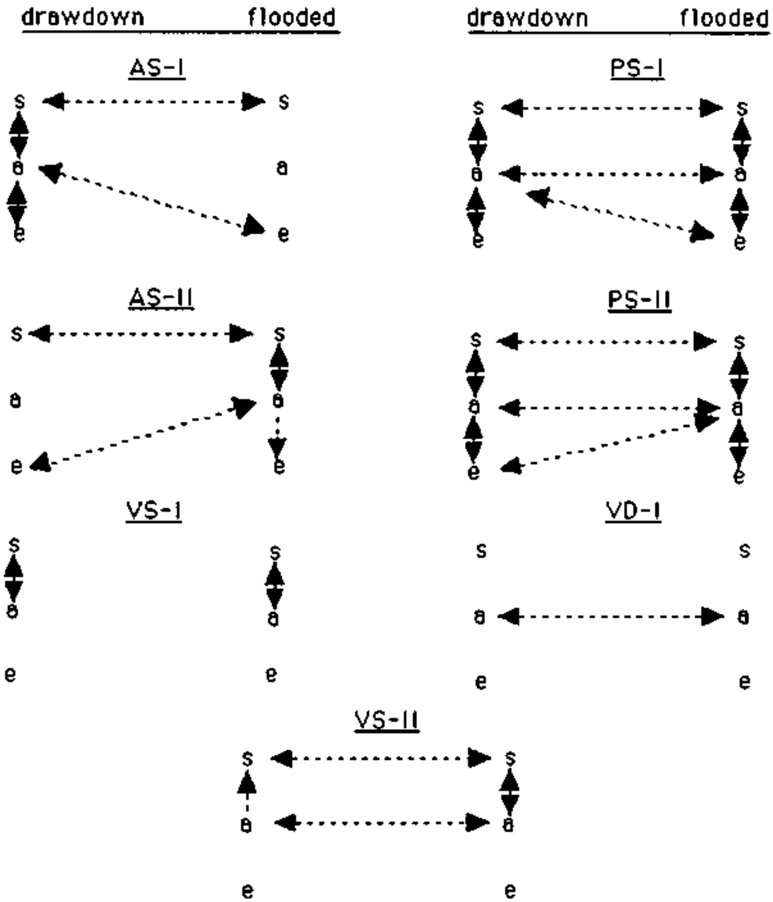

Fig. 4 Predicted transitions in flooded and drawdown states (increasing versus decreasing, respectively) in monsoonal wetlands. Transitions are represented by *solid* and *dashed lines* between environmental states (within versus between states, respectively). The species types are: s = present as long-lived propagules in a persistent seedbank; *a* = mature adults; and *e* = locally extinct. Other letters and symbols are defined in Fig. 5 (From Finlayson 1991, reprinted with permission of the author)

Future Challenges

Wetland ecosystems altered by climate change in the future may experience different successional trajectories than the current ones, particularly if climates change as predicted by the IPCC (2007), so that these ecosystems are exposed to increased CO_2, temperature, salinity, or flooding-drought episodes.

1. Australia, Northern Territory (Magela floodplain)

DRY SEASON		→→→ flooding	WET SEASON		→→→ drawdown
no standing water			standing water		

<u>1.a. *Pseudoraphis* Community</u>

AS-I	*Cyperus* spp.	AS-II	*Blyxa**
	Fimbristylis		*Hygrochloa**
	Glinus		*Najas**
	Heliotropum		*Nymphoides* spp.*
			Utricularia spp.
VS-I	*Polygonum*	VS-I	*Polygonum*
	*Pseudoraphis**		*Pseudoraphis**
PS-I	*Mimosa*	VS-II	*Eleocharis* spp.*
			Nymphaea *
		VD-II	*Salvinia* spp.*
		PS-I	*Mimosa*

<u>1.b. *Hymenachne* Community</u>

VS-I	*Ludwigia*	AS-II	*Aeschynomene* spp.
	Pseudoraphis		*Oryza*
VS-II	*Azolla*	VS-I	*Ludwigia*
	Eleocharis spp.		*Pseudoraphis*
	Hymenachne *		
	Lemna	VS-II	*Azolla*
	*Nelumbo**		*Eleocharis* spp.*
	Nymphaea		*Hymenachne**
	Urochloa		*Lemna*
			Nelumbo
VD-II	*Salvinia*		*Nymphaea*
			Urochloa
		VD-II	*Salvinia*

<u>1.c. *Oryza* Community</u>

AS-I	*Coldenia**	AS-II	*Aeschynomene spp.*
	Commelina		*Blyxa* spp.
	Digitaria		*Hygrochloa*
	Heliotropum		*Ipomoea*
	*Phyla**		*Maidenia*
			Nymphoides spp.
VS-I	*Ludwigia*		*Oryza**
	Pseudoraphis		*Utricularia* spp.
PS-I	*Mimosa*	VS-I	*Ludwigia*
			Pseudoraphis
PS-II	*Isoetes*		
		VS-II	*Eleocharis* spp.*
			Nymphaea
		VD-II	*Salvinia*
		PS-I	*Mimosa*
		PS-II	*Isoetes*

Fig. 5 (continued)

2. India, North-Central (Keoladeo National Park floodplain)

DRY SEASON →→→ flooding		WET SEASON →→→ drawdown	
no standing water		standing water	

2.a *Paspalum* Community

AS-I	*Elatine triandra*	AS-I	*Glinus oppositifolius*
	Rotala indica		*Nymphoides* spp.+
	Ludwigia perennis		*Potentilla supina*
	Eleocharis atropurpurea		*Gnaphalium polycaulon*
	Nymphoides spp. +		*Limnophila indica*
			Hydrolea zeylandica
PS-I	*Acacia nilotica**	VS-I	*Paspalum distichum**
VS-I	*Paspalum distichum**		*Paspalidium punctatum**
	*Paspalidium punctatum**		*Ipomoea aquatica**
	*Ipomoea aquatica**		
		AS-II	*Hydrilla verticillata*
			Najas graminea
		VS-II	*Nymphaea nouchali**

2.b. *Hydrilla* Community

AS-I	*Ammannia sessilus*	AS-I	*Alternanthera sessilus*
	Echinochloa crus-galli		*Blumea obliqua*
	Elatine triandra		*Dactyloctenium aegyptium*
	Eleocharis atropurpurea		*Glinus oppositifolius*
	Lindernia parviflora		*Gnaphalium polycaulon*
	Nymphoides spp.+		
		AS-II	*Najas graminea**
PS-I	*Hemiadelphus polyspermum*		*Vallisneria spiralis*
			*Potamogeton nodosus**
PS-I	*Scirpus tuberosus*		*Sagittaria guayanensis*
		VS-II	*Nymphaea nouchali*
			*Hydrilla verticillata**

2.c. *Oryza* Community (Ecotone with *Acacia nilotica* savanna)

AS-I	*Aeschynomene indicum*	AS-I	*Alternanthera sessilus*
	Ammannia auriculata		*Gnaphalium polycaulon*
	Bergia ammannioides		*Laggera aurita*
	Caesulia axillaris		*Melochia corchorifolia*
	Cyperus difformis		*Nothosaerva brachiata*
	Echinochloa crus-galli		*Nymphoides* spp.+
	Scirpus supinus		*Polygonum plebeium*
			Polypogon monspeliensis
PS-I	*Cyperus rotundus*		*Potentilla supina*
	Hemiadelphus polyspermus		*Rumex dentatus*
PS-I	*Acacia nilotica**	VS-I	*Paspalum distichum**
VS-I	*Cynodon dactylon/Sporobolus helvolus**		*Typha angustata**
		AS-II	*Ceratophyllum demersum*
			Hydrilla verticillata
			Limnophylla indica
			Najas graminea
			Oryza rufipogon
			Potamogeton nodosus
			Sagittaria guayanensis
			Vallisneria spiralis
		VS-II	*Nymphaea nouchali*
		PS-I	*Acacia nilotica**

Fig. 5 Predicted species succession due to seasonal water level changes in monsoonal wetlands in (**1**) north-west Australia and (**2**) north-central India. Symbols are as follows: A = annual, V = vegetative, P = perennial, S = short-lived propagule, D = long-lived propagule, I = propagules established in areas without standing water, II = propagules established in standing water, "+" establishes in areas with and without standing water (Based on Finlayson 1991a, reprinted by permission of the author as rendered in Middleton 1999)

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Environmental Sieve Model of Wetland Succession

3

Arnold van der Valk

Contents

Abstract

In the first environmental sieve model, water level (standing water, no standing water) determined which species could become established and which would be extirpated at a given time in a wetland. Species attributes were used to predict what would happen to each species in a wetland as water levels changed. The three attributes used were: (1) life expectancy (annual, short-lived perennial, and perennial with vegetative growth), (2) seed dispersal (seeds present in seed bank, seeds had to be dispersed to wetland), and (3) seed germination requirements (seeds can germinate underwater or seeds can germinate only on wet soil). The model was tested successfully in a long-term, water manipulation study carried out in the Delta Marsh in Canada, the Marsh Ecology Research Project (MERP). More sophisticate versions of this type of model have been developed that can make quantitative predictions about changes in the composition of wetland vegetation over time.

Keywords

Assembly rule model · Succession · Vegetation composition · Vital attributes · Water-level fluctuations

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Introduction

What causes temporal changes in the species composition of vegetation over time (succession) has been one of the central problems in ecology since its origins as a science in the late nineteenth century. German ecologists were the first to postulate that it was the morphological, anatomical, and physiological characteristics of plant species that adapted them to a specific climate or micro-environment (van der Valk 2011). Although this approach provided an explanation for why some species are found in the tropics and others in the arctic and why vegetation changes when climates changed over geologic time, it did not explain how a given species became established or why it might disappear from a particular site over much shorter time intervals. To understand this required additional knowledge about the characteristics of species in the local flora that was ecologically rather than geographically and geologically relevant (Grime 1979): how a species is dispersed, the conditions under which it can become established, its life expectancy, its growth rate, how it responds to various kinds of disturbances, etc. These ecologically relevant characteristics of plant species came to be called their vital attributes (Noble and Slatyer 1980). One of the first attempts to use this approach for understanding and predicting vegetation change was developed in the 1950s by Frank E. Egler (1954). He proposed that light requirements for seed germination and seedling growth were largely responsible for the changes in vegetation observed during old-field succession in New England. This approach was expanded in the 1960s, 1970s, and 1980s by terrestrial ecologists (Grime 1979; Noble and Slatyer 1980) and applied to wetlands by van der Valk (1981). The use of community assembly rules (Weiher and Keddy 1999; Toth and van der Valk 2012) to predict vegetation composition and change is a closely related approach.

Vital Attributes

In wetlands, many vital attributes of species have been identified that can be used to predict when they can become established and when they will be extirpated. These include seed dispersal (present in seed bank, not in the seed band (dispersal dependent)); seed establishment requirements (germinate only on mudflats, germinate only under water, will not germinate in shade, etc.); types of vegetative growth (long vs. short rhizomes, turions, terrestrial form, etc.); and life expectancy (annual, short-lived perennial, long-lived perennial). For example, van der Valk (1981) was able to predict when species with certain vital attributes would become established and when they would be extirpated during wet-dry cycles in prairie potholes in North America (Fig. 1). Only three vital attributes were needed (Fig. 2): life expectancy (A – annuals, P – short-lived perennials, and V perennials with vegetative growth); seed dispersal capacity (S –seeds in seed bank, D –seeds not in seed bank); and germination requirements (type I – seeds germinate only when there is no standing water, II – seeds germinate under water).

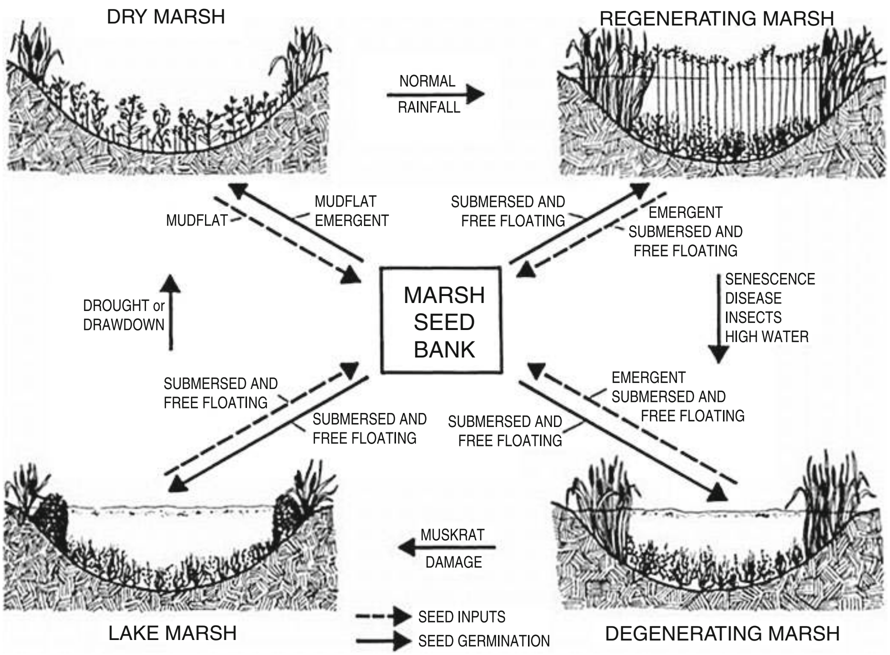

Fig. 1 Changes in vegetation caused by cyclical change in water levels in a prairie pothole over a period of 10–20 years typically (van der Valk 2012)

Many models using some variant of the vital attribute approach have been developed to predict changes in the composition of wetland vegetation after a major disturbance (fire, flooding, drawdown). See van der Valk (2007) for a description of several of these models.

As noted, a vital attribute model for wetlands was first developed to describe and predict changes in the composition of emergent vegetation in prairie potholes caused by alternating years of above normal and below normal annual precipitation (Fig. 1). During such a "wet-dry" cycle, the emergent vegetation of potholes is nearly eliminated by two or more years of high water. The emergent vegetation, however, cannot become re-established from seed until the next dry period when the pothole is largely or entirely free of standing water. For vital attributes to be useful for understanding and predicting vegetation changes, they have to be linked to changes in environmental conditions.

Environmental Sieve Models

In vital attribute models, the environment acts as a "sieve" that determines which species can become established and which will persist under specific environmental conditions (Fig. 2). Whenever the characteristics (state) of the environmental sieve changes, the species composition of the impacted community is altered because

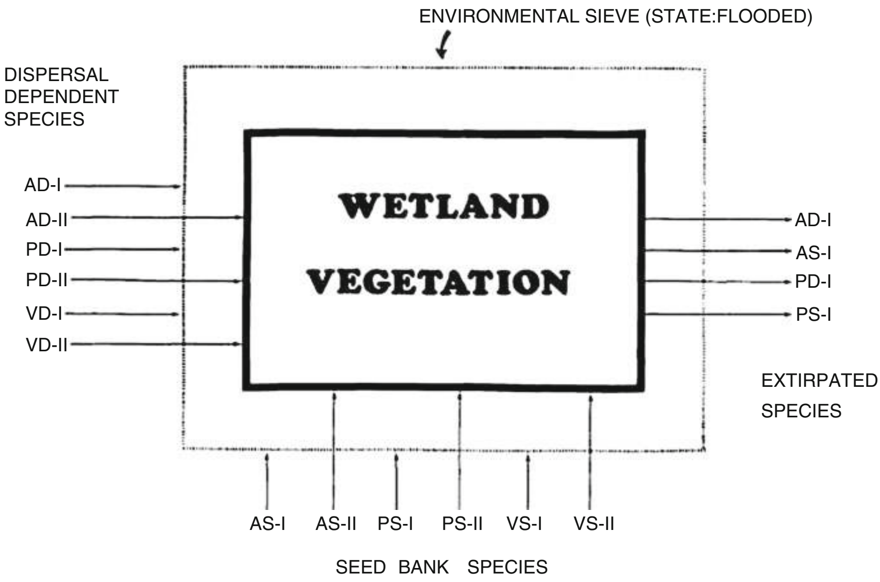

Fig. 2 Environmental sieve model of succession. The species composition of a wetland changes if the state of the environmental sieve changes from flooded (illustrated) to drawdown. The various types of species (AD-I, AS-I, AD-II, etc.) defined by their vital attributes are described in the text (From van der Valk 2012)

established species are eliminated and new species become established. For example, in Fig. 2, when the sieve is in the flooded state, species with some combinations of vital attributes cannot become established (AD-I, AS-I, PD-I, PS-I, VS-I, and VS-I) because their seed cannot germinate under water, but others can (AD-II, AS-II, PD-II, PS-II, VS-II, and VD-II). When the environmental sieve is in this state, most annuals (AS-I species and AD-I) and short-lived perennials (PS-I and PD-I species) will be eliminated from the pothole after a couple of years of flooding because they reach the end of their life expectancies and they cannot become re-established from seed. High water may also result in an expansion of populations of muskrats (*Ondatra zibethicus*). They can destroy most of the remaining emergent vegetation because of their lodge building and feeding activities. When the wetland has no standing water during drought periods, species that were eliminated during flooded periods become re-established from seed.

The most rigorous test of this modeling approach to predicting changes in wetland vegetation was a 10-year long experimental study done in the Delta Marsh, Manitoba, Canada. In this study, ten large experimental cells were constructed in the Marsh and water levels were manipulated using pumps to reproduce previously observed changes water levels over a wet-dry cycle in this wetland. Water levels in each cell were (1) raised for two years to one meter over mean water levels to simulate historic high water years; (2) then lowered to 0.5 m below normal for one or two years to simulate drawdown conditions; and (3) finally raised to

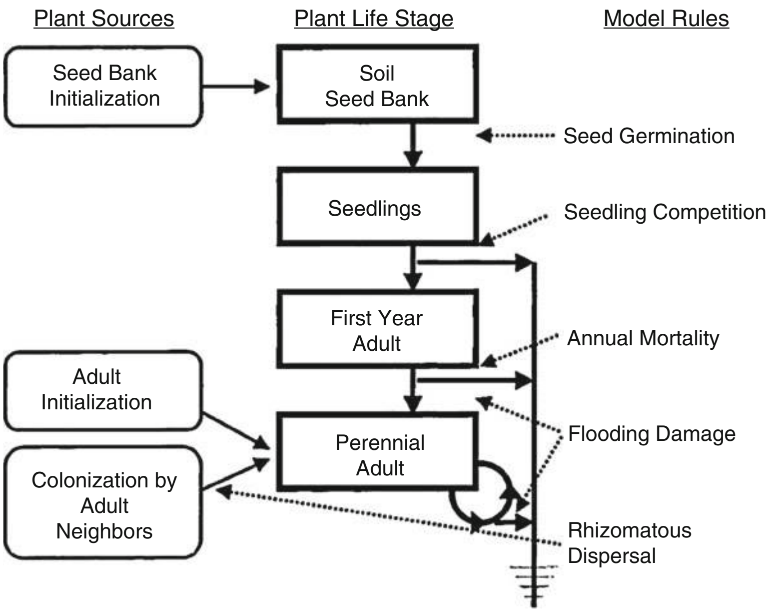

Fig. 3 A vital attribute/environmental sieve model of the dominant emergent species of the Delta Marsh, Manitoba, Canada (From van der Valk 2012)

various depths for 5 years. Seabloom et al. (2001) developed a model based on vital attributes of the dominant species and how changes in water levels affected the survival, recruitment, and location in each cell of each of the dominant emergent species (Fig. 3). This model was developed using vital attribute information gleaned from studies done previously on other prairie wetlands. A detailed description of this study, called the Marsh Ecology Research Program (MERP), can be found in Murkin et al. (2000).

A comparison of the actual vegetation found in the MERP cells and that predicted by the model was used to test the accuracy of the model (Seabloom et al. 2001). Figure 4 illustrates the mean predicted and actual percent of the cells covered by one of the dominant emergent species, *Phragmites australis*, over the course of the study. The model predicted the observed decline of *Phragmites* during the 2 years (1981, 1982) that the cells were flooded to one meter above normal. It also predicted that *Phragmites* would be much more abundant (>20%) in the cells during the drawdown years (1983, 1984 in Fig. 4) than was the case (ca. 10%). *Phragmites* seed germination was lower than predicted during the drawdown. During the reflooding years, the model predicted that the percent of a cell covered with *Phragmites* would initially be high (ca. 20%) during the first couple of years (1985, 1986) and then would decline for the last 3 years. In reality, the percent of a cell covered by *Phragmites* during the reflooding years was initially lower than predicted. After 1986, however, the model and predicted percent coverage were fairly close and had a similar pattern (Fig. 4).

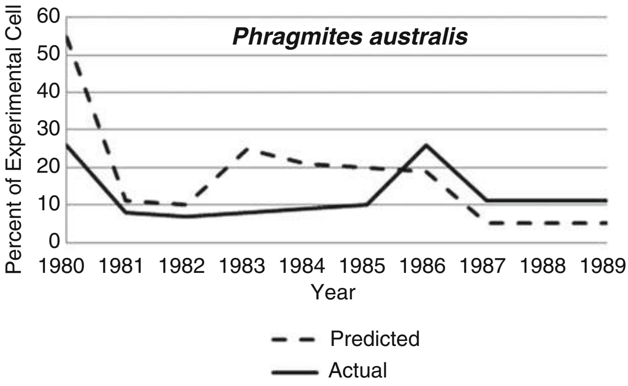

Fig. 4 Mean predicted and actual percent of a cell covered by *Phragmites australis* in the MERP complex. The cells were flooded to 1 m above normal in 1981 and 1982; drawndown in 1983 and 1984; and reflooded from 1985 to 1989. Actual percentages of *Phragmites* in the cells were estimated from low-level aerial photographs (Adapted from Seabloom et al. 2001)

The reason for the discrepancies between the predicted and actual percentages of a cell covered with *Phragmites* varied depending on the state of the environmental sieve. During the drawdown, seed germination was not as high as predicted. The reason for this discrepancy is unknown, but it is clear that more detailed information about the seed germination requirements of *Phragmites* needs to be incorporated into the model. The seed germination of the dominant emergent species in the Delta Marsh is known to be affected by temperature, soil moisture, and salinity (van der Valk 2007). Interactions among emergent species were evident in the field, especially preemption. *Phragmites* was not able to expand into some areas with optimal water depths until other species that became established in these areas began to die back. Preemption effects on the establishment and spread of emergent species were not part of the model and need to be added to increase its accuracy.

Future Challenges

Over the years, many difficulties with the application of the vital attribute/environmental sieve models for predicting changes in the composition of vegetation have been noted. These include: (1) uncertainties about which species are in the regional species pool due to the introduction or invasion of new species and recent local extinction of species; (2) uncertainties about the dispersal potential for most species to a given site; (3) uncertainties about the environmental conditions under which species can become established from seed at a given site; (4) uncertainties about pre-emption or priority effects, i.e., the order in which species arrive; and (5) uncertainties about the consequences of species interactions (competition, allelopathy, herbivory, etc.). Many studies suggest that variation in seed germination rates, priority effects, the vagaries of dispersal, and establishment of new invasive species can have major impacts on community composition (Toth and van der Valk 2012; van der Valk 2012). These problems do not invalidate this approach, but they do indicate that, to increase the accuracy of this class of models, additional life-history attributes may have to be added as abiotic and biotic conditions change over time at a site and as new species become established in a region.

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Egler's $10,000 Succession Challenge

4

John Anderson

Contents

Abstract

Frank Egler was a proponent of a variant of Gleasonian succession called "initial floristics," which suggests that the early establishment of plant species has a great influence on later vegetation succession. In contrast, Clementsian succession suggests that vegetation progresses in stages to a climax. Historically, this idea has been widely held by teachers and managers, but Egler did not observe this type of succession. Egler offered a $10,000 reward for anyone who could give an example of vegetation stages progressing to a climax stage in support of Clementsian succession. No one has successfully challenged Egler's concept. Initial floristics as a variant of Gleasonian succession has withstood many decades of scrutiny.

Keywords

Climax · Clementsian succession · Gleasonian succession · Individualistic concept · Initial floristics · Relay floristics

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Introduction

Frank Edwin Egler, Ph.D., was an eminent plant ecologist and conservationist in his time. Born April 26, 1911, in Manhattan, NY, he was drawn to nature and analytical thinking. As a college student, he studied under such noted ecologists as Henry Cowles (Chicago), William Skinner Cooper (Minnesota), and George Nichols (Yale). He earned his Ph.D. from Yale University in 1936 and went on to publish over 400 papers and five books. He eventually made his home on his family's summer home in Norfolk, Connecticut, which came to be known as Aton Forest. By the continual acquisition of adjacent properties, Egler eventually amassed over 1,100 acres of forest lands, old fields, and wetlands, mostly kept as a natural area (as free as possible from human interference). These lands were his base of operations and his inspiration, on which he conducted much of his research, especially applied ecological vegetation management. Egler died on December 26, 1996. He left his estate to a nonprofit conservation organization that he founded called Aton Forest, Inc. The property continues to be owned and managed by Aton Forest, Inc.; its mission is to carry on Egler's legacy of scientific research and conservation activities.

Ecological Theories

Early on Egler found himself in a controversy over the ecological theories of plant ecology. Fredric Clement proposed the dynamic ecology concept (see Allred and Clements 1949), which came to be known as plant succession to climax, where discrete plant communities not only succeed one another in a deterministic way but each proceeding stage preparing the way for the next stage. The final stage would vary with climate, but would be stable, i.e., the climax. Henry Gleason proposed the individualistic concept (Gleason 1926), where individual species associate with one another to form communities, a low degree of holistic integration, but in a nondeterministic way. Egler proposed a variant on Gleason's individualistic concept, which was called initial floristic composition (Egler 1954), where early establishment of plant species had a great influence (barring external effects of, e.g., fire, flood on the future development of succeeding plant communities). During his life, Egler repeatedly pointed out that the title of his paper was Initial Floristic Composition: *a factor* in old-field vegetation development. In this paper, he renames Clement's plant succession concept as Relay Floristics and acknowledges that it does play a role in vegetation development, but is not the sole or even the most important factor.

As Egler developed his ecological and vegetation insights into applied science and specifically right-of-way vegetation management, the importance of this distinction of concepts came to light. The assumption, for example, that a shrub community is the natural and necessary stage to proceed a pioneer forest type would naturally influence management decisions. From this perspective, shrubs pave the way for trees, so it would be logical to get rid of the shrubs if you do not want trees. If initial floristic composition is more important, then eliminating trees early on should prevent a future forest if the shrub community resists rather than encourages new tree invasion. Egler spent much of his professional life investigating and demonstrating these ideas. He was vexed that so often he came up against the belief in a succession to climax theory from land and vegetation managers and students and teachers, despite the fact that Clement's concept had lost favor or been discounted by ecologists.

Egler's Challenge

With the publication of his two-volume book *The Plight of the Right of way Domain: Victim of Vandalism* (Egler 1975), Egler printed in its appendix his $10,000 Challenge (p. 283). Here he offers a reward for any ecologist who can scientifically prove the cause and effect relationship of Clementsian Relay Floristics through at least five stages of development over a minimum of 25 years with evidence to be judged by a panel of noted ecologists. No one ever took him up on this offer. Egler did speak about being referred to someone in Ohio who was managing vegetation where tree invasion into dense shrub stands was common. He contacted the person and went to Ohio to see the site and discuss its history and the observations that had been made. He was told, yes, trees were very commonly growing up through the shrub canopy, and after some further questioning, he was told, "But I don't know which are older, the shrubs or the trees." This, of course, is the essence of the whole argument!

So the Challenge stands unchallenged. Many foresters and right-of-way vegetation managers now acknowledge the significance of Egler's ideas to their planning and work, though full understanding and implementation of his concepts and management practices is yet to be realized.

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Succession in Ecological Education

5

David J. Gibson and Beth A. Middleton

Contents

Abstract

As complex and dynamic systems, wetlands offer the opportunity to investigate and incorporate the ecological concept of succession in educational settings. For example, the well-known, classic hydrosere concept is illustrated in numerous ecology and life-science textbooks. In this chapter, the drawbacks of using the hydrosere successional concept are assessed, and two examples of using wetlands to illustrate the process of succession for educational purposes are described. In each case, the premise and approach is that students best "learn ecology by doing ecology."

Keywords

Education · Succession · Wetlands

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Introduction

As complex and dynamic systems, wetlands offer the opportunity to investigate and incorporate the ecological concept of succession in educational settings. For example, the well-known, classic hydrosere concept is illustrated in numerous ecology and life-science textbooks. In this article, the drawbacks of using the hydrosere successional concept are assessed, and two examples of using wetlands to illustrate the process of succession for educational purposes are described. In each case, the premise and approach is that students best "learn ecology by doing ecology" (Gibson et al. 1999).

Be Careful Using the Classic Hydrosere Succession Concept as a Teaching Tool

The classic hydrosere model of succession, where infilling of a lake or pond through time leads to a "climax" forest community, has a long history and is reproduced in many textbooks as a good example to teach the concepts of primary succession. However, numerous exceptions to this classic sequence of hydrosere succession have been observed, and forest communities often do not represent a final, stable community. Infilling of basins with organic matter does occur (i.e., terrestrialization), but succession is rarely unidirectional, and does not result in mature upland or mesic "climax" forests replacing earlier successional bog stages (Klinger 1996). Instead, for understanding wetland succession, a Gleasonian approach based upon species life histories is more appropriate (Van der Valk 1981; Middleton 1999). The fallacy of using the Clementsian model in teaching successional concepts was outlined by Gibson (1996) who advocated a more probabilistic, hierarchical approach.

The Olentangy River Wetland Research Park and EARTH University's Tropical Wetland Project

Two riverine wetlands constructed by William Mitsch and colleagues at Ohio State University provide examples of ecosystem-scale systems that can be used for conservation education and research in a successional wetland setting (Mitsch et al. 2008; Mitsch et al. 2012). The Olentangy River Wetland Research Park (ORWRP) is a pair of 1 ha flow-through-created riverine wetland basins established in 1994. One basin was planted with 13 native species of macrophytes; the other was unplanted and allowed to colonize naturally (Mitsch et al. 2012). After 15 years, the planted basin had higher plant community diversity and lower primary production than the unplanted basin. In addition to monitoring vegetation, soil development, water quality changes, and carbon and nitrogen dynamics, the faculty at The Ohio State University developed at least 35 courses taking advantage of the successional dynamics at the ORWRP (http://swamp.osu.edu/Academics/academic_courses.html). Similarly, a swamp palm (*Raphia taedigera*

Mart.) forest in La Reserva Wetland at EARTH University, Costa Rica, is one of several natural and constructed wetlands (112 ha in size), which has been developed as a campus research wetland since 2005 (Mitsch et al. 2008). In addition to improving effluent water quality of an animal farm, dairy plant, landfill, and banana plantation, studies on these wetlands have been integrated into the educational curriculum at EARTH University for over 14 years. As the constructed wetlands change in plant composition and structure over time, they offer opportunities to teach successional theory.

Integrating Wetland Succession into Undergraduate Life Science Courses

To incorporate the "learning ecology by doing ecology" concept, a set of experimental plots were established in a postagricultural bottomland forest in southern Illinois (Gibson et al. 1999, 2005) (Fig. 1). Mowing and rototilling, fertilizer addition, and herbivore exclusion treatments allowed a test of the effects of disturbance as defined by mowing and rototilling, and deer browsing and resource availability on secondary succession in this wetland habitat (Mathis 2001). One hundred and forty-four 5 × 5 m plots in 16 15 × 15 m blocks were established

Fig. 1 Aerial view of experimental research plots in a postagricultural bottomland habitat in southern Illinois. The plots contrasted mowing and fertilizer treatments. Twelve of the 16 blocks excluded large herbivores. Undergraduate ecology laboratory classes tested the initial- versus relay-floristics models. Each of the square blocks is 15 × 15 m and contained nine 5 × 5 m plots assigned to mowing (control, mowed annually, mowed and rototilled annually) and fertilizer (control, fertilizer every 5 years, and annual fertilizer application) treatments (Photo by David Gibson)

Fig. 2 Undergraduate students counting tree seedlings in the bottomland hardwood forest secondary succession plots (Photo by D. J. Gibson)

following plowing of the abandoned agricultural field in 1996, with experimental treatments and data collection maintained through 2002 (http://www.plantbiology.siu.edu/long-term/). A second set of plots with similar mowing and fertilizer treatments, but without herbivore exclusion, were established during the same time period in a nearby upland area (Gibson et al. 2005).

These plots were used as a context for inquiry-based undergraduate ecology laboratories to test the basic tenets of succession, including an evaluation of relay versus initial floristics, seedling growth experiments (Barko et al. 2004), and predator-prey relationships (Oyler et al. 1999), and for individual undergraduate research projects (Rice et al. 1999). Such plots could also be used for teacher training of inquiry methods for K-12 education. Evaluations showed that these laboratories were educationally effective in meeting our objectives (Bhattacharyya 1999).

Undergraduates were able to test relay versus initial floristics (i.e., contrasting sequential versus early species establishment) in these plots by noting the number of trees established in disturbed (mowed) versus undisturbed (unmowed control) plots (Fig. 2). Before collecting these data, students were asked to conduct a preproject assessment consisting of a multiple choice sentence stem and a concept map (Box 1) to evaluate their knowledge of succession prior to the exercise. After collecting data on tree density, the students were asked to repeat the assessment in light of the data that they had collected and their field experience (Fig. 3). After this field exercise, the students were introduced to the concepts of relay versus initial floristics and generally found that their observations supported the latter and that their knowledge and understanding of succession had improved (Bhattacharyya 1999).

Box 1 Pre- and Postproject Assessment

Student demonstrates their understanding of the concepts related to succession after the field exercise by completing the sentence stem (1) or drawing a concept map (2) as directed below.

1. Sentence Stem:

   Circle one of the following.
   After the farmer stops cultivating a field, trees come into abandoned fields: 25 years, 10 years, 3 years, 1 year, right away.

2. Concept Map:

   Draw a concept map of succession including the following terms (plus additional terms you think are necessary):
   Plant types: tree seeds, tree seedlings, grass seeds, grass seedlings, mature forest
   Changes: cutting, mowing, dispersal

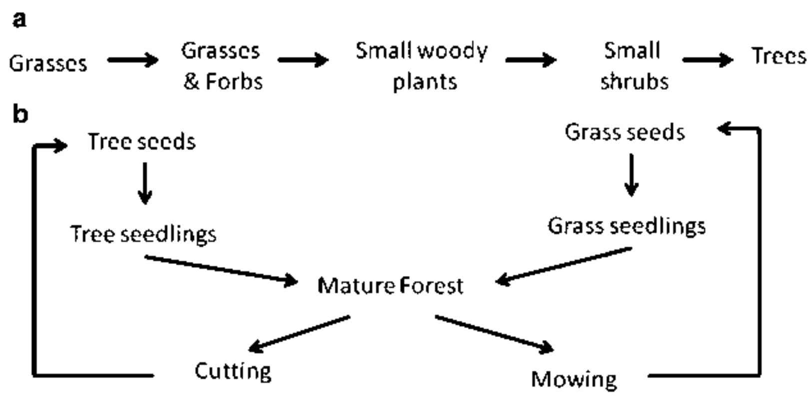

Fig. 3 Sample of student's concept maps illustrating their understanding of the process of secondary succession in bottomland hardwood forest. a) Low-scoring concept map, b) high-scoring concept map (Redrawn from Bhattacharyya 1999)

Future Challenges

Successional wetlands provide a setting to place educational opportunities into current ecological theory. However, it is necessary to avoid Clementsian successional ideas in which a predictable sequence of species changes leads to a "climax" community. There are limitations and caveats for using space-for-time substitutions (i.e., chronosequences where sites of different times since disturbance are compared instead of following a site continuously through time since disturbance) (Neiring 1994; Johnson and Miyanishi 2008; Walker et al. 2010) and spatial belting within wetlands as representations of temporal succession (Neiring 1987). Successional wetlands allow field-based laboratory classes to be developed based on using inquiry-based learning. This pedagogical approach can also be used to alleviate negative adult perceptions of these habitats that arise from the view that they are undesirable, dangerous places (Anderson and Moss 1993).

In addition to the examples cited here, some online education sources focused on using wetland systems are provided in Box 2.

Box 2 Online Educational Resources for Wetland Education

1. Ducks Unlimited's Teacher's Guide to Wetland Activities (http://www.greenwing.org/dueducator/teachersguide_educator.html) and Conservation Lesson plans (http://www.greenwing.org/dueducator/lesson_plans.html).
2. WOW! The Wonders of Wetlands (http://www.wetland.org/education_wow.htm).
3. EPA Wetlands Education (http://water.epa.gov/type/wetlands/outreach/education_index.cfm).
4. Environmental Concern Inc. is a 501(c)3 public not-for-profit Corporation that is dedicated to working with all aspects of wetlands (http://www.wetland.org/index.htm).

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Self-Design vs. Designer Theories and Wetland Restoration and Creation

6

Arnold van der Valk

Contents

Abstract

Wetland restoration and succession are essentially attempts to accelerate and direct succession. However, competing theories about the nature of succession exist. One theory developed by F. E. Clements postulated that all the vegetation in an area would eventually reach a final, stable stage that he called the climax. This deterministic succession theory, now called the self-design theory, has been adopted in numerous restoration/creation projects. It implies that environmental conditions are the main determinant of the vegetation that develops. An alternative theory of succession, associated with H. A. Gleason, emphasizes that the characteristics of each plant species (e.g., seed dispersal potential) and contingent environmental factors (e.g., disturbances) have a major influence on the composition of the vegetation that develops and that there is no fixed end point. This individualistic theory of succession in the restoration field is known as the designer theory. Most plant ecologists today are advocates of the designer theory.

Keywords

Climax · F. E. Clements · H. A. Gleason · Succession · Wetland vegetation

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Introduction

When it comes to the restoration/creation of wetlands, it is widely assumed that it is possible to establish vegetation in them that closely resembles that of some target wetland, all else being equal. This is one of the underlying assumptions of wetland mitigation policy in the United States and elsewhere. Because wetland restoration/creation is essentially accelerated succession, the extensive literature on succession provides both theoretical concepts for designing and practical guidelines for executing restoration/creation projects. Consequently, the need to apply succession theory to aid in the design of restoration/creation projects has been frequently advocated. However, for this to be effective and successful, it is necessary to use up-to-date successional theory.

Pickett et al. (2011) compare and contrast classical and contemporary theories of succession. The classical theory, developed primarily by Frederic E. Clements in the early part of the twentieth century, was a deterministic theory that postulated that all the vegetation in a given area would eventually reach a climax. This climax vegetation would be a stable assemblage of species that was in equilibrium with the regional climate. Interactions of species on a site during succession (primarily competition) plus changes in environmental conditions caused by the plants themselves, called reactions (e.g., increased soil organic matter and available nutrients), caused changes in species composition over time until environmental conditions stabilized and the plant species were in competitive equilibrium. In the classical theory, it is the climate that is the major determinate of the type of climax vegetation that develops because only the species best adapted to this climate will ultimately survive. Nevertheless, the classical theory recognized that succession on any given site within a region was affected by site conditions (soil moisture, soil nutrients, local disturbances, herbivores, etc.), limitations on species dispersal to the site, and the performance (seed germination, seedling survival) of species that reached the site. One or more of these factors sometimes prevented the development of the expected climax vegetation.

The contemporary theory of succession, which was formulated in the last half of the twentieth century, differs from the classical theory primarily in its emphasis on the importance of contingent factors (disturbances, site conditions, vagaries of seed dispersal, etc.) for determining whether a given species is found or not on a site at a given time. In the last 60 years, numerous studies of succession have demonstrated that in a given area it is largely driven by contingent factors. Consequently, succession does not have a fixed trajectory or outcome, and many different vegetation types could potentially become established on the same site. Rather than being deterministic, succession is viewed as a probabilistic process.

Self-Design and Designer Theory

Although the terminology is different, in the wetland restoration literature two different successional theories have been proposed, self-design and designer, to guide efforts in vegetating newly restored or created wetlands (Mitsch and

Wilson 1996). One of the earliest proponents of the self-design concept was Howard T. Odum. Odum, however, called it self-organization (Odum 1988). Odum believed that if an area was "seeded" with suitable organisms that, as a result of their differential survival and interactions (feedbacks) among them, self-organization would result in development of the most energy efficient ecosystem under a given set of environmental conditions. Like the classical theory of succession, of which it is a variant, self-organization is a deterministic process with a fixed endpoint. For Odum, the climax ecosystem operates at optimal efficiency, and less energy efficient ecosystems will be replaced by more energy efficient ones. See Månsson and McGlade (1993) for a critique of Odum's concepts. One of Howard Odum's students, William J. Mitsch, applied Odum's self-organization/self-design concept to the restoration/creation of wetlands (Mitsch and Wilson 1996). As was the case with Odum, self-design involves introducing propagules of as many species as possible and letting "natural forces" choose the most appropriate species. Again, it is classical succession theory under another name.

The designer approach is described by Mitsch and Wilson as "introducing species and expecting their survival in Gleasonian zones, akin to gardening and landscape architecture." The designer theory is a variant of the contemporary theory of succession. There is no fixed endpoint, and, within the constraints imposed by environmental conditions, many different kinds of vegetation could be established in a restored or created wetland because it is possible to manipulate the species composition of the initial vegetation (e.g., using different seed mixes) and to manipulate environmental conditions to favor some species over others. In other words, it is possible to direct succession to obtain a desired vegetation type.

As pointed out by van der Valk (1998), the self-design concept has two important implications: (1) it suggests that it easy to establish vegetation – it only requires introducing suitable propagules; and (2) because of self-design, the type of vegetation that develops is inevitable and cannot be altered. Thus, according to the self-design theory, if the vegetation that develops is composed mostly of weedy species and does not resemble that in target wetlands, there is nothing that can be done to prevent this. On the other hand, the designer theory implies that the role of restoration ecologist is much more than just a sower of seeds. In order to establish to establish a particular kind of vegetation, it is necessary to have detailed knowledge of the life-history attributes of the species: how species are dispersed, under what conditions they can become established, what is their life expectancy, what are the main sources of mortality, etc. In fact, restoration ecologists have demonstrated repeatedly that it is possible to establish desired vegetation in restored and created wetlands if you understand the establishment and growth requirements of the constituent species. In short, self-design and designer restoration theories are not new. They are restatements, of the discredited classical theory and the contemporary theory of succession, respectively. Only the application of the contemporary succession theory to wetland restoration and creation projects will improve their outcomes.

Future Challenges

Improving the quality of wetland restoration and creation projects is essential if funding for them is to continue. For projects where self-design has been assumed to result in the establishment of suitable vegetation, the failure of the expected vegetation to develop has increasingly become an embarrassment to agencies and organizations funding these projects. Because the self-design assumption greatly reduces the cost of projects, convincing funding agencies to allocate the funds needed to establish suitable vegetation, not just suitable hydrology, is needed. Not only funding will be needed to establish suitable vegetation, it will also require developing sources of wetland seeds and plants of local provenance and effective techniques for successfully establishing suitable species where this has previously not been attempted. Fortunately, a great deal of work has been done on propagating wetland species and developing planting techniques in parts of the world where self-design has not been assumed (van der Valk 2009). The control of invasive species may also be needed to prevent them from dominating recently restored or created wetlands.

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Cattle Grazing in Wetlands

7

Beth A. Middleton

Contents

Abstract

Cattle grazing drives successional change in wetland vegetation by removing tall grasses and other vegetation. As a disturbance, cattle grazing in some ways resembles natural disturbances such as native mammal grazing and lightning-strike fire, which can support higher biodiversity in wetlands. To encourage rare and Red-Listed species, natural land managers sometimes incorporate a variety of techniques to remove tall vegetation including mowing, hand-cutting, burning and cattle grazing. As a farming practice, cattle grazing was once very common in world wetlands, but as agriculture intensified after WWII, small-scale farmers slowly stopped grazing cattle in natural wetlands. As a result, tall macrophyte and woody species have overgrown some wetland types once used as pastures for cattle.

Keywords

Disturbance dynamics · Gleasonian succession · Shrub invasion · Succession

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Introduction

Cattle grazing is a disturbance, which can alter the successional dynamics of wetlands (Fig. 1; Middleton 1999). This change in species composition is caused by the cattle because they create openings in the dominant vegetation, soil disturbances (hoof prints, lounging, manure, cattle paths), swim tracks, alien species invasion, interference with native animal species, and sometimes salt water intrusion (Finlayson and Von Oertson 1993; Fig. 2). Most wetlands can recover from grazing, even though the species composition of the wetland may be changed permanently.

How Cattle Change Wetlands

Heavy cattle grazing damages natural wetland vegetation more than lighter grazing (Middleton et al. 2006). During times of grazing, the cattle promote the spread of shrubs by creating invasion opportunities in otherwise dense vegetation and dispersing shrub seeds via manure. If the cattle are removed, the shrubs may mature and come to dominate the wetland (Fig. 3).

The restoration of wetlands damaged by cattle grazing can be difficult. If shrubs invade after cattle grazing, these can be difficult to remove because of their deep roots (Fig. 4). Stream banks may need costly rehabilitation after cattle grazing. Cattle damage stream banks in their pursuit of water in stream beds.

Historically, hay in wetlands was cut to feed to cattle in North America, Europe, Asia, Africa, and India. These wetlands included northern peatlands, salt marshes,

Fig. 1 Feral cattle (*Bos indicus*) graze in monsoonal wetlands of the Keoladeo National Park, Rajasthan, India (Photo by Beth Middleton)

Fig. 2 Cattle on their evening walk home from the wetland in the western Jilin Province, China. Note the cattle trail in the foreground; hoof prints are a major disturbance in wetlands grazed by cattle (Photo by Beth Middleton)

floating freshwater marshes, and monsoonal wetlands (Middleton 1999). In North America and Europe, these labor-intensive farming practices were discontinued gradually after World War II (Middleton et al. 2006). Recently, to preserve the high biodiversity of wetland vegetation in Europe and elsewhere, wetland vegetation is managed to curtail the growth of tall plants.

Cessation of Cattle Grazing in World Wetlands

Agriculture around the world has become more industrialized around the world since WWII, so that marginal land has been abandoned (Benayas et al. 2007). At the same time, cattle grazing in natural areas has stopped in many regions of the world (Middleton et al. 2006), so that a loss of biodiversity has occurred in adjacent natural ecosystems (Benayas et al. 2007). The loss of biodiversity occurs as taller vegetation such as shrubs or macrophytes overtops shorter vegetation (Middleton et al. 2006).

The reasons for the exodus of people from rural areas is mainly socioeconomic factors, including dwindling resources for small farmers with farm industrialization, but also better opportunities in the city than in the countryside (Benayas et al. 2007). Also, many rural people left the country after the collapse of the former Soviet Union (Baumann et al. 2011). Other historical occurrences have caused declines in the farm population, resulting in landscape-level changes in the countryside. After the

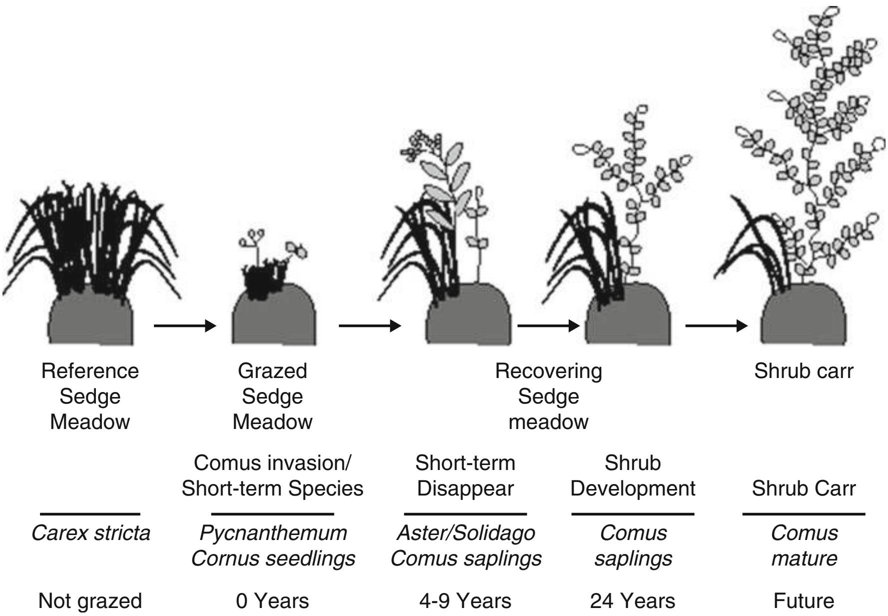

Fig. 3 Cattle grazing damages sedge tussocks in northern sedge meadows and the damage allows an invasion pathway for shrubby species such as red osier dogwood (*Cornus sericea*). Shrubs are browsed by cattle, so that the sedge meadow may not appear changed while cattle still graze the wetland. In the years immediately following cattle grazing, the shrubs mature and eventually dominate formerly grazed sedge meadows. In certain northern peatlands, the formation of a shrub carr is not an inevitable product of succession, but instead often is facilitated by cattle grazing

Eurasian plague of AD 1347, shrubs proliferated as pastures were abandoned in Europe with fewer people to tend the cattle (Yeloff and Van Geel 2007). Large-scale socioeconomic changes in the rural countryside have other impacts on surrounding wetlands and their biodiversity (following Benayas et al. 2007).

Future Challenges

Small-scale farmers have stopped grazing cattle in many natural areas around the world, and this land-use change has caused a crisis in land management in wetlands. In many types of wetlands, the spread of unwanted macrophytes and woody species has occurred on formerly pastured wetlands, resulting in the loss of rare and Red-Listed species (Middleton et al. 2006). One challenge for land managers is to find ways to maintain the biodiversity of these wetlands following these rural land-use shifts. Managers have incorporated various strategies to control tall vegetation including mowing, hand-cutting, burning, large mammal grazing, and even the

Fig. 4 The removal of woody species from former grazing areas is difficult and labor intensive. Cattle eat and spread the seeds of mesquite (*Prosopis juliflora*) in the Keoladeo National Park (India) (Photo by Beth Middleton)

reintroduction of cattle (Middleton et al. 2006). In mountain grasslands, management to restore species from seed banks and other sources by mowing still has some benefits for reestablishing biodiversity even after 20 years of cattle grazing abandonment (Galvánek and Lepŝ 2008).

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Fire in Borneo Peatlands

8

Sue Page and Agata Hoscilo

Contents

Abstract

Fire frequency has been increasing in recent decades in Southeast Asia, particularly with the expansion of agriculture and wood extraction on the islands of Borneo and Sumatra. These peatland fires are causing high emissions of carbon and particulates, as well as long-term changes to forest composition, biomass, and structure. Only a few tree species in this ecosystem have fire adaptations, e.g., tumeh (*Combretocarpus rotundatus*), so that such species are common on fire-degraded peatlands. The protection of remaining forests on the islands of Borneo and Sumatra could be possible as a result of policy initiatives to reduce greenhouse gas emissions. In addition, rehabilitation of these forests may present opportunities for mitigation. At the same time, there are a number of technological and sociological challenges for the restoration of these forested peatlands. These landscapes are now highly fire-prone, and the livelihoods of local people need attention for these measures to succeed.

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Keywords
Agricultural expansion · *Combretocarpus rotundatus* · Disturbance · Forest degradation · Peatland fire · Southeast Asia

Introduction

Fire is presently the most important disturbance affecting vegetation cover at a global scale, with an estimated 6 million km^2 affected on an annual basis (Mouillot and Field 2005). The largest increase in regional fire activity is currently occurring in the humid tropics, particularly in the rainforests of Southeast Asia and South America. Before recent times, fire activity was both infrequent and atypical of these tropical forest environments, but now wildfires are increasing rapidly, resulting in extensive forest loss and degradation and a wide range of environmental, social, and economic impacts (Cochrane 2006).

In recent decades, the Southeast Asia region has experienced extreme rates of forest loss, degradation, and wildfire (Achard et al. 2002; Langner et al. 2007; Hansen et al. 2009). During the period 1950–2000, 40% of Indonesian forests were cleared, with agricultural expansion and wood extraction identified as the main drivers of forest loss (Geist and Lambin 2002). Both activities have also increased the risk of forest fire. Langner and Siegert (2009) demonstrated that around 21% of land in Borneo was subject to fires during 1997–2006, with 6.1% (45,000 km^2) of the forest affected by multiple fires, most if not all of which were anthropogenic in origin. Some of the most extensive fires since the mid-1990s have occurred in this region's peat swamp forests, particularly on the islands of Borneo and Sumatra (Langner et al. 2007; Langner and Siegert 2009). In 2002, for example, 73% of the fire-affected forest area of Borneo was peat swamp forest and in 2005, it was 55% (Langner et al. 2007) (Fig. 1).

Tropical peatland, which supports a natural vegetation cover of peat swamp forest, covers 250,000 km^2 of the coastal lowlands of Southeast Asia (Page et al. 2011) and is characterized by the presence of very thick peat deposits often exceeding depths of 10 m or more. In undisturbed peat swamp forest, peat accumulation is maintained by a strong interrelation between the forest vegetation and the peat under anoxic, waterlogged, nutrient-poor conditions, which reduce the rate of decomposition enabling organic matter derived from forest biomass to accumulate on the land surface. Peat swamp forest contains a diverse range of tree species, many of them commercially valuable for timber or other uses, while the forest and underlying peat are important for carbon storage, with approximately 100–150 t C ha^{-1} stored in the forest biomass and 2775 t C ha^{-1} in the peat (assuming an average peat thickness of 5.5 m) (Page et al. 2011).

Over recent decades, very large areas of peat swamp forest in Southeast Asia, especially in Indonesia and Malaysia, have been deforested and drained to make way for large-scale industrial plantations of oil palm and pulpwood (*Acacia*), and for smallholder agriculture (Miettinen et al. 2011). Many plantation companies, particularly in Indonesia, have employed fire as a cheap and rapid land clearance tool to dispose of

Fig. 1 Peat swamp forest recovering from a single fire in 1997 in southern Central Kalimantan, Indonesian Borneo; after 9 years of regrowth, the woody biomass had reached around 10–15% of the biomass in primary peat swamp forest (Photo taken by A. Hoscilo in 2006)

logging residues, while smallholders have used fire to keep their land clear of invasive vegetation. The risk of wildfire on peatland has been increased considerably by recent forest disturbance and drainage activities (Langner and Siegert 2009). In the past, small scale wildfires escaping from agricultural land would have quickly burnt-out in the humid and waterlogged peat swamp environment, but the combination of forest disturbance and drainage have allowed peatland fires to run out of control, destroying both the forest vegetation and burning away the dry surface of the peat (Page et al. 2002). This has resulted in large carbon emissions to the atmosphere as well as a wide range of other detrimental environmental impacts.

Environmental Impacts of Tropical Peatland Fires

Tropical peatland fires make a substantial contribution to the global burden of greenhouse gases (Bowman et al. 2009) and the devastating peatland fires of 1997–1998 in Indonesia were one of the largest peak emissions events in the recorded history of fire in Southeast Asia, if not globally (Schultz et al. 2008; van der Werf et al. 2006). Page et al. (2002) conservatively estimated that these fires released more than 870 Mt of carbon to the atmosphere, which was equivalent to

14% of the average global annual fossil fuel emissions released during the 1990s. In contrast to forests on mineral soils, combustion losses from belowground (i.e., peat) carbon stocks in peat swamp forest fires can be much greater than the losses from surface and canopy fires, which consume only the aboveground biomass. In addition, the smoldering nature of peatland ground fires (i.e., fires which burn below the peat surface in a low oxygen environment) results not only in the release of greenhouse gases (CO_2, CO and CH_4) but also fine particulates and a range of volatile organic compounds. The resulting dense pall of smoke can affect the whole region and result in a wide range of human health problems (Heil and Goldammer 2001).

The most extensive peatland fires of the last two decades have occurred during ENSO events, when the relatively short dry season affecting the western part of the Southeast Asia region is extended, from 2 or 3 months to up to 6 months or longer. While ENSO may provide suitable dry weather conditions, most of the region's uncontrolled fires are fostered and propagated by the increased fire susceptibility of over-drained peatlands. This problem is demonstrated by a study of the fire regime in the former Mega Rice Project (MRP) area on peatland in southern Central Kalimantan (Hoscilo et al. 2011). Over a 23 year period (1973–1996) prior to peatland drainage in the mid-1990s, fires affected 23% of a 4,500 km^2study area, with most fires located along the forest edge, i.e., in disturbed forest in close proximity to human settlements. More remote, intact forests were unaffected by fire, even during extended ENSO droughts. This situation changed markedly in the next decade. Following construction of an extensive canal network in 1995–1996 to drain the peatland for agricultural use, fires during the ENSO year of 1997 affected about 33% of the area, i.e., 10% more than had burned during the previous 23 year period. In total, more than half of the area burnt during 1997–2005, with many locations experiencing multiple fires. This rapid increase in burned area was a consequence of deforestation, peatland drainage, and increased human access; drainage and forest disturbance increased the fire risk, whilst people provided the fire ignition source.

In addition to high emissions of carbon and particulates to the atmosphere, peatland fires also result in long-term changes to the species composition, biomass, and structure of the peat swamp forest and, particularly in the case of repeated fires, greatly reduced rates of post-disturbance vegetation recovery. Peat swamp forest trees are not fire-adapted; most are thin barked, so post-fire tree mortality is high. In addition, many trees fall once the supporting peat around their root systems has burnt away. In a study of the post-fire vegetation response in Central Kalimantan, Page et al. (2009) and Hoscilo et al. (2011, 2013) have shown that peat swamp forest subject to a single, low intensity fire can undergo relatively rapid recovery to secondary forest, achieving a biomass equivalent to about 10% of that of the undisturbed forest over a 9 year period. Following multiple fires, however, the species richness and density of trees, saplings and seedlings within secondary vegetation are greatly reduced, which decreases the capacity of the vegetation to recover naturally. Fire damaged peat has diminished seed availability and dispersal is impaired, leading to a decline in seedlings and saplings. Fire also removes the vegetative regeneration potential of tree bases and roots, which are burned away. Intact peat swamp forest can usually support more than 100 tree species per hectare,

Fig. 2 The most advanced stage of vegetation regrowth following a low severity second fire (burnt in 1997 and 2002). The woody regrowth is dominated by the tree C*ombretocarpus rotundatus* (Photo taken by A. Hoscilo in 2006 – four years after the last fire)

while areas affected by multiple fires are colonized by only a very small number of pioneers. In Central Kalimantan, *Combretocarpus rotundatus* (tumeh) is the dominant species on peatlands subject to several fires. Tumeh has small, winged, wind-dispersed seeds, and has adaptations, which enable it to disperse over large areas and tolerate environmental conditions in a fire-degraded ecosystem. It is a light-tolerating species, which is able to reproduce very easily both by quick production of seeds and re-sprouting from the roots. Because of these traits, tumeh spreads relatively easily into burned areas. It is also a fire-tolerating species, which can survive a second less severe fire owing to a thick bark that protects the underlying vascular tissues (Fig. 2).

At the highest levels of fire-degradation (i.e., following multiple fires), the vegetation succession back to peat swamp forest is diverted to a community dominated by ferns with very few or no pioneer trees. The dominant species of fern is usually *Stenochlaena palustris*, which can grow to heights of 1.5 m or more and its dense vegetation can overgrow and outcompete many tree seedlings and saplings in the early stages of their development. The dead and live fronds of this species are highly flammable and their presence increases the risk of future surface fires, although the low biomass of this non-woody vegetation means the fuel load is low and thus it is less likely that surface fires will generate sufficient heat to establish ground fires within the peat. Rapid colonization by non-woody vegetation may lead to the irreversible transformation of peat swamp forest into herbaceous, savanna-type habitats. A further consequence of repeated peatland fires is land subsidence, a result of peat dewatering, biological oxidation and combustion losses, which leads to an increased risk of surface flooding during the wet season. Areas burnt only once and subject to shallow, short duration flooding can undergo succession to relatively species-diverse, well-structured forest vegetation if there is a sufficiently long period without disturbance. In contrast, sites subject to multiple burns, peat subsidence and deep or prolonged

Fig. 3 Invasion of non-woody fern vegetation in locations subject to multiple fires of high severity (burnt in 1997 and 2002); the fern *Stenochlaena palustris*, growing in the foreground, is dominant (Photo taken by A. Hoscilo in 2006 - four years after the last fire)

flooding have a much more poorly developed and less diverse vegetation dominated by flood-tolerant, non-woody vegetation (Page et al. 2009) (Fig. 3).

Future Challenges

Future land use and climate changes will likely increase the frequency and severity of fires in the Southeast Asian region due largely to predict enhanced climatic seasonality, i.e., wet season precipitation increase and dry season decrease. The future behavior of ENSO is uncertain, but a recent study indicates that Indonesia as a whole could expect more frequent and longer droughts in the future (Abram et al. 2007). Deforestation itself, i.e., large-scale alterations in land cover, may also lead to more localized reductions in rainfall (Werth and Avissar 2005). These changes will be critical for the remaining fragments of peat swamp forest, which are increasingly degraded by over-logging and drainage and at considerable risk of continuing or increasing fire risk in the coming decades, unless large-scale improvements in forest management are implemented.

Opportunities to protect the remaining peat swamp forests from fire damage may come about as a result of new policy initiatives to reduce greenhouse gas emissions through avoided deforestation in developing countries (REDD and the voluntary carbon market) while the rehabilitation of degraded peatland landscapes may offer opportunities for mitigation through rewetting, fire control, and reforestation under agreements to reduce greenhouse gas emissions. Initial restoration trials in the former Mega Rice Project area in Central Kalimantan have been undertaken to investigate the effectiveness of various forms of intervention to protect the remaining carbon store (Page et al. 2009) and a REDD demonstration project, funded by the Australian Government, is underway in this area. There are many technical challenges to restoration of these ecosystems including establishing effective fire control in what are now highly fire-prone landscapes but also socioeconomic ones of persuading local people that these measures will benefit livelihoods.

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Succession in Coastal Wetlands

9

John Teal

Contents

Abstract
Vegetation species and abundance in coastal marshes is controlled by salinity and immersion time. Future changes will be affected by human activities such as climate change, coastal pollution and and introduction of foreign species.

Keywords
Salinity · Phragmites · Spartina · Pollution · Climate change

Introduction

This article deals with changes in vegetation in wetlands located on coasts, subject to saline waters of the oceans.

Two factors control the species, abundance and distribution of plants occupying saltwater wetlands: salinity of the tidal water and duration of inundation – the length of time salinity stress is present. This is the water regime. The dominant species of saltwater wetlands are evolved from upland plants that developed salt tolerance and ability to survive soil anoxia, which reduces their competitive ability in other environments.

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An elegant picture of salt marsh succession was given by Alfred Redfield (1972) in his discussion of the development of Great Barnstable Marsh in Cape Cod, Massachusetts. Sediment accumulated in an inlet protected from storms until it reached about mid-tide level. *Spartina alterniflora*, which grows when flooded no more than half the time by saline water, invaded the newly created environment. This is primary succession. Succession to other species occurred when the surface elevation increased via accumulation of organic material from *Spartina* roots and rhizomes and with sediment brought in by the tides. *Spartina patens* replaced *Spartina alterniflora*, forming "high marsh." This process continued and gradually filled the inlet. As sea level rose, the marsh moved into the adjacent upland dominated by *Juncus gerardii*, black rush. Since salinity stress is reduced at elevations less frequently flooded by the tides, other plants, such as salt marsh lavender, *Limonium nashii*, sedges, and other grasses (Tiner 1987) also invaded the higher marsh. Davy (2000) gives detailed examples from Europe that involve different types of plants but show the same pattern of change as outlined by Redfield. These changes are called secondary succession.

Since salinity controls marsh plant distribution, changes in soil salinity affect succession. Soil salinity results from salinity of tidal water, frequency of flooding, rain diluting or washing salt away, and evapotransporation. Mid- and high-marsh areas that develop poor drainage can become more saline allowing more salt-tolerant species to invade and outcompete other higher plants (Bertness and Pennings 2000). In the highest marsh areas, vegetation composition is determined by frequency of tidal inundation and rain (Costa et al. 2000).

Human actions also affect salt marsh succession, a special case of autogenic succession, where humans are the biotic factor. Nitrogen pollution can shift the competitive relations of New England marsh plants allowing species to move higher in the tidal range than normal (Bertness and Pennings 2000). Shoreline development, especially if accompanied by removal of woody vegetation just landward of marshes, increases freshwater runoff and nitrogen inputs to adjacent marshes. The result is a movement of marsh zonation to higher elevations (Bertness et al. 2009). Reductions in salinity can also allow invasion of upland species. In Eastern USA, the invasive form of *Phragmites australis*, introduced from Europe in the 1950s, invades the upper edges of freshened marshes and dominates other vegetation. When combined with increased nitrogen input, the result can be a marsh vegetated by only *Phragmites* and *Spartina alterniflora* excluding all other marsh plants (Bertness et al. 2009).

Humans have modified marsh succession by introducing other species. *Spartina alterniflora*, introduced into San Francisco Bay marshes, has hybridized with the native *Spartina foliosa.* The hybrid, more vigorous than either parent, has taken over the marsh and the hybrid now occupies lower tidal elevations as well. *Spartina alterniflora*, introduced into Wallapa Bay, Washington State, has taken over mud flats and shoreline marshes (Strong and Ayres 2009). Similar changes due to introductions have occurred in Europe (Davy 2000). *Spartina* has been introduced into coastal areas in China and has increased elevations in developing marshes much as Redfield described for the USA.

Future Challenges

Future challenges are due to human actions. Climate change affects marsh succession with change in sea level and average water and air temperature. Marshes drown in the face of higher sea level and adjacent upland filled with building, roads, or commercial ports. With increased temperatures, mangroves move into temperate areas. A fivefold increase in abundance of black mangrove (*A. germinans*) from 2002 to 2009 in the Louisiana delta (Michot et al. 2010) illustrates the future marsh succession with climate warming.

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Section III

Landscape Ecology

Jere A. Boudell

Landscape Ecology of Wetlands: Overview 10

Jere A. Boudell

Contents

Abstract

Landscapes are heterogeneous areas consisting of interacting biotic and abiotic components. Landscape ecology emphasizes landscape pattern, connectivity within the landscape, and the interaction between landscape pattern and ecological process. The interaction of people with the landscape is also an integral component of landscape ecology due to the significant reciprocal nature of the relationship between humans and the environment. The section, *Landscape Ecology of Wetlands*, describes common landscape ecology principles and how they are being applied by wetland scientists and practitioners.

Keywords

Connectivity · Disturbance · Fragmentation · Landscape Ecology · Metacommunity · Metapopulation · Patch/Gap Dynamics · Pattern · Process ·

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Restoration · Riparian Buffer · Scale · Scaling Issues · Source-Sink Dynamics · Spatial Heterogeneity

Introduction

Landscapes are heterogeneous areas consisting of interacting biotic and abiotic components. Landscape ecology emphasizes landscape pattern, connectivity within the landscape, and the interaction between landscape pattern and ecological process. The interaction of people with the landscape is also an integral component of landscape ecology due to the significant reciprocal nature of the relationship between humans and the environment. Many landscape ecologists seek to apply research results to improve landscape management, conservation, and restoration. Wetland ecologists have also been applying landscape ecology principles explicitly or implicitly to answer basic questions concerning wetland landscapes and to improve wetland conservation and management. The section, *Landscape Ecology of Wetlands*, describes common landscape ecology principles and how they are being applied by wetland scientists and practitioners.

Historical Background

Landscape ecology has a relatively recent history. It was founded by Carl Troll, a German biogeographer, in the late 1930s as he sought to integrate ecology with geography to form one holistic branch of science. His effort culminated with the publication of *The Geographic Landscape and Its Investigation* in 1950. As a new discipline in Europe, landscape ecology focused on the landscape as mosaic of interacting abiotic and biotic features and also on the interaction between humans and the environment. It is through this holistic view of the environment that European landscape ecologists saw the applied sciences such as landscape management and conservation as vital to the discipline.

Landscape ecology was introduced to North America 30+ years after Troll's seminal work (Forman and Godron 1981, 1986; Risser et al. 1984). North American scientists dove into this emerging field, but rendered a slightly different form of the discipline. Rather than focusing on the applied sciences, North American scientists took a more basic and theoretical approach to landscape ecology and concentrated on spatial and temporal heterogeneity, the interaction of pattern and process, and scaling issues within the landscape. In response to these seemingly divergent paths, landscape ecologists on both sides of the world called for unification of the European and North American visions of the new emerging field of landscape ecology (Farina 2007; Wu and Hobbs 2002; Wu 2009a).

While landscape ecology was emerging as its own distinct interdisciplinary field, the principles of landscape ecology were being applied by wetland scientists and practitioners. Certainly, many of the central topics within landscape ecology were not unique to the field. In the following section, common principles and topics of landscape ecology are explained and how they are applied within the field of wetland ecology as described by section contributors is introduced.

Key Topics in Landscape Ecology

Pattern and Process

Spatial patterns across the landscape, and how processes influence and are influenced by patterns within the landscape, are a central focus of landscape ecology (Turner 1989; Turner et al. 2001). Spatial patterns describe the distribution of elements such as ecosystems, communities, species, nutrients, and water across the landscape. Processes such as species dispersal, natural disturbances, human land use, nutrient movement, and water flow influence the spatial patterns of the aforementioned landscape elements, but they are also influenced by the arrangement of the elements across the landscape. Emergent from the relationship between pattern and process are the principle topics of landscape ecology.

Spatial Heterogeneity

Landscapes are mosaics of elements that compose the landscape itself (Fig. 1). The *spatial heterogeneity* that emerges from the mosaic influences multiple ecological processes within the landscape and arises from the processes operating within the landscape (Pickett and Cadenasso 1995). The process of moving water creates rivers and associated riparian ecosystems that are elements of watersheds and contribute to landscape heterogeneity. Watersheds, in turn, affect multiple processes within the landscape such as nutrient movement as dissolved nutrients flow towards streams in watersheds and influence, and are influenced by, nutrient cycling in riparian zones. Wetlands also contribute to landscape heterogeneity and, like riparian ecosystems (Ward et al. 2002), are heterogeneous in space and time. Larkin (2017) describes types and sources of wetland heterogeneity and how wetland heterogeneity impacts community dynamics and ecosystem functions such as carbon sequestration and denitrification.

Disturbance also contributes to landscape heterogeneity (Levin and Paine 1974), and disturbance parameters such as severity, duration, and frequency are influenced by the patterns and locations of elements within the landscape. For example, natural disturbances such as floods can scour soil and vegetation, sculpting the floodplain and creating a patchy environment (Fig. 2). Watershed size and shape, relief ratio, and depth to bedrock are some landscape elements that impact flood disturbance parameters. A steep fan-shaped watershed with a high relief ratio increases flood magnitude and shallow bedrock decreases the time it takes for a flooding stream to reach bankfull stage.

Fig. 1 Heterogeneous riparian landscape consisting of abiotic and biotic components such as stream, soil, boulders, vegetation, and other organisms (Photo by Jere Boudell)

Fig. 2 Flooding of the San Pedro River, Arizona USA (Photo by Juliet Stromberg)

One of the functional outcomes of disturbance is that it produces areas, or patches, of newly available resources. This creates an opportunity for *patch/gap dynamics* (Levin and Paine 1974; Pickett and White 1987), further contributing to landscape heterogeneity. Patch/gap dynamics occur as species take advantage of the

Fig. 3 Regeneration from the seed bank after simulated flood scour (Photo by Jere Boudell)

newly available resources released by the disturbance by dispersing or immigrating into gaps and successfully inhabiting the previously occupied space. For instance, in response to floods, buried and dispersed wetland seeds once exposed to light and water can germinate and occupy the space created when floods scour vegetation and surface soils (Fig. 3).

Scale and Scaling Issues

The issue of *scale* is integral to the discipline of landscape ecology, and ecology in general (Wiens 1989; Levin 1992). Pattern, process, and scale are interrelated concepts. Phenomena can change spatially from millimeters to kilometers and temporally from seconds to centuries. Phenomena can also change through levels of organization from cells to ecosystems. For example, the pattern and process of nutrient cycling varies over the scale of meters to kilometers, minutes to eons, and from individual organisms to the ecosphere. Hierarchy theory incorporates scale and describes a system and its components. Within a hierarchical system, the top level of the system constrains the focus level and the lower level components impact and provide the mechanism that explains what is happening at the focus level (Turner et al. 2001). An investigator incorporating hierarchy theory into his/her investigation of plant productivity in wetlands needs to be cognizant of the top or broad scale controls on plant productivity such as climate and the bottom or finer scale controls such as light, water, and nutrient availability in plant microenvironments.

Scaling issues arise when investigators seek to understand phenomena and must pick the appropriate spatial and temporal scale and organizational level at which to study the phenomena. They can be confounded by scaling issues if they seek to apply their understanding of the phenomena to other scales and levels. Knowledge and application of scaling theory and methods can help investigators cope with scaling issues (Peterson and Parker 1998; Wu 2009b).

Connectivity and Fragmentation

Connectivity links landscape elements (Taylor et al. 1993). Connectivity also describes the flows of materials, such as water and nutrients, across landscapes as well as species dispersal and migration (Fig. 4). Research on connectivity typically focuses on the patterns and dynamics that significantly impact organisms that are reliant upon functional links through the landscape. The movement of water between wetlands, between groundwater and wetlands, and of migratory birds and amphibians among wetlands requires, and effectively constitutes, functional links between wetlands. Connectivity between wetlands and the role of connectivity in migration and wetland protection is discussed by Rittenhouse and Peterman. *Landscape genetics* focuses on the movement and interaction of elements and species within the landscape and the resulting effects on gene flow (Holderegger and Wagner 2008). Spear describes this newly emerging field in an essay on landscape genetics in wetlands and how it is being applied to understand in what way landscape elements and configurations impact gene flow of wetland species.

Metapopulations (Hanski and Gilpin 1997) and *metacommunities* (Holyoak et al. 2005) describe networks of populations and communities, respectively, whose members disperse between constituent populations and communities and potentially interact. Without linkage between populations and communities within the networks, these systems become fragmented, and metapopulations and metacommunities fall apart. Schooley and Cosentino discuss metapopulation theory and its application in the conservation of wetland species. A natural extension of metapopulation theory, metacommunity theory, is described by Boudell in an essay on metacommunity dynamics in riparian and lotic ecosystems. *Source-sink dynamics* of species, a component of metapopulation and metacommunity theory, occur as individuals migrate or disperse from robust populations to sparse populations. These dispersal dynamics ultimately help prevent local extinction of smaller, less successful populations (Pulliam 1988). Peterman and Rittenhouse follow up their essay on

Fig. 4 Wetland hydrologically connected to a nearby stream (Photo by Jere Boudell)

connectivity with an essay on source-sink dynamics in wetlands. Here they focus on source-sink dynamics of amphibians and reptiles and on the variety of methods used to study these dynamics.

Fragmentation of the landscape whether created by natural perturbations such as landslides or more commonly by humans through road construction and other activities disrupt connectivity within the landscape (Fisher and Lindenmayer 2007) (Fig. 5). As a consequence, fragmentation impacts landscape elements and interrupts ecological processes such as metapopulation and source-sink dynamics that are dependent on robust connections. Indeed, it is challenging to describe the many ecological processes that require connectivity without discussing fragmentation impacts. Thus, many of the aforementioned essays include discussions of fragmentation impacts. Cosentino and Schooley explicitly describe the effects of fragmentation on wetlands and specifically, the impact of fragmentation on dispersal and its outcome on biodiversity.

The Human Element

Humans are profoundly connected to the landscape they inhabit. Like other animals, our lives are significantly influenced by the environment affecting where and how we live. As a successful dominant species and as ecosystem engineers we, in turn, affect our environment. In recognition of the reciprocal nature of our relationship with the environment, landscape ecologists have explicitly included the human element within the discipline. Landscape ecology, particularly as it is practiced in Europe, has focused on how our surrounding landscape

Fig. 5 Riparian landscape fragmentation caused by repeated mowing (Photo by Jere Boudell)

influences where live, where we grow our food, where we recreate, and how these factors impact us culturally (Farina 2007). Its principles have been included in the applied sciences of conservation, management, restoration, sustainable agriculture, and landscape design. For example, maintaining connectivity and understanding how fragmentation disrupts landscape genetics, source-sink dynamics, and metacommunities are essential in order to manage and conserve species and allows managers to predict how species will react to the fragmented landscapes they inhabit.

In many of the contributed essays in the *Landscape Ecology of Wetlands* section, authors describe how to manage to maintain connectivity and/or mitigate the outcomes of disruption to the myriad dynamics occurring within the landscape. Several contributors solely focus on conservation and attempts to ameliorate anthropogenic impact on the environment through management and restoration of wetlands and riparian areas. Zedler and Miller describe the need for, and efforts to, restore degraded wetlands at the landscape scale. Ma focuses on the role of riparian buffers, the interface between streams and impacted lands such as agricultural fields, not only in wetland protection and water quality maintenance but also as habitat for many species.

Future Directions

Landscape ecology, as a relatively new discipline, is still evolving and defining itself. Many of the early issues that arose as the field was developing have been partially resolved through attempts to unify the European and North American perspectives on landscape ecology. And yet, there is room for further integration of the basic and the applied sciences within the discipline (Wu and Hobbs 2002; Wu 2009a). There are still topics to be investigated within the field such as interactions between the varying causes of spatial patterns, further exploration of heterogeneity, and looking beyond the typical temporal scale of many studies (Turner 2005). Our changing climate and its impact on disturbance regimes and species migration and dispersal are important topics that can benefit from the holistic approach of landscape ecology.

Landscape ecology has great potential to improve conservation and management. Indeed, the call was issued for applied landscape ecology as a subdiscipline in hopes that landscape ecology principles can be tested and incorporated into conservation, management, and restoration (Turner 2005). Wetland ecology can benefit from incorporating a broader view of the landscape by including both individual wetlands and surrounding environments and how patterns and processes can interact to influence ecological phenomena at multiple spatial and temporal scales. Wetland science can only improve and become more effective at solving pressing issues such as species conservation in the face of a continually fragmented and changing environment by incorporating landscape ecology principles and methods.

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Connectivity of Wetlands

11

Tracy A. G. Rittenhouse and William E. Peterman

Contents

Abstract

Connectivity implies connection or movement of resources across the landscape. Wetlands are dynamic ecosystems imbedded within or at the edges of larger systems and thus wetlands are natural "connectors" between upland and aquatic systems. Although many ecological processes have been used to describe wetland connectivity, here the focus is the movement of water and animals within and among wetlands as these are known agents of connectivity. Prior to these examples, a brief discussion of how connectivity has been used in the legal protection of wetlands demonstrates why connectivity is important to the conservation of wetlands.

Keywords

Animal movements · Amphibians · Buffer zone · Hydrology · Metapopulations · Isolated wetlands · Water · Wildlife

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Introduction

Connectivity implies connection or movement of resources across the landscape (Taylor et al. 1993). It is widely accepted that connectivity is an important ecological process that greatly influences resource patterns on the landscape. Research addressing this topic has identified several forms or definitions of connectivity. One common distinction is between structural connectivity and functional connectivity (Baguette and Van Dyck 2007). In other words, how connected is the vegetation or land cover on the landscape as opposed to how connected are populations, species, or ecosystem processes as a result of landscape features. Alternatively, connectivity has been defined based on three classifications: landscape connectivity, habitat connectivity, and ecological connectivity (Fischer and Lindenmayer 2007).

Wetland connectivity results from the connections provided by both biotic and abiotic resources, including water, nutrients, carbon, pollutants, and wildlife species, which move across the landscape. Wetlands are dynamic ecosystems at the edges of larger systems making them natural "connectors" between upland and aquatic systems (Brinson 1993). Furthermore, wetland connectivity extends to the movement of resources among wetlands that are spatially separated yet exchanges resources. Notably, the connectivity among these dynamic systems is not always readily evident, as resources may flow above, over, and below ground.

This entry focuses on the movement of water and wildlife among wetlands as these resources are widely known as common agents of connectivity among wetlands. Prior to these examples, this entry includes a brief discussion of how legal protections of wetlands have been justified based on this ecological process of connectivity.

Connectivity Used to Define Legal Protection of Wetlands

In the United States, some wetlands receive legal protection through the Clean Water Act. Regulated wetlands are defined based on their connection to navigable waters or inclusion in the definition of navigable waters (Downing et al. 2003). Therefore connectivity via the movement of water is central to the definition of regulated wetlands. The US Supreme Court case Solid Waste Agency of Northern Cook County (SWANCC) versus US Army Corps of Engineers (Corps) (351 U.S. 159, 2001) put into question federal jurisdiction over isolated wetlands. The Court found that the Corps' use of the Migratory Bird Rule, which was based on connectivity via movement of wildlife among critical wetland habitats, exceeded regulatory authority (Kusler 2001).

Legal documents use the term "isolated wetlands." This term implies that wetlands can be classified as connected or isolated. Classification is a way to simplify and improve understanding of systems, but classification does not represent the full complexity of dynamic systems. Wetland systems exist along gradients that constantly change over both space and time. Distinctions between hydrologically isolated or ecologically isolated wetlands have been made, yet the movements of water or wildlife among wetlands are both examples of functional connectivity. Guidance documents produced by the Corps are now under review and many groups have provided comments on wetland connectivity. Comments from Ducks Unlimited summarize various forms of connectivity documented in peer-reviewed literature (Schmidt 2011).

Other countries use a less regulatory approach to protection of water resources and wetlands. For example, the Water Framework Directive serves as the guiding document for maintaining and enhancing the quantitative status of all water bodies in the European Union. A review of the directive highlights the importance of connectivity and uses a definition that recognizes connectivity as process along a continuum (Peacock 2003). Connectivity is defined as “the ease with which organisms, matter or energy transverse ecotones between adjacent ecological units” (Ward et al. 1999). The review documents situations where connectivity is being lost in natural systems in the European Union, such as the loss of hydrological connections between main stem channels of rivers and other water bodies within the floodplain (Peacock 2003).

Wetland Connectivity via Flow of Water

Connectivity occurs via flow of water between wetlands and surrounding uplands, lakes, rivers, coastal waters, or other wetlands. Furthermore, it is the hydrologic conditions of wetlands that support the structure and function of these systems and provide distinctions among wetland types. The entry briefly highlights three wetland types with different hydrologic pathways. These example wetland types span the connectedness gradient and demonstrate how wetland connectivity varies across that landscape.

Ombrotrophic bogs provide a clear example of a wetland ecosystem with limited hydrologic connectivity with adjacent ecosystems. These are perched wetlands only receiving water and nutrients from precipitation and largely losing water via evaporation. Evaporation occurs as evapotranspiration through plants, and thus the level of the water table affects rate of water loss from the wetland to atmosphere (Lafleur et al. 2005).

Hydrologic connections are often hidden. For example, the prairie pothole region is a landscape that when viewed from aerial images contains a large number of spatially separated wetlands that appear to have limited connectedness via flow of water. Surface flow may differ among years with drought or abundant rainfall. The lack of surface water flow is not an indication of limited hydrologic connectivity. Pothole wetlands are fundamentally connected through groundwater (Tiner 2003). The placement in the landscape and ground water levels may influence whether a wetland is a recharge wetland, a flow through wetland, or a discharge wetland. Furthermore, the biological communities within the wetlands align along the continuum of the hydrologic connectivity to ground water (Euliss et al. 2004).

Tidal marshes and mangrove swamps provide examples of wetland ecosystems with readily apparent hydrologic connectivity. Water flows between the ocean and upland ecosystems such that the wetlands undergo daily and seasonal fluctuations of water level. Water levels that fluctuate affect salinity concentrations, transport nutrients, and aerate the soil within these wetlands (Furukawa et al. 1997; Kaplan et al. 2010; Menon et al. 2000).

Wetland Connectivity via Movement of Wildlife

Wetlands provide habitats for foraging, cover, overwintering, and reproduction to many wildlife species. Each species' unique use of wetland habitat creates connectivity with the surrounding upland habitat matrix, other wetlands, or both (Semlitsch and Bodie 2003). Disruption of movements among habitats can have profound implications for the persistence of local wildlife populations (Marsh and Trenham 2001). Migratory movements of birds among wetlands occurring at various densities and spatial patterns on the landscape affects the functional connectivity of wetlands at local, regional, and continental scales (Amezaga et al. 2002; Haig et al. 1998). A variety of other organisms also provide connectivity among wetlands including turtles, plants, and invertebrates (e.g., Roe et al. 2009).

Amphibian movements among wetlands are one apparent example of connectivity via movement of wildlife. The more common movements made by pond breeding amphibians are annual, round-trip migrations between terrestrial uplands surrounding wetlands that serve as nonbreeding habitat and wetlands that serve as breeding sites. Migratory movements have been documented by radio-tracking adult amphibians (Fig. 1) and these movements define the core habitat or life zone used by amphibians (Semlitsch and Bodie 2003). Core habitat is composed of both breeding and nonbreeding habitat (Fig. 2a). Wetland buffers must include and extend beyond core habitat to provide protection for amphibians (Semlitsch and Jensen 2001). Although the use of core habitat is highest near wetlands, local populations use habitat up to 664 m from wetlands (Rittenhouse and Semlitsch 2007; Fig. 3). Annual migratory

Fig. 1 Wood frog belted with a radio-transmitter. (Photo by Tracy Rittenhouse)

movements of amphibians connect wetlands to uplands, but can also provide connections among wetlands. For example, wood frogs breeding in vernal pools, migrate to forested wetlands for the summer, and then overwinter in upland hardwood stands (Baldwin et al. 2006). Migratory movements are distinct from the less common but ecologically very important dispersal movements (Semlitsch 2008; Fig. 2b, c).

Dispersal is the permanent movement of an individual from its birth site to the place where it reproduces (Johnson and Gaines 1990). Amphibian dispersal is a process that connects wetlands and is defined as the one-way movement from the natal wetland to a different wetland to breed (Semlitsch 2008; Fig. 2c). The majorities of individuals breed in their natal wetland and thus never disperse. Dispersal movements primarily occur during the juvenile life stage but occasionally adults may disperse. Dispersal is difficult to detect, but a 7-year mark-recapture study of marbled salamanders at 14 wetlands showed that 9% of successful breeding juveniles moved among ponds and <2% of breeding adults moved among ponds (Gamble et al. 2007; Fig. 4). Amphibian dispersal is an example of how wildlife provides connectivity among wetlands: a critical ecosystem function. Amphibian dispersal among wetlands also decreases the risk of extinction of local populations especially in short-lived species such as wood frogs, and thus successful dispersal among wetlands is crucial to the long-term persistence of amphibian populations (Harper et al. 2008).

The density of wetlands is one factor that affects connectivity among wetlands. The likelihood that amphibians will successfully disperse among wetlands is greatest when wetlands are in close proximity to one another. Connectivity among wetlands that results from amphibian movements has been used to identify areas of conservation priority for wetlands within the state of Massachusetts, USA (Compton et al. 2007). Land use in the uplands between wetlands also influences connectivity among wetlands provided by amphibians. Urban development (Hamer and McDonnell 2008; Safner et al. 2011), road or railways (Eigenbroda et al. 2008; Gibbs and Shriver 2005), and agriculture (Gagne and Fahrig 2007; Rittenhouse and Semlitsch 2006) are often highly resistant land cover types that prevent amphibian dispersal among wetlands. However, land use effects on dispersal can be nuanced upon detailed examination. For example, salamanders oriented toward and experienced lower desiccation risks in soybeans compared to corn (Cosentino et al. 2011).

Since dispersal movements involve a small proportion of an animal population, detecting connectivity among wetlands by observing the movement of individuals requires monitoring a large number of marked animals. Such endeavors are often time and cost prohibitive. Further, there is no guarantee that an observed dispersal event results in functional population connectivity via the transfer of genes. Landscape genetic methods provide an alternative approach to estimate connectivity, rate of movement, and/or differentiation of populations across the landscape (Storfer et al. 2007). Such methods are particularly suited for studies of wetlands because a wetland provides a discrete boundary from which to sample individuals. Landscape genetic approaches not only allow for the assessment of connectivity but provide a direct framework for determining the effect of the intervening habitat matrix on animal movement (Zeller et al. 2012). The effective distance between any two wetlands can be measured as a function of the intervening habitat matrix.

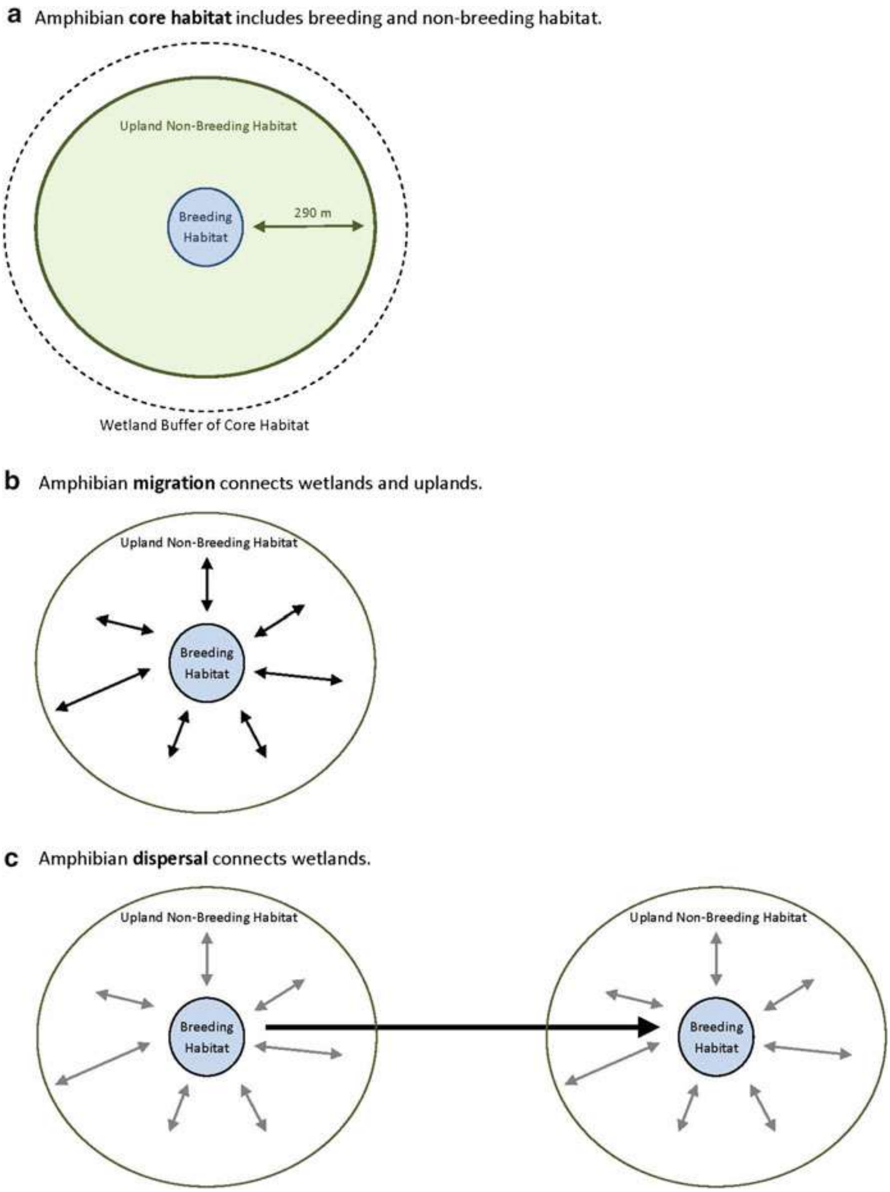

Fig. 2 Core habitat (**a**) is comprised of breeding and nonbreeding habitat. Migration (**b**) is round-trip, annual movements connecting wetlands to uplands. Dispersal (**c**) is one-way movement that provides connectivity among wetlands

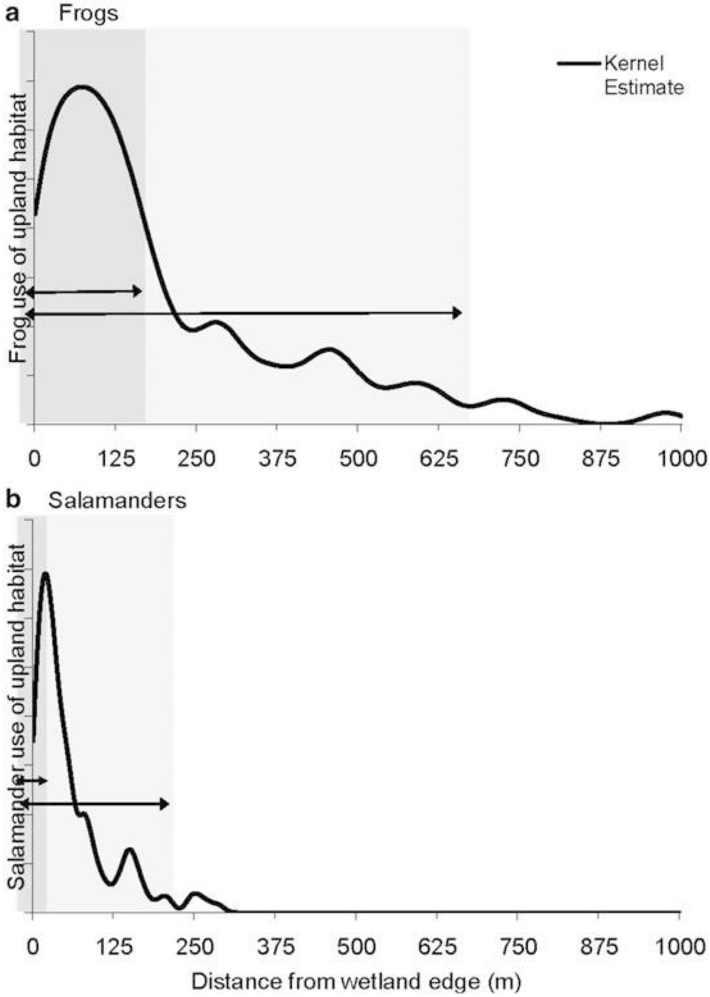

Fig. 3 Amphibian use of upland habitat surrounding wetlands. The wetland edge is at 0 m and the x-axis is the distance from the wetland edge. For frogs (**a**), 50% of a local population lives within 183 m and 95% of the population lives within 703 m. For salamanders (**b**), 50% of the local population lives within 41 m and 95% of the population lives within m. For all pond-breeding amphibians (frogs and salamanders together), 50% of the population lives within 93 m and 95% of population lives within 664 m (Reprinted from Biological Conservation, 139, Lloyd R. Gamble, Kevin McGarigal, Bradley W. Compton, Fidelity and dispersal in the pond-breeding amphibian, *Ambystoma opacum*: Implications for spatio-temporal population dynamics and conservation, 11 pp, Copyright (2007), with permission from Elsevier)

By evaluating genetic distance or gene flow estimates with these effective distance estimates, one can test a variety of hypotheses concerning the relative influence of different landscape features (e.g., roads, agriculture, etc.) as they relate to the successful dispersal of genes between wetlands.

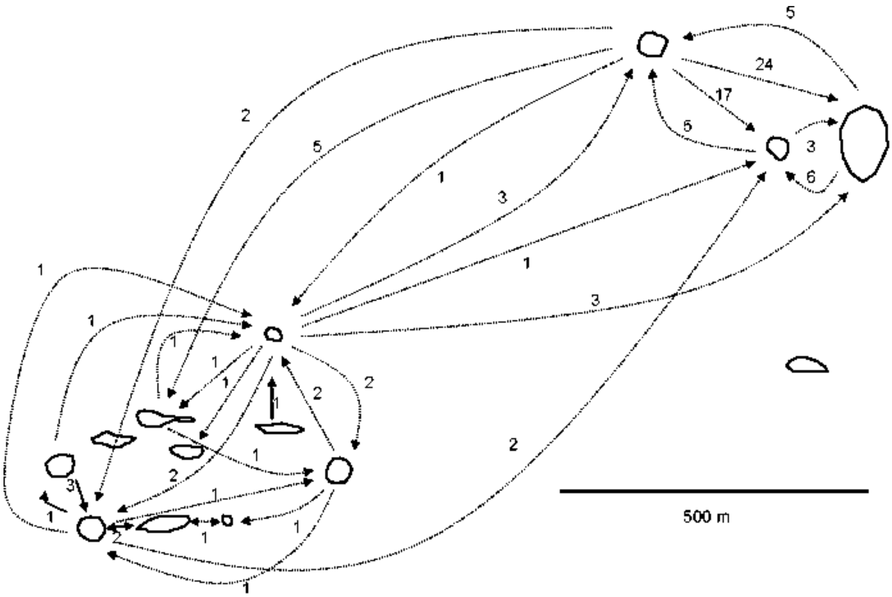

Fig. 4 This figure was published in Gamble et al. (2007) to describe a mark-recapture effort that documented the number of marbled salamanders dispersing among wetlands from 1999 to 2005 (Reprinted from Biological Conservation, 139, Gamble, McGarigal, Compton, Fidelity and dispersal in the pond-breeding amphibian, Ambystoma opacum: implications for spatio-temporal population dynamics and conservation, 247–57, 2007, with permission from Elsevier)

Conclusion and Future Challenges

Wetland connectivity concerns the movement of resources within and among wetlands. Future challenges will include how climate change is and will continue to alter connectivity in wetlands (Roelke et al. 2012). Hydrological flows among wetlands as well as the abundance of wildlife moving among wetlands are dynamic processes providing functional connectivity. These processes vary spatially and temporally, changing with climate and land use (Johnson et al. 2010; Wright 2010). Wetlands may become more or less functionally connected if the frequency between droughts and floods change, which may severely impact connectivity and metapopulation capacity of wetland networks in the future (Wright 2010). Alternatively, the magnitude of these events may change while the frequency remains relatively constant. Is wetland connectivity the same in a system with high magnitude/low frequency of movement as opposed to a system with low magnitude/high frequency? Furthermore, altered flow regimes within riverine systems are known to greatly affect biodiversity within wetlands. As climate change results in altered precipitation patterns, one should expect that effects on biodiversity in other wetland types could be as evident as that described in riverine wetlands (Bunn and Arthington 2002).

A difficult aspect of understanding connectivity via animal movement is the behavior and motivation of each individual in the population. Experiments that seek to describe these characteristics, such as movement rate, distance, path sinuosity, etc., often reveal tremendous intrapopulation variation. A promising future research path is the incorporation of these fine-scale, experimental data on movement behavior to parameterize individual-based models (IBMs) (DeAngelis and Mooij 2005). Such models incorporate individual variation in behavior, while assessing the broader-scale properties that emerge from simulation experiments. Once parameterized, IBMs can be utilized to conduct virtual experiments to determine the effects of land use, fragmentation, and habitat loss on movement and connectivity.

Although genetic data provide direct estimates of realized movement (i.e., gene flow), there is still great difficulty and uncertainty in determining the effects of the intervening habitat matrix on movement (Zeller et al. 2012). Resistance values for landscape features are often determined based upon expert opinion, but such approaches should be used with caution as expert-parameterized models may perform suboptimally (Charney 2012). Analytical methods and frameworks for determining optimal landscape resistance values as they relate to movement and/or genetic data remain an area of active research. Methods for determining least-cost paths (Sawyer et al. 2011) or landscape resistance (McRae 2006) are continually being evaluated and developed, and their integration with tools for assessing connectivity (Carroll et al. 2012) should make future regional assessments of wetland connectivity much more objective.

One caveat to promoting wetland connectivity is the potential for invasive species to propagate throughout the wetland network (Aquiloni et al. 2005; Peterson et al. 2013). Invasive species such as red crayfish (*Procambarus clarkii*) and the North American bullfrog (*Rana catesbeiana*) can have dramatic effects on local wetland processes and biodiversity. Further, wetlands are particularly susceptible to invasion by nonnative plants (Zedler and Kercher 2004). While functional connectivity of wetlands is critical for the maintenance of native biodiversity, serious consideration must be given to the potential for invasive species colonization.

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Corridors

12

Jere A. Boudell

Contents

Abstract

Corridors are contiguous areas of land, which connect habitat patches together. These features can also exist as routes within bodies of water or in the case of rivers; the river itself is the corridor. Corridors facilitate movement between patches, and can be used by land managers to alleviate the effects of fragmentation by providing a dispersal and migration passageway.

Keywords

Connectivity · Corridors · Dispersal · Migration · Patch

Definition

Corridors are contiguous areas of land, which connect habitat patches together (Farina 2007; Gilbert-Norton et al. 2010). These features can also exist as routes within bodies of water or in the case of rivers; the river itself is the corridor (Fig. 1). Corridors occur as natural landscape elements, protected strips of land, or constructed pathways, which serve to connect patches and isolated habitats. These connections facilitate movement between patches and can be used by land

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Fig. 1 A river corridor (Photo by Jere Boudell)

managers to alleviate the effects of fragmentation by providing a dispersal and migration passageway. However, just as corridors facilitate the movement of native species across the landscape, these also facilitate the movement and dispersal of nonnative species (Planty-Tabacchi et al. 1996). Within watersheds, rivers and associated riparian zones function as corridors (Naiman et al. 2005). Aquatic organisms travel across the landscape within the river network, while terrestrial organisms travel via the riparian zone where resources are often abundant. For example, seeds and other propagules are dispersed laterally across the floodplain as well as downstream when rivers flood, effectively dispersing species to other communities and potentially, other streams within the river network. Rivers often connect wetlands and thus serve as migration corridors between wetlands, particularly for wildlife that use different wetlands at different times within their life cycles (Naiman et al. 2005; Keddy 2010). Fish ladders are a type of artificial corridor that connects upstream and downstream river reaches when impoundment prevents fish migration.

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Dispersal and Wetland Fragmentation

13

Bradley J. Cosentino and Robert L. Schooley

Contents

Abstract

Wetlands provide habitat for a diverse array of aquatic and semiaquatic species, many of which provide direct economic and recreational value. Despite the ecosystem services provided by wetland fauna and flora, historical wetland loss has been dramatic. Wetland loss was >50% in the USA and 60–70% in Europe by the 1980s, with most losses resulting from agriculture and urban development. Although habitat loss can result from natural, stochastic events, anthropogenic habitat loss and subsequent fragmentation are among the most important drivers of biodiversity loss. One mechanism underlying the loss of biodiversity after habitat loss and fragmentation is the breakdown of wetland connectivity previously maintained by dispersal.

Keywords

Biodiversity · Connectivity · Dispersal · Fragmentation · Landscape

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Introduction

Wetlands provide habitat for a diverse array of aquatic and semiaquatic species, many of which provide direct economic and recreational value (Woodward and Wui 2001). Despite the ecosystem services provided by wetland fauna and flora, historical wetland loss has been dramatic. Wetland loss was >50% in the USA and 60–70% in Europe by the 1980s, with most losses resulting from agriculture and urban development (Groom et al. 2006). Although habitat loss can result from natural, stochastic events, anthropogenic habitat loss and subsequent fragmentation are among the most important drivers of biodiversity loss (Millennium Ecosystem Assessment 2005). One mechanism underlying the loss of biodiversity after habitat loss and fragmentation is the breakdown of wetland connectivity previously maintained by dispersal.

Importance of Dispersal

For wetland species, dispersal is the one-way movement of an organism between its natal wetland and the wetland in which it reproduces (Clobert et al. 2012). Dispersal is a fundamental biological process that influences individual fitness, population dynamics, and community structure (Clobert et al. 2012). Dispersal can be essential for the maintenance of biodiversity because it increases species persistence by counteracting demographic stochasticity (i.e., rescue effect; Groom et al. 2006). Additionally, dispersal can lead to gene flow between populations, which can be important for introducing beneficial alleles into populations and minimizing the loss of genetic variation due to genetic drift and inbreeding (Groom et al. 2006).

For many insects and vertebrates, dispersal occurs through direct, active movement of individuals overland or along water corridors. For fully aquatic species such as fish, physical connections among wetlands can be essential for dispersal. Alternatively, plants and some invertebrates rely on passive dispersal of gametes or zygotes. Waterfowl play an important role in maintaining wetland connectivity over large scales by transporting the pollen, seeds, and eggs of other species, including fish (Amezaga et al. 2002).

One-way dispersal from natal to breeding wetlands can be contrasted with movements of individuals to and from breeding wetlands (i.e., migration; Clobert et al. 2012). Many semiaquatic organisms have complex life cycles with both terrestrial and aquatic stages. For example, many amphibians use uplands for foraging and overwintering and wetlands for breeding, whereas many turtles use uplands for nesting and wetlands for breeding and foraging (Semlitsch and Bodie 2003). Movement between terrestrial and aquatic habitats is critical for semiaquatic organisms to complete their life cycles. Compared to among-wetland dispersal, breeding migrations are generally completed by direct, active movements of individuals.

Landscape Connectivity in Wetland Networks

For wetland species, landscape connectivity can be defined as the degree to which the landscape facilitates movement of organisms among wetlands or between wetland and upland habitats (Taylor et al. 2006). From a physical perspective, structural connectivity depends strictly on the spatial configuration of habitat in the landscape (Taylor et al. 2006). Wetland species should have high rates of dispersal between wetlands separated by short distances, whereas dispersal should be uncommon to wetlands that are spatially isolated. Similarly, breeding migrations of semiaquatic species are facilitated when required terrestrial habitat is adjacent to a wetland. The simplest index of structural connectivity is nearest-neighbor distance, which may be calculated as the distance from a wetland to the nearest wetland or upland habitat (Calabrese and Fagan 2004). More complex measures of connectivity include graph-theoretic and incidence function metrics that incorporate spatially explicit occupancy data and information on the focal species' dispersal ability (Calabrese and Fagan 2004).

Structural connectivity can be distinguished from functional connectivity, which considers how landscape features affect the movement behavior of organisms during dispersal (Taylor et al. 2006). Landscape features vary in their effects on movement due to differential resource availability, habitat complexity, predation risk, and physiological costs. For example, movement of the tiger salamander (*Ambystoma tigrinum*), a pond-breeding amphibian, was constrained by desiccation risk in upland habitats, and desiccation risk varied dramatically among habitats (Fig. 1; Cosentino et al. 2011a). Furthermore, the effects of desiccation risk on individual movement scaled up to affect the probability that tiger salamanders colonized uninhabited wetlands across a metapopulation (Cosentino et al. 2011b; ► Chap. 20, "Metapopulation Dynamics of Wetland Species" by Schooley and Cosentino). Wetlands surrounded by habitats with low desiccation risk were more likely to be

Fig. 1 Interpond movements of amphibians are influenced by desiccation risk experienced in upland habitats. For example, desiccation risk for tiger salamanders (*Ambystoma tigrinum*) in Illinois, USA is greater in prairie and corn than in forest and soybean habitats, and individuals orient their movements towards low-risk habitats (Cosentino et al. 2011a)

colonized than wetlands surrounded by habitats with high desiccation risk. Thus, movements among wetlands or between wetland and upland habitats can be limited by the composition and spatial configuration of habitats with variable dispersal costs. Least-cost modeling can be used to quantify the distances along paths between habitats that minimize dispersal costs (i.e., effective distances, Adriaensen et al. 2003), although parameterization of least-cost models requires detailed information about how habitats in the landscape affect movement behavior.

Compared to species that move over land, connectivity for species confined to aquatic habitat (e.g., fish) depends more strongly on aquatic connections such as stream linkages and overland sheet flow. Structural connectivity for fully aquatic species is predominantly determined by distances between wetlands along stream corridors or across areas inundated by water. However, stream flow and sheet flow can both depend on precipitation, introducing temporal variation in connectivity among wetlands (Leibowitz and Vining 2003). Functional connectivity for fully aquatic species may be limited by factors such as topography (e.g., stream cascades) and predation risk experienced within streams during movement between wetlands.

Wetland Fragmentation

Habitat fragmentation occurs when habitat is reduced in size and the distance between remaining habitat patches increases. Although wetlands are naturally patchy and separated by a terrestrial matrix, structural and functional connectivity can be disrupted by habitat loss and fragmentation in two main ways. First, when wetland habitat is lost or entire wetlands are destroyed, the density of wetlands decreases and the physical isolation of remaining wetlands increases (Fig. 2, Gibbs 2000). For species that move among wetlands over land or via passive dispersal,

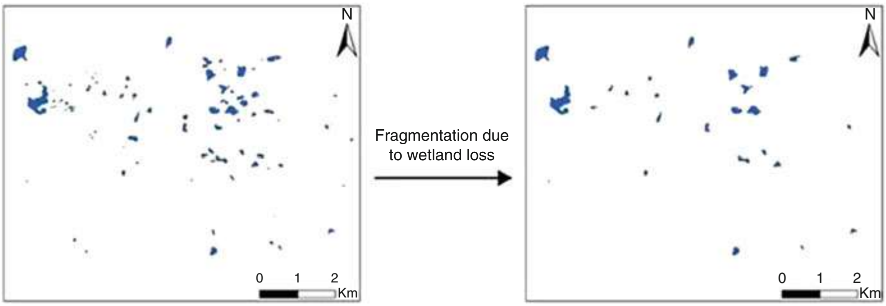

Fig. 2 Fragmentation increases spatial isolation of wetlands remaining after wetland loss

even small or isolated wetlands can be important by functioning as stepping-stones for dispersal to other wetlands (Semlitsch 2000). Thus, reducing the density of wetlands in the landscape, even small wetlands, decreases the likelihood of direct or indirect dispersal between wetlands. This reduction in structural connectivity can leave populations susceptible to extinction due to inbreeding and low population size (Clobert et al. 2012). Increased isolation also reduces the probability of colonization of uninhabited wetlands, which can be particularly important for regional persistence of species that exhibit metapopulation dynamics (Hanski and Gilpin 1997; ► Chap. 20, "Metapopulation Dynamics of Wetland Species" by Schooley and Cosentino). For species confined to aquatic habitats, the loss of stream linkages may also decrease wetland connectivity. Headwater streams are frequently lost from developed landscapes due to drainage, channelization, filling, piping, and groundwater withdrawal (Meyer and Wallace 2001).

Second, fragmentation occurs at a smaller spatial scale when terrestrial habitat surrounding wetlands is lost. Wetlands are linked to upland habitats through migratory movements of semiaquatic species (Semlitsch and Bodie 2003), and migration of semiaquatic species can be an important mechanism of energy and nutrient transfer between wetland and upland systems (e.g., Regester et al. 2008). In fragmented landscapes, maintaining wetland-upland linkages and connectivity among wetlands are primary goals for conserving biodiversity. Small, terrestrial buffer zones are often designated around wetlands to protect water resources and ecosystem services, but larger buffer zones are recommended to protect core terrestrial habitat used by semiaquatic species (Fig. 3; Semlitsch and Bodie 2003; Rittenhouse and Semlitsch 2007).

Maintaining Connectivity in Wetland Networks

In general, structural connectivity can be maintained by (1) minimizing the loss of wetlands and other aquatic habitats that function as corridors between wetlands (e.g., streams), and (2) maintaining core terrestrial habitats immediately surrounding wetlands. However, because some species move extensively among wetlands, maintenance of wetland connectivity likely requires management at the landscape scale (Amezaga et al. 2002). Within seasons, many species make temporary movements among wetlands that can vary dramatically in quality (e.g., vegetation structure, food, hydroperiod). For example, turtles tend to use permanent lakes during drought, but they move to ephemeral wetlands during periods of high precipitation (Roe and Georges 2007). Because wetlands vary in the resources they provide, conservation of wetland biodiversity likely requires a heterogeneous assemblage of wetlands each with terrestrial buffers, as well as the presence of upland habitats that allow seasonal movements and dispersal (see Fig. 3, Roe and Georges 2007).

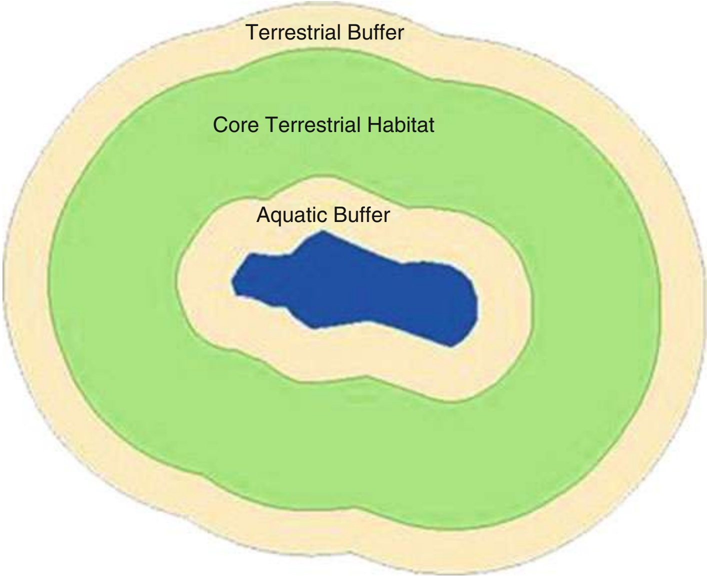

Fig. 3 Habitat buffers around wetlands are recommended to maintain wetland-upland connectivity for semiaquatic species. Semlitsch and Bodie (2003) recommend an aquatic buffer (30–60 m wide) to protect water resources, a zone to preserve core habitat for amphibians and reptiles (142–289 m, encompassing the aquatic buffer), and an additional 50-m buffer to protect core terrestrial habitat. Rittenhouse and Semlitsch (2007) indicate that some species require much greater areas of core habitat (upwards of 700 m) (The illustration is based on the original concept from Semlitsch and Bodie (2003))

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Disturbance

14

Jere A. Boudell

Contents

Abstract

Disturbance is a natural process that disrupts the environment and can impact species through injury, death, or migration. A disturbance also creates landscape heterogeneity if landscape elements (e.g., logs and soil) are modified or removed. Disturbance can remove competitors and release resources, allowing new species and individuals to occupy the newly available space. These dynamics are referred to as patch/gap dynamics.

Keywords

Disturbance · Gap dynamics · Patch dynamics

Definition

Disturbance is a natural process that disrupts the environment and can impact species through injury, death, or migration (Pickett and White 1987; Keddy 2010). A disturbance also creates landscape heterogeneity if landscape elements (e.g., logs and soil) are modified or removed. Disturbance can remove competitors and release resources, allowing new species and individuals to occupy the newly available space. These dynamics are referred to as patch/gap dynamics (Pickett and White

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Fig. 1 Flooding of the San Pedro River, Arizona, USA (Photo by Juliet Stromberg)

1987). Both small- and large-scale disturbances can occur. Small-scale disturbances take place when, for example, an animal digs through soil or a tree falls in a forest. Large-scale disturbances can be caused by volcanic eruptions or high magnitude floods. The degree of impact of a disturbance is related to the timing, frequency, magnitude, and duration of the disturbance. These properties are used to describe a disturbance event. For example, a 100-year flood caused by a significant weather event and significant snow melt might cause a high and sudden rise of water (Mitsch and Gosselink 2000; Naiman et al. 2005; Keddy 2010). Such an event can be described as a low frequency (approximately a 1% chance of occurring each year), high magnitude flood (high volume of water) event. This type of flood could occur during the fall (timing) and last for 1 week (duration). A flood with the aforementioned properties would impact the environment and associated species significantly.

Because disturbance can impact species considerably, it can act as a selective pressure on species inhabiting environments so that with repeated disturbances, species occupying the habitat will be adapted to such disturbances. For example, hydrology is the primary driver of disturbance that occurs at the landscape scale in wetlands and riparian (riverine) ecosystems (Mitsch and Gosselink 2000; Naiman et al. 2005; Keddy 2010). In frequently flooded riparian ecosystems, plant communities are dominated by annuals and short-lived perennials. Flooding rivers can scour vegetation and soil, deposit sediment along with propagules, and cause a temporary increase in water availability (Fig. 1). Flooding, within this context, is a natural disturbance event. Cessation of the disturbance caused by impoundment of the river by a dam would constitute an anthropogenic disturbance and significantly impact the environment and associated species that are adapted to flood disturbance.

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Ecosystem Function

15

Jere A. Boudell

Contents

Abstract
Ecosystem functions are the many processes conducted by ecosystems. The term, "ecosystem function," can also refer to the individual components involved in various ecosystem processes such as soil, water, and organisms. Ecosystem functions vary throughout the landscape and through time.

Keywords
Ecosystem Services · Process

Synonyms

Ecosystem processes

Definition

Ecosystem functions are the many processes conducted by ecosystems (Daily et al. 1997; Keddy 2010). The term, "ecosystem function," can also refer to the individual components involved in various ecosystem processes such as soil, water, and organisms. Nutrient cycling, pollination, fruit and seed dispersal, climate regulation,

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Fig. 1 Wetland habitat (Photo by Jere Boudell)

Fig. 2 Pollination in a riparian ecosystem (Photo by Jere Boudell)

and flood attenuation, are a few examples of ecosystem functions. Ecosystem functions vary throughout the landscape and through time. Nutrient cycling, for example, depends on multiple factors such as soil quality, microbial richness, temperature, and moisture availability, and these factors may be present in hotspots within the landscape but can also fluctuate with seasonal changes. A healthy ecosystem with all of its processes operating effectively can be described as a functioning ecosystem. When ecosystems are significantly impacted by natural or anthropogenic disturbances, ecosystem function can decline. Attempts to protect ecosystems include the identification of ecosystem functions that benefit humans. These functions are referred to as ecosystem services (Daily et al. 1997; Millennium Ecosystem Assessment 2005).

Many ecosystem functions occur within wetlands and riparian (riverine) ecosystems (Figs. 1 and 2) (Millennium Ecosystem Assessment 2005; Naiman et al. 2005;

Keddy 2010). Habitat is provided to the many species that populate these ecosystems year round or seasonally. Nutrient cycling and transport supports decomposition and the movement of nutrients up the food chain as vegetation absorb available nutrients and herbivores predate upon plants. Excess water drains from surrounding uplands into riparian ecosystems and is transported downstream. All of these functions and many more are present in healthy wetlands and riparian ecosystems.

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Ecosystem Services

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Jere A. Boudell

Contents

Abstract

Ecosystem services are the many processes conducted by ecosystems, which provide resources of benefit to humans and other organisms. Typically the inherent value of these services only becomes apparent after the services are interrupted and the disruption impacts the quality of human life.

Keywords

Ecosystem Services · Monetization

Introduction

Ecosystem services are the many processes conducted by ecosystems, which provide resources of benefit to humans and other organisms (Cairns 1995; Daily et al. 1997; Millennium Ecosystem Assessment 2005a). Decomposition of wastes by scavengers, detritivores, and microbes; the pollination of crops and other vegetation by pollinators; and protection from UV radiation by the Earth's ozone layer are a few examples of ecosystem services that support life. Typically the inherent value of

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Fig. 1 Urban riparian ecosystem providing multiple ecosystem services such as water filtration, nutrient cycling, dispersal, and habitat (Photo by Jere Boudell)

these services only becomes apparent after the services are interrupted and the disruption impacts the quality of human life (Avise 1994; Millennium Ecosystem Assessment 2005a). Attempts have been made to monetize these services in order to underline the value of environmental conservation and the protection of our natural resources to the general public and politicians. For example, one outcome of the Biosphere 2 project was the discovery that the total cost to artificially support one person per year through the ecosystem services provided by the man-made human terrarium was $9,000,000 (Avise 1994).

Wetland Ecosystem Services

Wetlands and riparian (riverine) ecosystems provide many ecosystem services (Fig. 1) (Millennium Ecosystem Assessment 2005b). The purification of water as polluted water migrates through surrounding wetland and riparian ecosystems is a critical service, which provides clean, fresh water. Riparian ecosystems attenuate flood waters as water drains from surrounding uplands into river channels. Rivers disperse seeds and support aquatic diversity. Wetlands and riparian ecosystems provide critical habitat, which supports innumerable species. These are just a few examples of the many ecosystem services provided by wetlands and riparian ecosystems.

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Gap and Patch Dynamics

17

Jere A. Boudell

Contents

Abstract
Gap and patch dynamics refer to functional differences in a landscape matrix, which are initiated by disturbances. Disturbances remove biomass and change the environment. This creates heterogeneity in the environment and provides opportunities for organisms to disperse into newly created spaces and establish.

Keywords
Gap · Matrix · Patch · Patch dynamics

Definition

Gap and patch dynamics refer to functional differences in a landscape matrix, which are initiated by disturbances (Pickett and White 1987; Wu 1995). Disturbances remove biomass and change the environment. This creates heterogeneity in the environment and provides opportunities for organisms to disperse into newly created spaces and establish. For example, a small-scale disturbance caused by a falling tree creates a gap. Because the tree has fallen, resources such as soil nutrients and light

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Fig. 1 Regeneration from the seed bank after simulated flood scour (Photo by Jere Boudell)

that were previously used by the tree are now available to other organisms. Seeds buried in the newly exposed soil may germinate into the space previously occupied by the tree and seeds from surrounding plants may disperse into the open space. The gap created by the fallen tree may also be considered a "patch." In riparian (riverine) ecosystems, patch dynamics at the landscape scale may happen relatively quickly as large floods restructure the floodplain removing whole communities and altering the environment (Naiman et al. 2005; Keddy 2010). Recovery would occur through seed dispersal from surrounding communities, germination from the seed bank, and as surviving woody species produce new shoots (Fig. 1). The patchy environment created by small- and large-scale disturbances can be viewed as a matrix of patches. The matrix of patches, the relationship between the patches, and the movement of organisms and materials such as nutrients and water between the patches are described as patch dynamics (Pickett and White 1987).

Gap and patch dynamics occur in space over the landscape and time as gaps and patches are created and filled. Patch dynamics can also be hierarchical where an ecosystem consists of a nested hierarchy of patches (Wu 1995). For example, a forest ecosystem consists of a matrix of larger patches created by a variety of events such as flooding and windfall, and within each of these patches is a matrix of smaller patches. A smaller patch created by a prior flood which has since been colonized consists of smaller patches created by fallen trees and areas disturbed by burrowing animals. Hierarchical patch dynamics would include the relationships and interactions between the levels within the hierarchy.

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Patch

18

Jere A. Boudell

Contents

Abstract
A patch is an area that is distinguishable within an otherwise fairly homogeneous distribution of one or more landscape elements.

A patch might also refer to an area of resources that becomes available after a tree has fallen or as an ecosystem remnant created via landscape fragmentation. Patchy ecosystems become elements of the overall landscape matrix and contribute to landscape heterogeneity.

Keywords
Disturbance · Habitat · Patch · Patch/Gap Dynamics

Synonyms

Habitat patch

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Definition

A patch is an area that is distinguishable within an otherwise fairly homogeneous distribution of one or more landscape elements (Pickett and White 1987; Keddy 2010). For example, a habitat patch might be identified as a community within a matrix of communities or as a localized concentration of soil nitrate (Figs. 1 and 2). A patch might also refer to an area of resources that becomes available after a tree has fallen or as an ecosystem remnant created via landscape fragmentation. Patchy ecosystems' become elements of the overall landscape matrix and contribute to landscape heterogeneity. Patches are often created through disturbance, and the processes that arise to create the patches and functional interplay within and between the patches are referred to as patch dynamics.

In wetlands and riparian (riverine) ecosystems, patches are created through a variety of processes (Keddy 2010). Patches may arise through disturbance caused by beaver or muskrats, for example, or by flooding as flood waters aggrade and degrade floodplains (Keddy 2010). Fire can create patches in Everglades wetlands via peat

Fig. 1 *Hymenoclea* sp. (burro brush) shrubland community, or patch, within a southwestern riparian ecosystem USA. This shrubland established in areas of sediment deposition created during flooding (Photo by Jere Boudell)

Fig. 2 This weedy community established within an urban riparian floodplain-upland forest matrix after repeated mowing to protect sewer line access (Photo by Jere Boudell)

burnings (Keddy 2010). Communities within riparian ecosystems, such as *Populus-Salix* (cottonwood-willow) forests in southwestern riparian ecosystems in the USA, can be destroyed during flood scour but reestablish in another location after seed dispersal from surrounding surviving populations.

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Metacommunity Dynamics of Riparian Ecosystems

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Jere A. Boudell

Contents

Abstract

A metacommunity is defined as a network of communities populated by potentially interacting species that are linked across landscapes through dispersal. Metacommunity theory unifies landscape or regional processes, namely dispersal, with local processes such as niche dynamics. This unification of ecological theories can provide wetland and riparian ecologists with a framework to understand how community structure and dynamics are impacted by both regional and local processes. Understanding how processes occurring at different scales structure wetland communities can help wetland ecologists predict how landscape fragmentation impacts biodiversity and ecosystem functioning.

Keywords

Mass effects model · Metacommunity · Neutral model · Patch dynamics model · Species-sorting model

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Introduction

A metacommunity is defined as a network of communities populated by potentially interacting species that are linked across landscapes through dispersal (Hanski and Gilpin 1991; Wilson 1992; Holyoak et al. 2005). Metacommunity theory unifies landscape or regional processes, namely dispersal, with local processes such as niche dynamics. This unification of ecological theories can provide wetland and riparian ecologists with a framework to understand how community structure and dynamics are impacted by both regional and local processes. Understanding how processes occurring at different scales structure wetland communities can help wetland ecologists predict how landscape fragmentation impacts biodiversity and ecosystem functioning (Holyoak et al. 2005).

Historical Background

Early metacommunity work focused on developing mathematical models (Wilson 1992; Mouquet and Loreau 2003) and testing model predictions (Cottenie 2005; Morgan Ernest et al. 2008). Four paradigms or models arose from this early work, and the majority of metacommunity research concentrated on evaluating the models experimentally and descriptively (Logue et al. 2011).

The four models that describe metacommunity dynamics all assume that species within metacommunities are able to disperse to all constituent communities (Leibold et al. 2004; Chase et al. 2005). What varies between the models is the degree of similarity between the communities, and the degree of impact of regional (e.g., dispersal) and local (e.g., environmental, interspecific interactions) processes on community characteristics such as taxonomic abundance, composition, and diversity (Holyoak et al. 2005).

The neutral and patch dynamics models propose that communities are similar and not explicitly distinct. In the neutral model, species within communities are functionally equivalent and species extinction occurs randomly due to demographic variation, resulting in "ecological drift" (Hubbell 2001, 2005). The patch dynamics model makes a similar prediction about species equivalency for most traits, but species vary in their competition-colonization abilities and species diversity is limited by dispersal (Leibold et al. 2004).

The two remaining metacommunity models state that communities are not similar and predict that species sort along environmental gradients (Holyoak et al. 2005). According to the species-sorting model, communities are arrayed along some environmental gradient and species populate the communities according to how their life-history trade-offs allow them to be successful (Chase et al. 2005). Dispersal does not allow a species to overcome its niche limitations. In the mass effects model, species are also limited by niche boundaries; however, dispersal enables a species to surmount niche restrictions (Holyoak et al. 2005). For example, a sink population may be sustained in a community in which a species is not typically successful because individuals are dispersed from source populations located in other communities (source-sink dynamics) (Mouquet and Loreau 2003).

Further work on metacommunity theory focused on incorporating additional local processes and evolutionary dynamics, understanding why evidence exists for multiple models, and investigating metacommunity dynamics in novel environments. Competition (Amarasekare et al. 2004) and predator-prey interactions (Pillai et al. 2011) are a few examples of local factors other than environment that have been investigated and included in the metacommunity framework. Urban and Skelly (2006) advocated for the inclusion of evolutionary approaches in metacommunity research as adaption impacts and interacts with a variety of local and regional processes. Other investigators have attempted to reconcile the four metacommunity models because they are not mutually exclusive in their predictions, and evidence supports multiple models, principally species-sorting and mass effects. Approaches have ranged from parsing communities into species guilds and matching results to the four models (Boudell and Stromberg 2008; Driscoll and Lindenmayer 2009) to suggestions that scale issues and temporal dynamics in metacommunities result in studies producing evidence that match multiple model predictions (Driscoll and Lindenmayer 2009). Finally, investigations of metacommunity dynamics in novel ecosystems reveal how metacommunity dynamics vary with different groups of organisms and across ecosystems. Riparian and lotic ecosystems are two prime examples.

Metacommunity Dynamics of Riverine Ecosystems: Riparian and Lotic

Riparian landscapes are excellent examples of metacommunities because they are spatially structured with constituent communities arrayed along moisture gradients that are connected by large-scale flood pulse dispersal (Fig. 1) (Naiman et al. 1993; Middleton 2002). Many plant species with varying life-history traits populate communities located along a declining moisture and flood disturbance gradient (Fig. 2) (Naiman et al. 2005). Investigations of southwestern riparian metacommunities found dispersed propagules (seeds and vegetative remnants capable of resprouting) across entire floodplains, linking the various floodplain communities together through a regional propagule bank (Figs. 2 and 3) (Boudell and Stromberg 2008). The regional propagule bank enables floodplain species to respond to changing environmental conditions locally by providing a pool of propagules that can successfully germinate or resprout in communities when supportive environmental conditions exist. Both the species sorting and mass effects metacommunity models make reasonable predictions about southwestern riparian metacommunities (Boudell and Stromberg 2008). Wetland plant species exhibit strong species-sorting dynamics as they are adapted to specific moisture conditions and are unable to overcome niche limitations through dispersal. Whereas facultative wetland species may be able to persist in communities outside their ideal moisture range due to source-sink dynamics and so match mass-effects model predictions (Boudell and Stromberg 2008). While environment was found to have a strong impact on southwestern riparian plant

Fig. 1 Researcher seining for seeds in the Verde River, Arizona (Photo taken by Danika Setaro)

communities, Muneepeerakul et al. (2008), using a neutral theory approach, demonstrated the importance of dispersal directionality and landscape structure to riparian plant diversity.

The application of metacommunity theory to lotic, or instream riverine ecosystems, is relatively new and has revealed unique challenges (Swan and Brown 2011). Riverine ecosystems form dendritic networks and this hierarchical network can constrain dispersal (Fagan 2002; Auerbach and Poff 2011; Brown et al. 2011). Because aquatic species such as fish and invertebrates are typically mobile, and water movement and dispersal of aquatic organisms are oriented downstream, the formation of distinct communities can be confounded (Auerbach and Poff 2011; Brown et al. 2011). Despite these challenges, metacommunity research in lotic ecosystems has found support for species-sorting and mass-effect dynamics (Brown and Swan 2010; Auerbach and Poff 2011). A graphical model of riverine metacommunity dynamics by Brown et al. (2011) illustrates the unique dispersal dynamics of lotic organisms and the potential impact of local factors on community structure and species interactions (Fig. 4). Metacommunity theory promises to be a

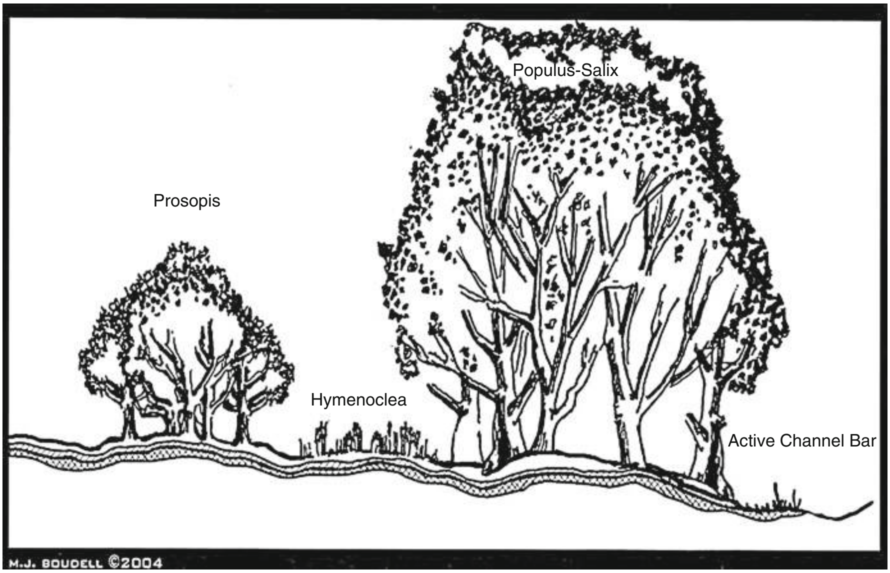

Fig. 2 Southwestern riparian floodplain containing typical dominant plant communities arrayed along a declining moisture and disturbance gradient. Propagules, here represented by *hash lines*, are dispersed across entire floodplain during large scale flood events (illustration MJ Boudell)

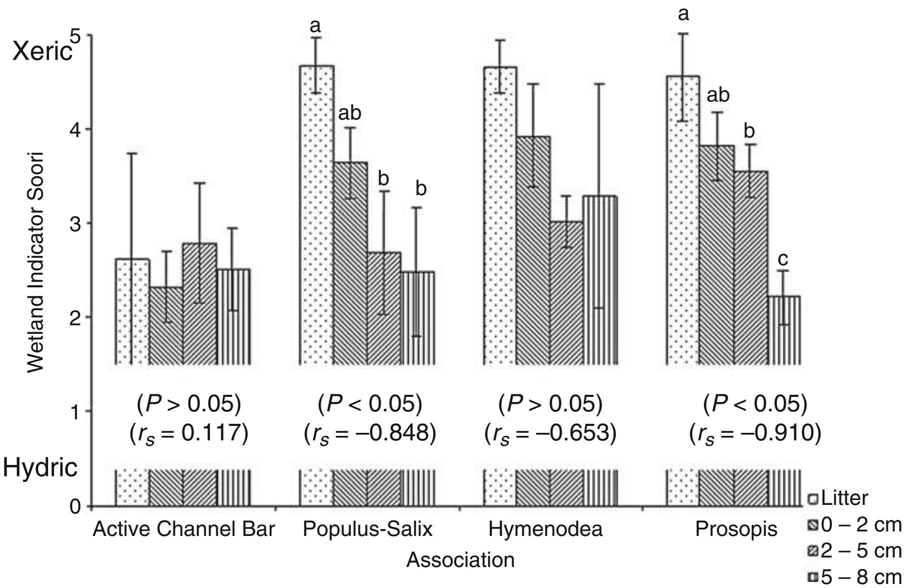

Fig. 3 Mean wetland indicator score, ±1 SD, per m^2 by community (association) and depth in a southwestern riparian plant metacommunity. Wetland (hydric) species have lower wetland indicator scores. Communities are arranged along a distance gradient with active channel bar closest to the river channel. Results are from ANOVA and correlation analyses (Depth vs. wetland indicator score; modified from Boudell and Stromberg 2008)

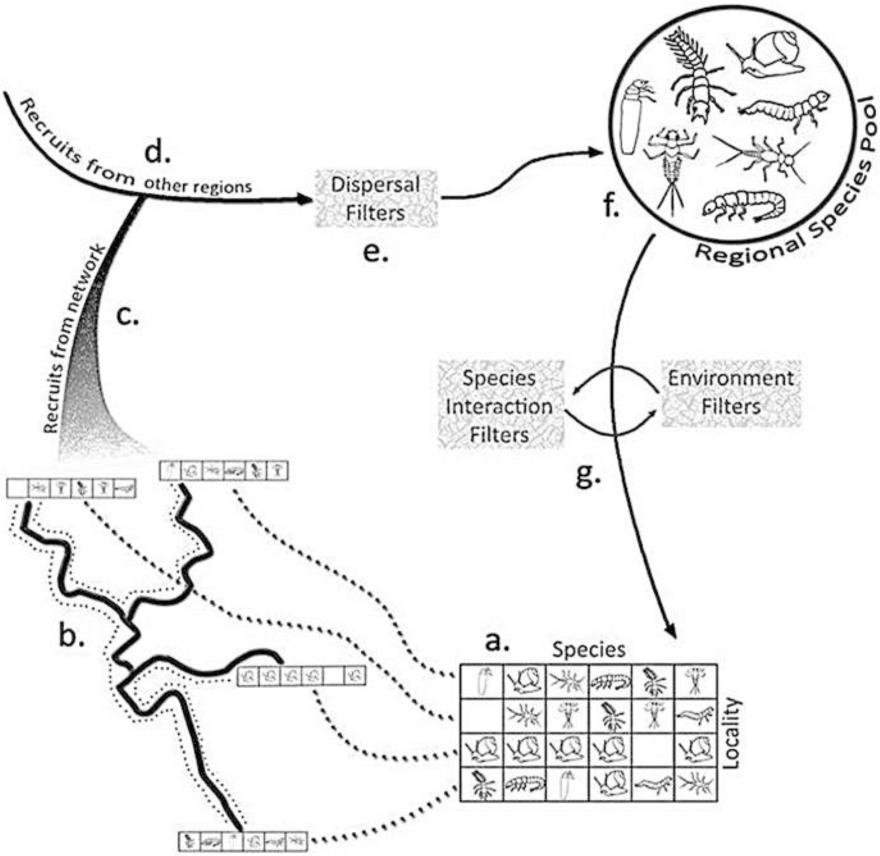

Fig. 4 Interactions between local (environmental conditions and species interactions) and regional factors (dispersal and landscape-scale extinction dynamics) in determining local community composition in riverine metacommunities. (**a**) The regional species pool represents the organisms available to occupy particular localities. (**b**) Organisms from the regional species pool are filtered based on environmental factors and species interactions. (**c**) A local community can be defined as an assemblage of species within a particular habitat. In this case, each table cell is a microhabitat within a local community occupied by a particular species or left unoccupied (*blank cells*). (**d**) The riverine network. *Large dotted lines* illustrate the location of each local community in the habitat x species matrix. *Small dotted lines* represent dispersal between local communities in the riverine network. (**e**) Dispersal of recruits into the regional species pool. This pathway includes both within-network dispersal (also illustrated by the *small dotted lines* in [**d**]) and out-of-network dispersal. (**f**) Recruits from other regions (i.e., neighboring riverine networks) that can potentially contribute to the regional species pool. (**g**) Dispersal filters, which are limits or barriers to the dispersal of organisms from a locality into the regional species pool. Filters may be particular to a species, or may affect any number of species. (Benthic macroinvertebrate illustrations are courtesy of Laura E. Smith. Reprinted with permission from Brown et al. (2011))

powerful tool to understanding community assembly of lotic organisms (Brown et al. 2011; Swan and Brown 2011).

Future Directions

As theoretical and empirical work on metacommunity dynamics continues, new questions and developments arise. The integration of the four metacommunity models and incorporation of other processes such as predator-prey interactions and evolutionary processes will produce a more comprehensive theory that realistically incorporates the multiple factors that structure communities (Boudell and Stromberg 2008; Driscoll and Lindenmayer 2009; Logue et al. 2011). Exploring metacommunities in varying ecosystems, such as wetlands and riparian ecosystems, and focusing on the community dynamics of a variety of species should also contribute to the development of a more powerful theory capable of predicting community structure and dynamics (Logue et al. 2011).

Perhaps the ultimate challenge for metacommunity theory is the ability to accurately predict the outcome of landscape fragmentation. The metacommunity framework provides a key approach to asking questions related to the management and restoration of riparian and lotic ecosystems. Individual species and communities will respond to anthropogenic disturbances such as dams, diversions, and levees, which dramatically alter environmental conditions and can disconnect constituent communities from riparian metacommunities. As wetland and riparian ecosystems are continually degraded and lost, it is more important than ever to understand how landscape fragmentation results in disruption of dispersal dynamics and how this disruption impacts local communities.

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Metapopulation Dynamics of Wetland Species

20

Robert L. Schooley and Bradley J. Cosentino

Contents

Abstract

For species inhabiting naturally patchy or fragmented landscapes, conservation often is guided by metapopulation theory. A metapopulation is a set of spatially separated populations connected by movement of individuals among populations. The metapopulation can persist, despite extinctions of local populations, if populations are connected enough to allow for adequate recolonization of vacant habitat. Because wetlands occur as geographically isolated habitats, many wetland-associated species could display metapopulation dynamics. However, classical metapopulations may be rare, and metapopulations can have a diversity of spatial structures. Practical metapopulation approaches are grounded in the "area-isolation paradigm" in which the area of a habitat patch is the main predictor of local extinctions, and connectivity to other source populations is the main predictor of colonization. The generality of the area-isolation paradigm has been questioned, however, and its shortcomings relate to the need to consider habitat heterogeneity. Wetlands can differ in habitat quality and they are embedded in a heterogeneous terrestrial matrix. Functional connectivity of metapopulations depends on how movements of individuals interact with the terrestrial habitat

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matrix. Despite these complexities, recognition of metapopulation dynamics for wetland species has forced managers to think about biodiversity conservation at landscape scales and highlights the importance of wetland-upland linkages.

Keywords
Colonization · Connectivity · Extinction · Habitat heterogeneity · Metapopulation · Wetland species

Introduction

For species inhabiting naturally patchy or fragmented landscapes, conservation often is guided by metapopulation theory (Hanski and Gaggiotti 2004). A metapopulation is a set of spatially separated populations connected by movement of individuals among populations. The whole metapopulation can persist, despite extinctions of local populations, if populations are connected enough to allow for adequate recolonization of suitable but vacant habitat. This on-and-off blinking of populations reflecting a balance between local extinctions and recolonizations is the hallmark of classical metapopulation dynamics (Fig. 1). The metapopulation concept was formalized by Levins (1969), but it has deeper roots including ideas on spatial dynamics of populations promoted by Andrewartha and Birch (1954). Maturation of the metapopulation approach and its application to biodiversity conservation, however, has mainly resulted from development of practical metapopulation models by Hanski and his colleagues (Hanski 1997; Hanski and Gaggiotti 2004).

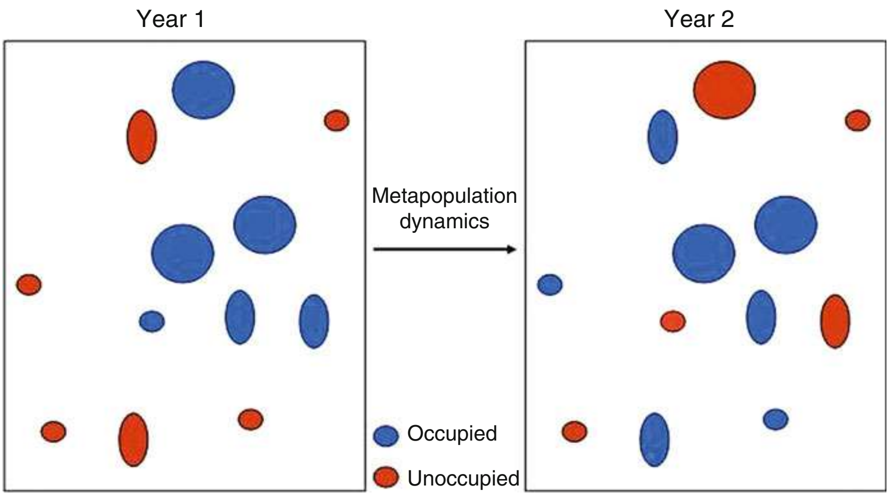

Fig. 1 Metapopulation dynamics for a hypothetical network of 12 wetlands. Wetlands in blue are occupied by a species; wetlands in red are unoccupied. Some wetlands experience local extinctions between years, and some vacant wetlands are recolonized

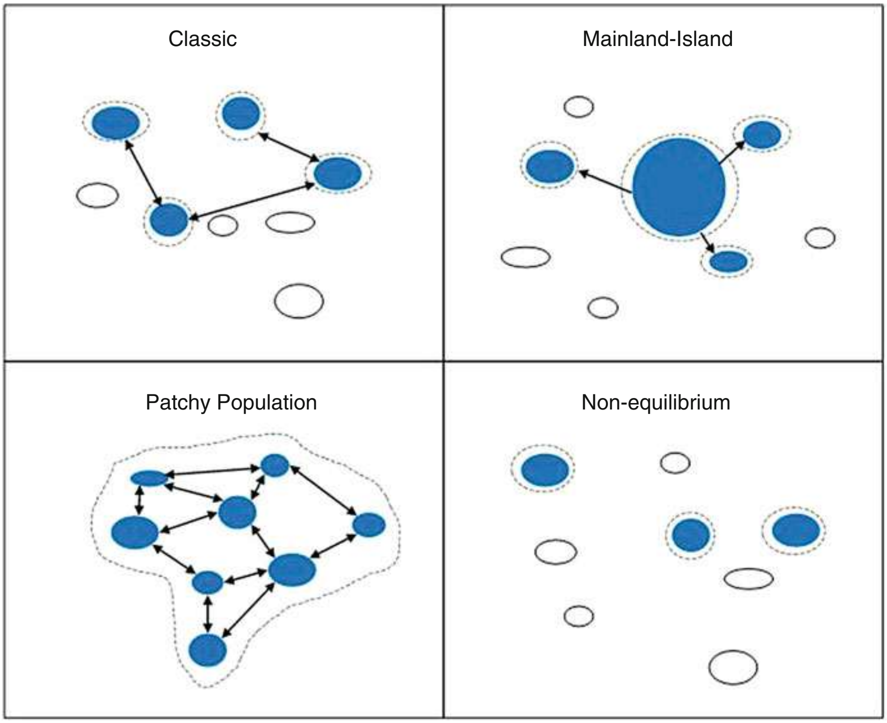

Fig. 2 Types of metapopulations (Harrison and Taylor (1997) with permission of Elsevier). Filled circles represent occupied wetlands; open circles represent unoccupied wetlands. *Arrows* indicate dispersal; *dotted lines* indicate boundaries of local populations. Metapopulation structure depends on degree and directionality of dispersal among wetlands

Hanski (1997) provided four conditions for a classical metapopulation: (1) suitable habitat occurs as discrete patches that can be occupied by local breeding populations, (2) the largest local population still has a nontrivial risk of extinction, (3) habitat patches are not too isolated so that recolonization is possible, and (4) local populations have asynchronous dynamics. Because wetlands often occur as geographically isolated habitats, many wetland-associated species could display metapopulation dynamics (e.g., Sjögren Gulve 1994; Schooley and Branch 2009). However, Marsh and Trenham (2001) warned against a naïve adoption of the "ponds-as-patches" view for wetland species such as amphibians that might not always function as metapopulations. More generally, Harrison and Taylor (1997) cautioned that classical metapopulations could be rare and that metapopulations in nature can have a diversity of structures including mainland-island, patchy population, nonequilibrium metapopulation (Fig. 2), and mixed structures at different spatial scales.

Area-Isolation Paradigm

Practical metapopulation approaches (Hanski and Gaggiotti 2004) are grounded in the “area-isolation paradigm” in which the area of a habitat patch is the main predictor of local extinctions, and connectivity to other source populations is the main predictor of colonization. For wetland species, large wetlands should contain relatively large populations and thus be less susceptible to extinction due to demographic stochasticity than are small populations occupying small wetlands. This area effect on local extinction probability holds true for some spatially structured wetland species such as round-tailed muskrats (*Neofiber alleni*; Schooley and Branch 2009) and painted turtles (*Chrysemys picta*; Cosentino et al. 2010). Size of wetlands also can affect colonization probabilities through the target effect in which larger wetlands are more likely to be colonized by individuals dispersing from other wetlands (Schooley and Branch 2009; Cosentino et al. 2010, 2011). Colonization of vacant wetlands should depend most on how isolated they are from other occupied wetlands. In fact, colonization probability has been positively correlated with connectivity for wetland-associated wildlife (Schooley and Branch 2009; Cosentino et al. 2010, 2011). Connectivity of wetlands can also reduce the likelihood of local extinctions through the rescue effect in which populations are bolstered by dispersers and maintained above levels highly vulnerable to effects of demographic stochasticity (Brown and Kodric-Brown 1977; Cosentino et al. 2011). For instance, isolated populations of the pool frog (*Rana lessonae*) in Sweden are more extinction prone than are more connected populations (Sjögren Gulve 1994).

Despite this evidence that patch area and isolation can be important predictors of extinction-colonization dynamics, the generality of the area-isolation paradigm has been questioned. Pellet et al. (2007) tested the ability of area and isolation to predict metapopulation dynamics of 10 species from diverse taxa including amphibians and concluded there was not overall strong support for these two key variables arising from metapopulation theory. Patch size may not always be a good surrogate for population size, and patch isolation measured as simple, straight-line distances among patches may be inadequate (Pellet et al. 2007). Likewise, Prugh et al. (2008) conducted a meta-analysis of 785 animal species and concluded that patch area and isolation explained a surprisingly small amount of the variation in patch occupancy (i.e., the outcome of extinction and colonization processes).

Habitat Heterogeneity and Metapopulations

Many of the shortcomings of the area-isolation paradigm as applied to metapopulations relate to the need to consider habitat heterogeneity in multiple ways. Real landscapes are not binary; wetlands can differ in habitat quality and are embedded in a heterogeneous terrestrial matrix (Schooley and Branch 2007, Prugh et al. 2008). Habitat quality of wetlands can depend on many factors including hydroperiod, water chemistry, vegetation composition, resource abundance, and predation risk (e.g., Sjögren Gulve 1994; Cosentino et al. 2010, 2011). For instance,

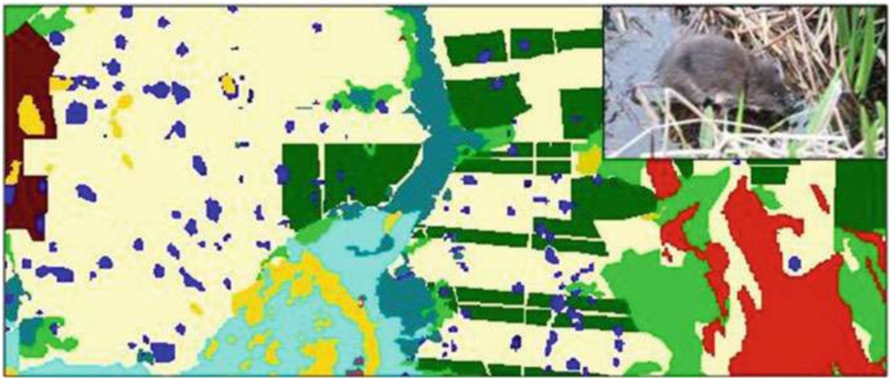

Fig. 3 The round-tailed muskrat (*Neofiber alleni*) is a species of conservation concern that exhibits metapopulation dynamics across depression marshes (*blue polygons*) in Florida, USA. Local extinctions and recolonizations depend not only on wetland area and geographic isolation, but also on habitat quality of wetlands and upland habitat between wetlands (Schooley and Branch 2009). Occupancy is less likely for wetlands surrounded by pine plantations (*dark green polygons*) and other forested upland habitat

site occupancy by round-tailed muskrats strongly depends on the cover of maidencane grass (*Panicum hemitomon*) in marshes because maidencane is the main diet item for muskrats, and they use it to build lodges (Fig. 3; Schooley and Branch 2007, 2009). Ignoring habitat quality is the likely reason that wetland size might not be a good substitute for local population size. In some situations, habitat quality can be the main driver of species distributions, extinctions are not stochastic but instead deterministic due to habitat becoming unsuitable, and species form a habitat-tracking metapopulation across fragmented landscapes (Thomas 1994). Habitat quality of wetlands could also affect colonization if dispersers actively select habitat before settling (Clobert et al. 2012), or the quality of nearby wetlands acting as sources of dispersers is variable in space (Schooley and Branch 2011).

Dispersal is a complex process influenced by many factors (Clobert et al. 2012). Functional connectivity of metapopulations depends on structural aspects of the landscape such as geographic isolation of wetlands, but also how movements of individuals interact with the terrestrial habitat matrix (▶ Chap. 13, "Dispersal and Wetland Fragmentation" by Cosentino and Schooley). Wetland pairs separated by the same straight-line distance could have different functional connectivity because the intervening habitat matrix varies in terms of movement costs (Fig. 3). For example, movement of Natterjack toadlets (*Bufo calamita*) is affected by land cover with forests providing greater resistance than more open habitats (Stevens et al. 2004). Hence, realized isolation of wetlands is species-specific and highly dependent on movement behavior.

For semiaquatic species, the role of habitat heterogeneity is even more complicated. For pond-breeding amphibians with complex life cycles that include a terrestrial stage, the distribution and quality of upland habitat may be especially important (Marsh and Trenham 2001). The joint habitat quality of wetlands and adjacent uplands may determine habitat suitability and the spatial distribution of species.

As such, delineating habitat patches for a wetland metapopulation could require going beyond just identifying wetlands from a map.

Conservation Implications and Future Challenges

Despite these complexities, many of which are related to how we should integrate habitat heterogeneity into the area-isolation paradigm, the metapopulation concept has been influential for land managers. The recognition of potential metapopulation dynamics for wetland species has forced managers to think about biodiversity conservation and restoration at landscape and regional scales instead of at the scale of single wetlands. Metapopulation dynamics also highlight the importance of landscape connectivity and wetland-upland linkages. Future challenges for conservation of wetland species forming metapopulations include maintenance of adequate habitat quality and connectivity despite ongoing habitat loss, fragmentation, and disturbance to the terrestrial matrix. Some species might already be functioning as nonequilibrium metapopulations (Fig. 2) just waiting for dynamics to play out. Finally, interactions between climate change and metapopulation dynamics (Opdam and Wascher 2004) will be crucial to persistence of many wetland species. Increased frequency of disturbances from large weather events will increase the temporal variability of wetland quality, and conservation of connectivity will be critical for facilitating shifts in geographic ranges.

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Riparian Buffer Zone for Wetlands

21

Maohua Ma

Contents

Abstract

Riparian buffer zone is an ecotone located between human-disturbed lands and wetlands, lakes, or rivers. The main functions of the riparian buffer zone is to protect wetland ecosystem through flooding control, water protection, soil conservation, habitat provision for wild species diversity, and the influence they have on ecosystem processes in wetlands. Width and vegetation composition of riparian buffer zone are the key features that enhance its functions essential to establishing and maintaining healthy wetlands. To achieve a sustainable buffering

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function for wetlands, careful multidisciplinary planning based on ecological knowledge and socioeconomic context is necessary.

Keywords
Riparian · Buffer zone · Wetlands · Water protection

Introduction

Definition

The word "riparian" is derived from Latin word "ripa," meaning river bank. One of the accepted definitions was termed by Lowrance et al. (1985): "Riparian ecosystems are the complex assemblage of organisms and their environment existing adjacent to and near flowing water." A simple definition comes from Malanson (1993): "the ecosystems adjacent to the river." In wetland ecosystems, a riparian buffer zone is the interface located between human-disturbed lands (agricultural or industrial areas) and wetlands, lakes, or rivers (Figs. 1 and 2). Vegetation-covered habitats and communities along the river margins and banks are called riparian vegetation.

The riparian buffer zone along the water bodies provides more than just a beautiful landscape. The main ecosystem function of the riparian strip is to protect wetlands (Gregory et al. 1991). The riparian zones are also significant in ecology, environmental management, and civil engineering because of their roles in water protection, soil conservation, wild species diversity, and the influence they have on ecosystem processes in wetlands (Castelle et al. 1992). Vegetation in the buffer zones is important for its function to protect from leaked agrochemicals and eroded soil (Jenssen et al. 1994). The buffer zones also provide important habitats for animal species (e.g., butterflies, bees, birds) in wetland ecosystems (Forman 1995). In some regions, the terms riparian woodland, riparian forest, riparian boundary/margin, or riparian strip are used to characterize a riparian buffer zone (Fig. 3).

Background

Riparian buffer zones begam to be used for wetland protection and restoration only from last century when humans realized the ecological importance of wetlands and negative impacts of human activities on the wetlands. Since early civilization, many cultures have learned to live in harmony with wetlands and have benefited economically from surrounding wetlands, whereas other cultures quickly drained the landscape (Dugan 1993). The propensity in Eastern civilizations, e.g., the ancient Babylonians, Chinese, Egyptians, and the Aztec, was not to drain the landscape, as is frequently done in the West, but to work within the aquatic landscape, albeit often in a heavily managed way. In the West, since wetlands were considered as disastrous realms, sources of disease, and trackless wasteland, wetlands were drained and filled so that people could build on them, grow crops on them, and build roads across them

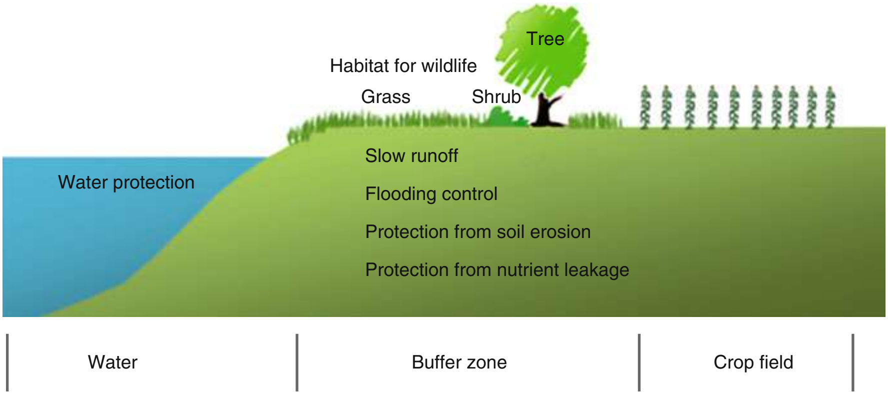
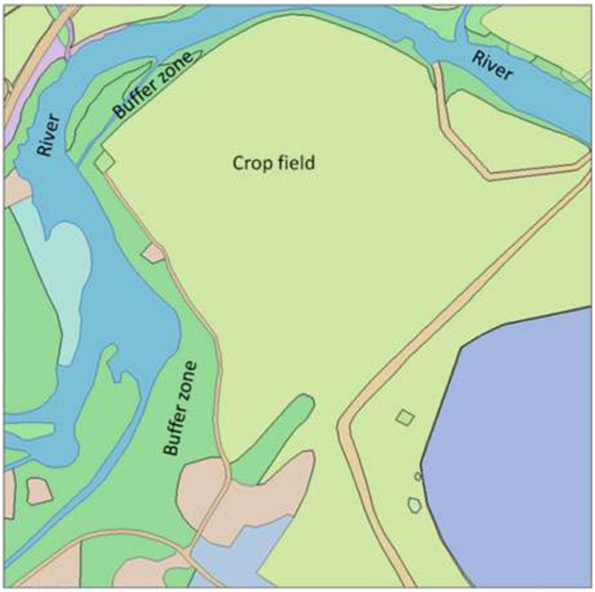


Fig. 1 Cross section of riparian buffer zone and its main functions

Fig. 2 A GIS map of a typical land mosaic composed of a river, buffer zone and crop field in a watershed agricultural landscape, southern Finland



(Mitsch and Gosselink 2007). No matter in the East and the West, heavy utilization, strong disturbances from human activities, drainage, and destruction of wetlands have caused rapid degradation and loss of wetland ecosystem (Mitsch and Gosselink 2007).

Starting in the 1970s, we became aware of the value and importance of wetlands and that they were disappearing. The Ramsar Convention is an international treaty signed in Ramsar, Iran, for conservation and wise use of wetlands (www.ramsar.org). The agreement was signed on February 2, 1971, and came into force from December 21, 1975 (Mattews 1993). It is one of the oldest specific conventions that deal not

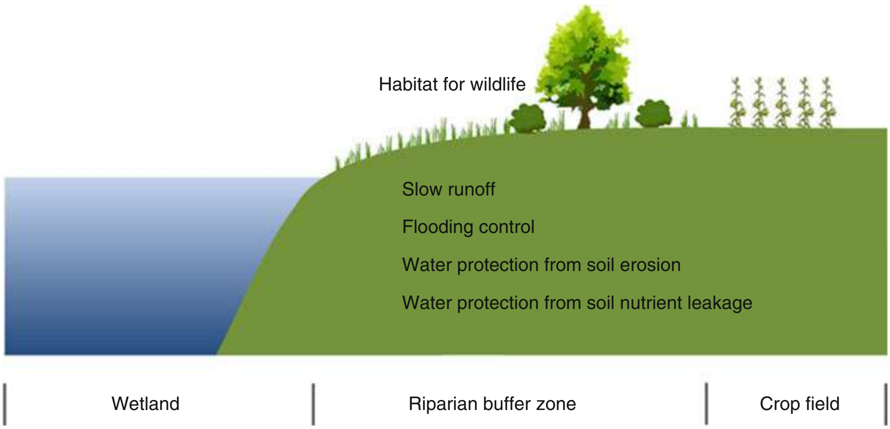


Fig. 3 Habitat in riparian buffer zone and wetland protection

only with the conservation of the wetlands but also its wise use (Ramsar Convention Secretariat 2010).

Wetland protection depends not only on managing wetlands themselves, but also on managing the surrounding water sources and landscape, above all in buffer areas. An abundance of scientific research published since the 1970s documents the value of establishing, maintaining, and enhancing vegetated buffers along wetlands. Wetland buffers provide important benefits including water quality improvement, stream-bank stabilization, flood control, wildlife habitat, and groundwater recharge (Castelle et al. 1992; Correll 1996; Wenger and Fowler 2000; USDA 2003). Wetland buffers also provide significant social and economic benefits by improving aesthetics and increasing property values (Lovell and Sullivan 2005; Qiu et al. 2006). For instance, the excessive nitrate-nitrogen flowing from Midwestern USA is exacerbated by the excessive drainage of the upper Mississippi River Basin that causes waters to leave the fertilized farmland quickly and immediately be transported to the Gulf of Mexico. The control of the nutrient leaching is important in the Gulf of Mexico because the continental shelf fishery in the Gulf is approximately 25% of the USA total. It has been suggested that restored riparian buffers would be necessary to provide enough denitrification to substantially reduce the nitrogen entering the Gulf of Mexico (Mitsch et al. 2001).

Roles and Functions

Water Protection

Wetland buffers improve water quality by trapping and/or transforming pollutants such as sediments, nutrients, pathogens, and pesticides in surface water via biofiltration. The meandering curves along water ways, combined with vegetation and root systems, dissipate stream energy, which results in less soil erosion and a reduction in flood damage. Surface runoff is slowed by buffer vegetation, causing larger sediment particles and pollutants adsorbed to sediment particles to settle out (Lee et al. 2003). This filtering function is greatly enhanced as the density of ground layer vegetation and the width of a buffer increase.

Flooding Control

Wetland buffers that are vegetated with trees and shrubs in floodplains greatly help to slow, store, and gradually release flood waters. Dense vegetation in a wetland buffer increases surface roughness of a floodplain and slows the velocity of overland flow while promoting shallow groundwater recharge, surface storage, and subsequent uptake of water by vegetation (Naiman et al. 2005).

Groundwater Storage

During dry periods, the water in rivers and wetlands may come solely from groundwater which is recharged in upland buffers. Vegetated wetland buffer areas help to slow surface runoff and promote infiltration thereby helping to maintain groundwater levels (Naiman et al. 2005). Elevated groundwater then discharges into streams to provide baseflow. They provide native landscape irrigation by extending seasonal or perennial flows of water.

Habitat for Wild Species

The riparian zones also provide habitats and corridors for wildlife, increase biodiversity, and enabling aquatic and riparian organisms to move along river systems avoiding isolated communities. They can provide forage for wildlife and livestock (Semlitsch and Bodie 2003). In addition, the strong, thick roots of trees and/or shrubs growing along the edge of a stream channel greatly increase the stability of the stream banks and can effectively promote soil biodiversity.

Recreation and Aesthetic

From a social viewpoint, riparian buffer zones provide nearby property values through amenity and views. Wetland buffers offer recreational, aesthetic, economic, and educational opportunities for neighborhoods and schools, promoting healthy lifestyles and enhanced community stewardship and relationships (Qiu et al. 2006). Riparian sports may include walking, running, biking, fishing, swimming, and boating.

Designing Buffer Zones

Width and vegetation composition of buffer zones are key features that enhance many functions essential to establishing and maintaining healthy wetlands.

Width

Width of buffer zone may be the most important factor in affecting buffering functions. Buffer zones are usually not wide enough to function in protecting water, especially in agricultural area. The recommended width of the buffer strip depends on many factors including slope, soil type, farming practices, size of crop fields, and the landowner's objectives. For instance, to remove chemicals and sediment from surface and subsurface runoff, buffer strips should be at least 20 m wide on each side of the waterway. A buffer strip less than 20 m wide does not hold water in the root zone long enough for chemicals to be removed from the water, although it can trap most sediment moving in surface runoff (Semlitsch and Bodie 2003).

However, the weakness of a fixed-width approach is that the buffer widths are not necessarily tailored to the specific conditions of and around the individual wetlands. Variable-width buffer approaches allow buffer area widths to vary according to site-specific or reach-specific conditions such as slope, soil condition, vegetative condition of the stream, or intensity of the existing land use. Typically with these approaches, a minimum buffer width is established that applies to all wetlands and then widths are widened based on site- or reach-specific conditions. The benefit of this approach is that the buffer area can incorporate protection for other sensitive natural features such as floodplains, wildlife habitat, and steep slopes.

Vegetation Types

The most effective riparian buffer zone normally has three zones of vegetation. This combination of trees, shrubs, and grasses helps protect the stream more than planting a single species. Trees and shrubs provide perennial root systems and long-term nutrient storage close to the stream. The warm season grass provides the highest density of stems to slow surface runoff from adjacent fields. The design can be modified to fit the landscape and the landowner's needs, for example, by replacing shrubs with more trees, substituting some of the trees with shrubs, or expanding the grass zone.

Maintaining Buffer Zones

Buffer zones must be monitored and managed to maintain their maximum water quality improvement. Trees and shrubs should be planted in early spring (between March and May, depending on the region) for the best performance of buffering function. Weed control is essential for survival and rapid growth of trees and shrubs in a buffer strip. However, since some types of weeds may provide proper habitats for wild species, maintaining buffer zones shall be also based on main benefits the buffer zones provide.

Conclusions and Future Directions

Scientific studies provide extensive evidence that buffer zone is an important tool for achieving wetland protection. Buffer zone planning shall depend on the individual target for wetland protection. For water quality benefits, buffers need vegetation to slow runoff and for groundwater to pass through and undergo pollutant removal. For flood control, a wide buffer should try to maximize available space in the floodplain so as to increase short-term water storage, and vegetated banks should be present to stabilize stream banks. For habitats, buffers shall be covered by native vegetation of a variety of strata, providing connectivity to other patches, and with minimal disturbances. For overall ecosystem purposes, buffers are most effective when they provide leaf litter to supply the food chain, canopy for shade, large woody debris for in-stream cover, and a place for groundwater recharge which in turn supports baseflow in the stream.

To achieve a sustainable buffering function for wetlands, careful multidisciplinary planning based on ecological knowledge and socioeconomic context is necessary at landscape scale (Wells and Brandon 1993). It is at the landscape scale that we shall continue to understand the relationships between buffer zones and landscape-level management. Increasing landscape heterogeneity can increase size of species pool, which may promote species diversity and vegetation density in buffer zones. A range of socioeconomic policies is needed for farmers to establish, manage, or maintain wider buffer zones and diversified landuse. For instance, the EU-level envrionmental legislation, the agro-environmental scheme, includes the establishment of riparian buffer zone (EU-Commission 2000). In Finland, the national-level agro-environmental scheme includes compulsory measures stating that buffer zones along rivers or lake or sea shores must be at least of 3 m wide, and the regulation also financially encourages farmers to apply on average at least of 15 m wide buffer zone along waters. In the future, a landscape-based approach integrating with policy, economic, and ecological aspects of wetland ecosystem composing of waters, buffers, fields, and other types of landuse is needed.

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Source-Sink Dynamics of Wetlands

22

Tracy A. G. Rittenhouse and William E. Peterman

Contents

Abstract

Source-sink dynamics stem from metapopulation theory, where sources are populations with births exceeding deaths and emigration exceeding immigration, and sinks are populations with deaths exceeding births. Sink populations are sustained by immigration from nearby source populations, and thus functional connectivity among wetlands is key to maintaining source-sink dynamics among wetlands. The wood frog is an example pond-breeding amphibian where source-sink dynamics among wetlands is critical to regional population persistence. We summarize how source-sink dynamics can be inferred from demographics, genet-

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ics, and network models. Challenges that remain include identifying source and sink wetlands within natural systems, as well as, incorporating source-sink dynamics into wetland mitigation.

Keywords
Connectivity · Genetics · Network models · Metapopulations · Mitigation wetlands

Introduction

Definition

Source populations are characterized as having birth rates that exceed death rates and thus an increasing population. This positive growth leads to emigration from the population, and the rate of emigration in source populations exceeds the rate of immigration. As such, sources are net exporters of individuals into the system. In contrast, sink populations are net importers or receivers of dispersing individuals. Sink populations are further characterized as having death rates that exceed birth rates, which leads to population decline. As such, the existence of such sink populations is entirely dependent upon the contributions of dispersers from source populations.

Source-sink dynamics stem from metapopulation theory, which has provided a useful framework to address conservation questions, especially in light of increasing loss or alteration of habitat and fragmentation due to land use that can jeopardize the persistence of species (Hanski and Ovaskainen 2000; Urban et al. 2009). Pulliam (1988) provided the formal description of the demographic source-sink model. Sources are populations in high quality habitat with births exceeding deaths and emigration exceeding immigration. In contrast, sink populations generally exist in low quality habitat, where deaths exceed births, and the populations are sustained by immigration from surrounding source populations. In order for source-sink dynamics to be realized, there must be functional connectivity among habitat patches on the landscape to allow successful dispersal and rescue of sink patches or for colonization of new habitat patches. Functional connectivity is a result of species vagility, distance between habitat patches, and quality of the matrix between patches.

Wetlands provide vital breeding and foraging habitat for many species, but are particularly important for reptiles and amphibians. Pond-breeding amphibians are distributed patchily across the landscape and are frequently assumed to exhibit metapopulation dynamics, especially in fragmented landscapes that limit connectivity (Semlitsch 2008a). Metapopulations of pond breeding amphibians can be viewed from a “ponds-as-patches” perspective (Marsh and Trenham 2001; Richter-Boix et al. 2007). There are several factors that should be considered when characterizing amphibian populations in this manner. First, the pond should be defined to include both the aquatic breeding habitat and the immediate surrounding terrestrial core habitat (95% core area usually within 300 m) where the breeding adult population resides (Rittenhouse and Semlitsch 2007; Semlitsch 1998; Semlitsch and Bodie 2003). Second, several ponds that are in close proximity may function as a unit, and thus several ponds compose the patch or population. This possibility occurs when the amount of amphibian breeding activity at one pond influences the amount at a neighboring pond or when adults shift breeding effort among several adjacent ponds across years (Petranka et al. 2004; Pope et al. 2000). Third, this characterization may not be true in all situations; despite the fact that many amphibians breed in distinct patches, not all amphibian populations exist as metapopulations. For example, if there is extensive movement among all ponds, then these ponds likely constitute a single population (Smith and Green 2005).

Methods for Identifying Source-Sink Dynamics

In the strictest sense, source-sink dynamics requires estimation of birth, death, immigration, and emigration rates. Estimation of these rates presents logistical challenges. Estimates of population growth rates often require detailed, long-term population monitoring (e.g., mark-recapture, nest survival). Such approaches can be both time and cost prohibitive, making robust estimates of these parameters unfeasible in many circumstances. There can also be significant difficulty in identifying movement among populations as this often involves direct monitoring of individuals in space and time. These observations have traditionally been made using mark-recapture or radio telemetry (Gamble et al. 2006, 2007; Rittenhouse et al. 2009), which has generally limited the scope of inference to a few select populations. The increased accessibility of molecular methods is now allowing for direct quantification of movement among numerous populations. These genetic estimates represent a more complete picture of population connectedness as well as provide an estimate of realized dispersal resulting in successful reproduction (Wang et al. 2009).

Other approaches to assess population connectivity are based on resistance kernels (Compton et al. 2007; Wasserman et al. 2010), electric circuit theory (McRae et al. 2008), or graph theory (Urban et al. 2009). Each of these methods can assess the potential connectivity among populations as a function of distance and intervening habitat. When paired with demographic models of population size, reproduction rates, and dispersal kernels (Schick and Lindley 2007), graph-theoretic models can be used to identify source and sink populations on the landscape. Although potentially less informed and more reliant on model assumptions, these network models are less data intensive than empirical observations of movement, and less expensive than genetic approaches. Further, these models can provide a robust framework for assessing connectivity, directionality of movement among populations, as well as management effects on the network as a whole (e.g., effects of wetland loss or mitigation on connectivity and movement; Schick and Lindley

2007). Recently, Murphy et al. (2010) combined network-based gravity models with spatial population genetic data to infer functional connectivity.

Using Genetics to Infer Source-Sink Dynamics

Genetic estimates of connectivity may be simpler to obtain than direct observation of dispersing organisms, and they provide a means to assess population connectivity across generations. One potential limitation of these estimates is that they can reflect historical processes and may not be representative of current dispersal patterns (Pearse and Crandall 2004). Nonetheless, genetic data can allow for the estimation of parentage, effective population sizes, dispersal rates, and connectivity among populations, all of which can be used to infer source-sink dynamics (Manier and Arnold 2005; Martínez-Solano and González 2008; Peery et al. 2008). In a study of garter snakes (*Thamnophis elegans* and *T. sirtalis*) in California, Manier and Arnold (2005) found gene flow (i.e., dispersal) was asymmetric among the 23 populations studied, and that a few populations contributed a disproportionate number of dispersers to the metapopulation. Further, they found that genetic measures of effective population size and migration rates were often significantly different from those obtained through direct observation via mark-recapture. These findings underscore the potential power of genetic approaches and the difficulty in obtaining direct estimates of population vital rates. In another study, Martínez-Solano and González (2008) found that recently established populations of common toads (*Bufo bufo*) showed both genetic and demographic signatures of sink populations. Specifically, they had lower effective population sizes, had a greater of rate of immigration, and had greater frequency of heterozygotes than expected (heterozyote excess).

Network Models

Wood frogs (*Rana sylvatica*) are a widely distributed pond-breeding amphibian in North America. Extensive demographic and movement studies have been conducted on wood frogs (Berven 1990; Berven and Grudzien 1990; Rittenhouse et al. 2009), making them an ideal species for parameterizing demographic network models (Schick and Lindley 2007). Using 6 years of egg mass counts from 41 ponds in east-central Missouri, a demographic network model was parameterized to estimate: (1) number of metamorphic individuals produced at each pond, (2) the number of metamorph frogs that would reach maturity and return to their natal pond for reproduction, (3) the number of metamorph frogs that would reach maturity and successfully disperse to a new breeding pond, and (4) the ratio of immigrant to emigrant frogs at each pond (ratio >1 = source population, ratio <1 = sink population; Peterman et al. 2013). Parameters for survival, rates of dispersal, and dispersal distance were obtained from empirical estimates found in the literature (Berven 1990; Berven and Grudzien 1990). The average of the 6 years of observations is shown in Fig. 1. Being an average, this figure encompasses annual variation

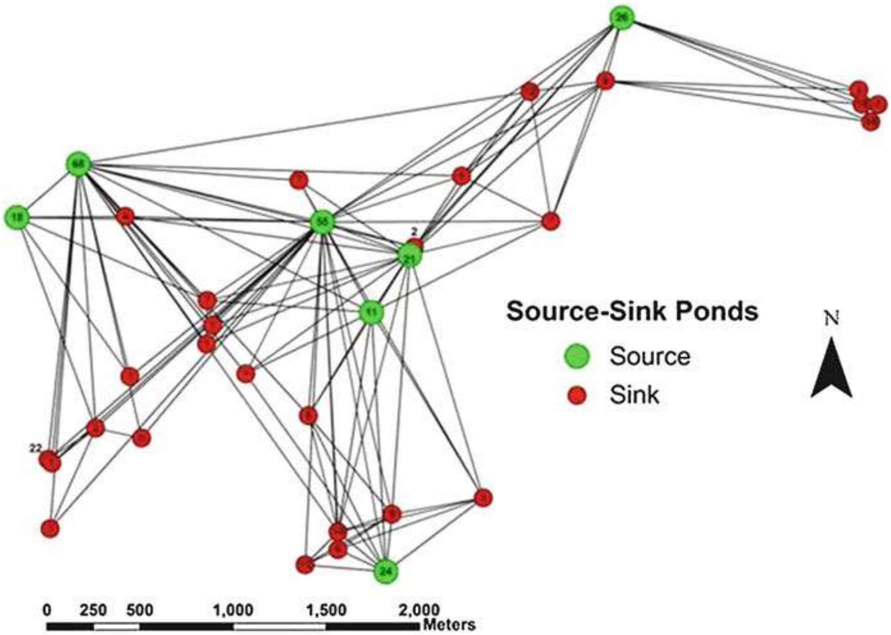

Fig. 1 A demographic network model of wood frog populations in east-central Missouri, USA. Each point represents the spatial location of a pond, and the numbers on each point represent the mean number of egg masses (equivalent to the number of breeding females) present at ponds from 2006 to 2010. Connectivity of ponds was determined as a function of distance and the number of potential dispersing juvenile frogs. Ponds with emigration rates greater than immigration rates were designated as sources

in reproductive success, which largely corresponded to variation in precipitation. This modeling approach, although simplistic and with many inherent assumptions, allows for spatial and temporal assessment of source-sink dynamics. Analyses similar to these could help inform future wetland mitigation to provide optimal placement of wetlands in relation to existing populations. Alternatively, if wetlands were to be lost or altered, these models could be used to minimize the damages to the connectivity of the metapopulation.

Future Directions and Challenges

As a specific case of meta-population dynamics, source-sink dynamics may be particularly relevant in the study of wetlands, which are often viewed as habitat patches in the context of the species using them (Marsh and Trenham 2001). Future management of wetlands should take into consideration connectivity among populations, but also recognize that all populations are not equal. By identifying source and sink populations within the metapopulation, ranked conservation or management priority can be given to assure that the most robust populations are conserved. For example the number of individuals that complete metamorphosis and successfully emigrate from a pond is an important factor to consider when developing management plans (Semlitsch 2000). Alternatively, isolated populations or perennial sink populations can be selectively targeted for conservation or restoration actions. Restoration of a wetland that is consistently a sink population (e.g., improper hydrology or presence of fish) has the potential to strengthen regional population persistence by increasing the number of source populations. If low connectivity due to low dispersal success is a concern, then restoration efforts may include the creation of new wetlands to serve as stepping stones (Petranka et al. 2007; Petranka and Holbrook 2006). An additional challenge to estimating connectivity using resistance kernels, circuit theory, or graph theory is determining the effects that land cover and landscape features have on dispersing individuals. Current assessments often rely on expert opinion, but this approach should be used with caution as Charney (2012) demonstrated that expert models performed poorly.

Source-sink dynamics are important processes to consider within the context of wetland mitigation, and a future challenge for wetland managers will be to incorporate source-sink dynamics into wetland mitigation (Keagy et al. 2005). Filling one wetland and creating a new wetland in an alternative location may result in an equal number of wetland acres, but functions are not often realized in mitigated wetlands (Semlitsch 2008b). If source populations are continually destroyed and mitigated with sink populations, species persistence on the landscape will be greatly affected. A pond's role in the metapopulation can be dependent on within ponds features such as hydrology and vegetation that affect productivity (Shulse et al. 2012), as well as placement on the landscape, which affects connectivity with other ponds and can influence the species community that colonize mitigated ponds (Shulse et al. 2010). Wetland mitigation will be more effective when taking into consideration demographic and life history characteristics of species.

A major challenge in conducting source-sink analyses has been, and will continue to be, the estimation of the necessary population parameters. Although technology for directly tracking individuals continues to improve, these methods are likely to remain costly and labor intensive. In contrast, genetic studies for non-model species are continuing to become more affordable. Genetic methods have limitations, but their use in studies of source-sink dynamics has the potential to provide the most complete picture of cryptic movement among populations. At the very least, genetic analyses can provide a starting point for identifying which populations warrant more intensive empirical studies of vital rates.

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Wetland Restoration

23

Joy B. Zedler and Nick Miller

Contents

Abstract

Wetland landscapes and wetland watersheds are both general, flexible terms that are often used interchangeably; however, the boundaries of landscapes are less discrete than those of watersheds, which have specific drainage areas. In expansive flat topography, as in Florida and the US Upper Midwestern prairie pothole region, it is difficult to locate watershed boundaries, so the term landscape is more appropriate. Elsewhere, wetland catchments are easily defined, and it is useful to refer to watersheds whenever emphasizing the inseparable interactions between the water that flows downslope and collects to form wetlands. Here, we consider watersheds to be parts of larger landscapes, which we also call regions to emphasize their large spatial scale. Most of our examples are drawn from the USA, where we have the greatest experience.

Keywords

Ecosystem services · Landscape · Restoration · Watershed

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Introduction

Wetland landscapes and wetland watersheds are both general, flexible terms that are often used interchangeably; however, the boundaries of landscapes are less discrete than those of watersheds, which have specific drainage areas. In expansive flat topography, as in Florida and the US Upper Midwestern prairie pothole region, it is difficult to locate watershed boundaries, so the term landscape is more appropriate. Elsewhere, wetland catchments are easily defined, and it is useful to refer to watersheds whenever emphasizing the inseparable interactions between the water that flows downslope and collects to form wetlands. Here, we consider watersheds to be parts of larger landscapes, which we also call regions to emphasize their large spatial scale. Most of our examples are drawn from USA, where we have the greatest experience.

Landscape-Scale Changes in Wetland Area

Extensive drainage, water diversions, and groundwater depletion eliminated over 100 million acres of wetland (53%) in the 48 contiguous states by the 1980s (Dahl 1990). In several states (e.g., California and the Upper Midwestern “corn belt”), over 80% of historical wetland area succumbed to drainage and oxidation of organic soils or to filling and development (Dahl 1990). Aldo Leopold (1949) vividly described the example of Central Wisconsin during the 1930s Dust Bowl years: “the marsh was gridironed with drainage canals, speckled with new fields and farmsteads. But crops were poor and beset by frost, to which the expensive ditches added an aftermath of debt. Farmers moved out. Peat beds dried, shrank, caught fire. . ..After a dry summer not even the winter snows could extinguish the smoldering marsh. . ..For a decade or two crops grew poorer, fires deeper, wood-fields larger, and cranes scarcer, year by year.” The historical wetland landscape was altered in irreversible ways.

Reductions in wetland area and the near-permanent loss of underlying peat create significant challenges for landscape-scale restoration, and most areas drained for agriculture have remained cultivated. Large projects have, however, been initiated to restore Kankakee Wetlands (Indiana), the San Francisco Bay Delta (California), the Everglades (Florida), and the Gulf of Mexico (notably in Louisiana). USA’s July 2012 RESTORE Act allocates billions of dollars in oil-spill penalties to BP toward restoration of Gulf Coast wetlands. Funds can support up to 109 projects included in Louisiana’s 50-year, $50 billion Coastal Master Plan. In addition, 80% of the Clean Water Act penalties will help implement a comprehensive plan for the Gulf Coast focused on ecosystem and coastal restoration, and 5% will support Gulf Coast research, science, and technology, including fisheries management, ecosystem monitoring, and a Gulf Coast Center of Excellence in each impacted state.

Regions and watersheds have individual “wetland profiles,” as revealed by large-scale analyses of wetland area and wetland resources. For example, Danz et al. (2007) characterized wetlands along the coasts (U.S. sides) of all five Great Lakes, based on 86 human-influenced stressors for each of 762 watershed-based units. Not surprisingly, the greatest environmental stress had accumulated in watersheds near centers of intensive land use and urbanization, namely, western L. Michigan, southwestern L. Erie, and southeastern L. Ontario. Subsequent analysis of biological responses indicated strong responses in water chemistry and fish, bird, and plant communities (Danz et al. 2007, Johnston et al. 2009). Among the vegetation responses, Frieswyk et al. (2008) demonstrated dominance by invasive cattails (*Typha angustifolia* and *T. x glauca*) at three spatial scales: individual wetlands, whole lakes, and the entire Great Lakes region (i.e., USA coasts). Wetland degradation was thus both direct (altered abiotic conditions) and indirect (invasions and lost species).

Wetland restoration efforts have typically focused on returning the amount of area lost and on efforts to establish native vegetation and attract native wildlife, without sufficient attention to wetland type or functions. The restoration of ecosystem services is now gaining prominence in restoration planning and implementation. Ecosystem services (functions that are valued by people) are often included in restoration goals, especially water quality improvement, flood abatement, and carbon storage (i.e., long-term sequestration).

Landscape-Scale Changes in Wetland Services

The Millenium Ecosystem Assessment (MEA 2005) classified ecosystem services as supporting (productivity, nutrient cycling), provisioning (e.g., timber production, gene pools that give rise to new foods and medicines), regulating (e.g., infiltrating runoff, abating flooding, improving water quality, storing carbon in peat), and cultural (esthetics, education, research, recreation). Within that assessment, biodiversity support was treated as encompassing all services, i.e., plants, animal, and micro-organisms (fungi, bacteria) were regarded as the key providers of wetland ecosystem services.

Even though wetlands typically comprise less than a quarter of a watershed's area, they provide disproportionately greater services (cf. data in Costanza et al. 1997). Losses in ecosystem services from the contiguous USA declined by approximately one-half after that mapped in the 1780s, but the associated loss of ecosystem services was not quantified until recently. Yet we receive occasional reminders of the historical services provided by large areas of wetland, such as flood abatement. The catastrophic flooding of the Mississippi River in 1993–1994 killed 50 people, wiped out levees, forced thousands of people to evacuate their homes for months, and caused $15 billion in damages; "The flood was unusual in the magnitude of the crests, the number of record crests, the large area impacted, and the length of the time the flood was an issue" (Larson 1996). Much was blamed on historical wetland loss; the restoration of just 3% of the floodplain may be able to prevent future catastrophic floods (Hey and Philippi 1995). The capacity of landscapes to abate flooding, clean the water, store carbon, and support biodiversity has clearly diminished as wetland area has declined, even though direct quantification of specific services is lacking. In fact, even today, historical losses and future gains are determined by hydrogeomorphic modeling (as in Miller et al. 2012), rather than direct measurement.

Attempts to reverse the degradation of landscapes by restoring wetlands to improve water quality, abate flooding, and sequester carbon requires careful attention to the location and type of wetlands to be restored and their basic hydrological conditions. The quantity of water available for restoration can either be too low (where water has been diverted or overconsumed) or too high (from runoff that fails to infiltrate upstream due to compacted soils and hardscapes). Watersheds produce excess runoff if they have undergone intensive land use for urbanization (and hard surfaces designed to shed, rather than infiltrate, water) and agriculture (which reduces soil structure and permeability and hence increase water discharge). In turn, excess water carries excess nutrients and other contaminants that further challenge wetland restoration at landscape scales. Nevertheless, Sweden embarked on an ambitious program of restoring 12,000 ha of wetlands (mostly shallow-water wetlands and ponds) within a large agricultural region. Large challenges catalyze landscape-scale restoration plans.

A surge in the construction of ponds between 1998 and 2004 in the conterminous USA (700,000 acres; Dahl 2006) led to a net gain in wetland area but did not compensate for continued losses of coastal and forested wetlands in their naturally occurring landscape positions. A pond cannot fully replace lost biodiversity or wetland ecosystem services performed by fens, bogs, marshes, meadows, and swamps. Given the serious consequences that follow large-scale loss and degradation of wetlands, the need for wetland restoration has never been greater. To regain lost services, NRC (2001) recommended restoring former wetlands, rather than creating wetlands, e.g., excavating ponds from uplands. NRC (2001) further proposed that restoration be done within a watershed context and following a strategic plan.

Watershed Planning for Strategic Wetland Restoration

Wetland restoration that is strategic at the watershed scale pays attention to location and wetland type (NRC 2001; Zedler 2003). For example, to compensate for wetland loss throughout the watershed, it is preferable to reduce contamination at the source (prevention) and filter and treat the water immediately downstream (via wetland restoration) than to rely solely on retention ponds at the bottom of the watershed (Fig. 1). Downstream retention does not compensate for a lack of preventive measures upstream. Landscape position and wetland type helped explain sixfold differences in the ability of hundreds of wetlands to remove nitrogen; those constructed in Sweden outperformed those in nearby Denmark (Ayala 1996). Large shallow-water wetlands with complex, vegetated shorelines removed more nitrogen and supported more wildlife than small deep ponds (Hansson et al. 2005).

Wetlands can restore water quality by removing sediments, nutrients, toxic contaminants, and pathogens, but not all the unwanted materials in water can be removed quickly or easily. Nitrogen (N) removal by bacteria, for example, requires the water to have anaerobic conditions in close proximity to aerobic conditions (as in shallow-water wetlands); then, denitrifying bacteria can convert ammonium to nitrates to harmless nitrogen gas. Given the extent of N-caused coastal eutrophication and large dead zones at the mouths of major rivers, it would seem that human

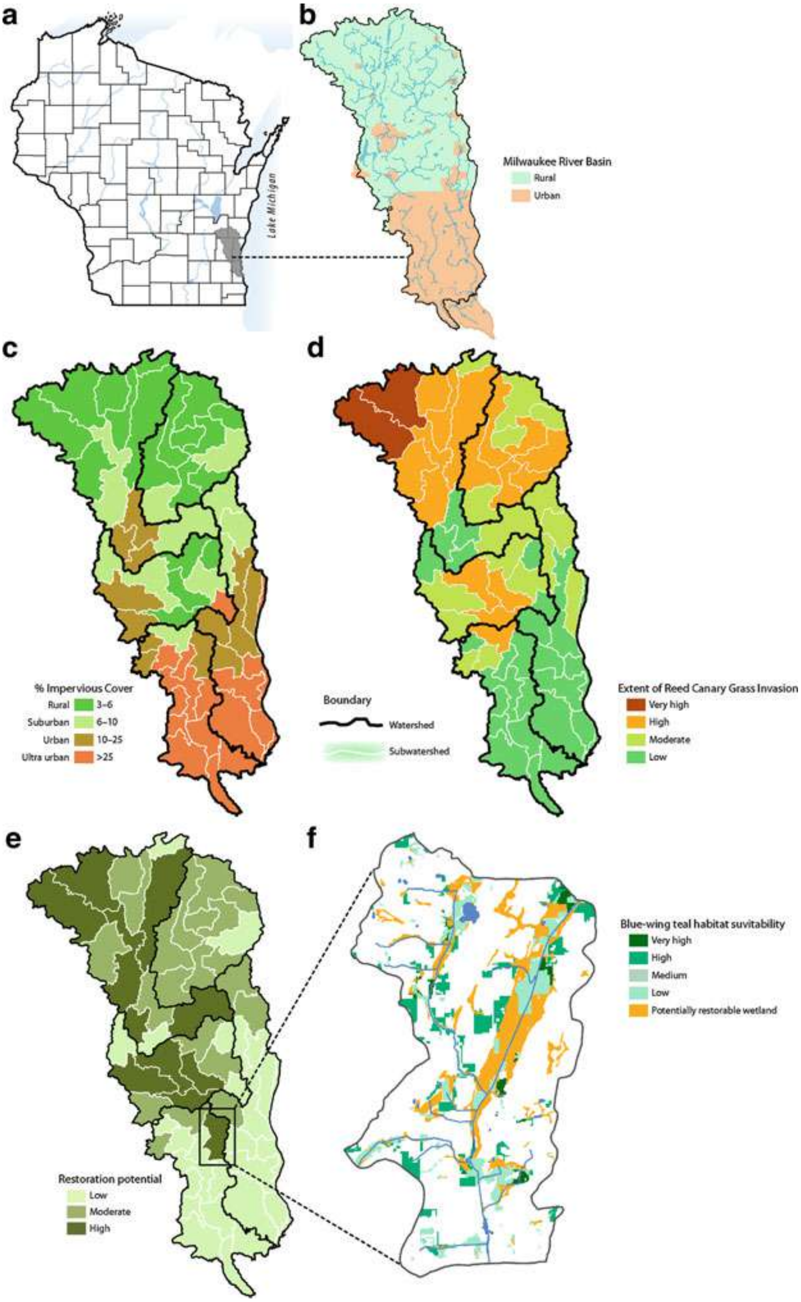

Fig. 1 Several attributes of the Milwaukee River Basin have been mapped by the Wisconsin Department of Natural Resources for use in watershed-scale planning for wetland restoration. The basin is located in eastern Wisconsin (1). The first basin-scale map (2) shows streams and municipalities. The percent area of impervious surfaces and classes of reed-canary-grass cover within the five 10-digit hydrological unit (HU) watersheds (2a) fed into an estimate of wetland

efforts to improve water quality would focus on N removal. Instead, efforts focus on phosphorus (P) reduction to reduce eutrophication in inland lakes, in part because P is more easily removed (Lewis and Wurtsbaugh 2008). Some P adheres to sediment particles and settles when water flows into deep ponds. P-removal is much simpler than N-removal; the latter relies on bacteria to denitrify large quantities of nitrates in shallow water where plant roots leak both organic matter (to fuel bacteria) and oxygen (for nitrification, the precursor to denitrification) (Paerl 2009).

Because P often enhances the growth of phytoplankton in lakes, the demands of swimmers, boaters, and fishers for water of high redundant quality have led to a focus on P removal. While retention ponds can accumulate P-rich sediment, and while the sediment can be excavated later and transported off site, such ponds do not make dirty water clean. Such ponds allow particulate contaminants to settle out during low-flow events (less so during major stormwater pulses). Because TSS and P removal are easily simulated in mathematical models, stormwater treatment systems are designed and built to meet specific sediment-removal standards. However, much of the N and dissolved contaminants flow through such systems. Because the actual performance of stormwater management facilities is rarely monitored, the shortcomings of retention ponds are poorly quantified (Pataki et al. 2011).

New landscape-scale tools, namely watershed plans, are becoming available to help decision-makers regain some of the lost services and to manage remaining resources (ELI and TNC 2014). Existing tools and plans span a spectrum from decision-making frameworks (e.g., Hruby et al. 2009) to prescribed outcomes for specific watersheds. Watershed plans can be tailored to regional needs and "allocate lands to their most suitable uses" (ASWM 2001). Because landscapes range from very dry (<0.3% of NV is wetland) to unusually wet (29.5% of FL is wetland) (Dahl 1990), the prioritization of types and locations of wetland restoration projects addresses varied challenges (Table 1).

Regulators (CoE 2008) adopted science-based advice to use a watershed approach in deciding where and how to restore wetlands to compensate for permitted losses (NRC 2001), except that wetland mitigation need not follow a watershed plan where there is none (Gardner et al. 2009). To implement watershed planning at the national scale would require funding to develop plans for model watersheds in all regions and coordinated efforts among agencies to fund wetland restoration, e.g., Fish and Wildlife Service, Natural Resources Conservation

Fig. 1 (continued) restoration "need" (= lost wetland/remaining wetland * % historical wetland). Opportunities for wetland restoration (3) were indicated as the relative area of potentially restorable wetlands within 12 digit HU subwatersheds. This map highlights the Little Menominee Subwatershed, where WDNR would like to increase Blue-wing Teal by restoring marsh and open water wetlands within a grassland matrix adjacent to the existing suitable habitat. Map 4 allows the user to visually evaluate where potential wetland restoration projects can best add to existing higher quality habitat (Maps courtesy of WDNR–Kate Barrett, Thomas Bernthal, Marsha Burzynski, Gary Casper, Joanne Kline, and Jeff Strobel)

Table 1 Examples of landscape-scale wetland restoration efforts

Priority ecosystem service	Large restoration program
Wetland diversity, especially endangered species	Kissimmee River (removing a flood control channel to restore the floodplain) and the Everglades (reducing eutrophication to reduce cattail invasions) (Toth 2010)
	Kankakee Sands, Indiana (>2,800 ha of hydrological modifications to restore plant communities) (http://www.nature.org/ourinitiatives/regions/northamerica/unitedstates/indiana/placesweprotect/kankakee-sands-project-office.xml)
	Southern CA coastal wetlands (restoring and sustaining tidal influence in multiple lagoons and salt marshes) (http://www.scwrp.org/)
Water quality in dead zones	Mississippi River (abating nitrogen in agricultural runoff in Iowa to protect the Gulf of Mexico) (Crumpton 2001)
	Restoring and creating 12,000 ha of wetlands in southwestern (agricultural) Sweden (Ayala 1996)
Environmental flows	San Francisco Bay Delta (less diversion of water to restore inflows, lower salinity, and sustain fish) (http://calwater.ca.gov/)
	Colorado River (restore flood pulses by managing the Glen Canyon Dam to restore multiple ecosystem services) (Lovich and Melis 2007)
Livelihoods based on fish and use of *Phragmites*	Mesopotamian Marshlands (restore water that was drained to exterminate Marsh Arabs). Large areas have been rewetted; *Phragmites* is once again widespread; rare teal are becoming abundant; Marsh Arabs are returning; fishing has resumed. Contaminated water and uncertain water supplies remain concerns (Alwash In press 2013; http://pdf.usaid.gov/pdf_docs/PDACF082.pdf)

Service, and many not-for-profit organizations. In fulfilling a landscape approach, decision-makers would include planning for all ecosystems, not just wetlands. To allow swapping of one type of wetland and its set of services for another, such as a pond, would require much better data on the values attributable to each wetland service. A further improvement would be the addition of economic profiles for regional wetlands including information on the specific ecosystem services provided by each wetland type. At present, economic valuations are inadequate (Aronson et al. 2010).

A Model Watershed Plan

The Duck-Pensaukee Watershed Approach (Miller et al. 2012) was developed for a 128,909 ha Great Lakes coastal watershed (8-digit US Geological Services hydrologic unit code [HUC]) by The Nature Conservancy in collaboration with Environmental Law Institute, Wisconsin DNR, St. Paul District of the Corps, and EPA

Region 5, tribes, counties, municipalities, conservation organizations, and academic researchers. The approach falls in the middle of the watershed planning spectrum (ELI and TNC 2014), as it evaluates and ranks conservation opportunities across the watershed but does not prescribe specific actions or outcomes. Instead, it enables users to match individual or collective restoration objectives (e.g., flood abatement, wildlife habitat, etc.) to subwatersheds and specific sites with the greatest likelihood of meeting those objectives. Unique features of the plan include: (1) a watershed profile to identify high priority subwatersheds, based on historical losses in services; (2) identification of specific sites with high potential to provide an array of ecosystem services; (3) a focus on wetland-upland connections important to wildlife; and (4) ability to tailor state Wildlife Action Plan priorities to a watershed scale and make them spatially explicit.

Users of the approach are first guided toward specific subwatersheds that have experienced the greatest decline in each of four ecosystem services: water quality improvement, flood abatement, surface water supply, and carbon storage. Declines in these services since the 1800s were determined by adapting the methods of the National Wetland Inventory-Plus (Tiner 2005), which correlate site characteristics (e.g., cover type, hydrogeomorphic features) with ecosystem services, for both historical and current wetland extent and condition. An index of change was then created for each of fifteen 12-digit HUCs (averaging 8,594 ha), enabling users to apply restoration efforts toward the subwatersheds with the greatest likelihood of returning ecosystem services.

After subwatersheds have been selected, individual sites can be chosen based on the relative potential to restore ecosystem services: water quality improvement, flood abatement, surface water supply, shoreline protection, carbon storage, and fish and wildlife habitat provision. Each potentially restorable wetland has been ranked for its ability to support a suite of priority and representative wildlife using the Wildlife Tool (Kline et al. 2006), which creates maps of probable habitat via expert-informed modeling. Sites have been further ranked for their potential to provide the remaining services by adapting the Wetland Evaluation Technique (WET) (Adamus et al. 1991), which calculates the probability of service provision based on a suite of criteria in three categories: opportunity, effectiveness, and social significance. For the example of water quality improvement, *opportunity* criteria included catchment attributes, such as impervious surfaces, land use, and nutrient sources. *Effectiveness* criteria included ecosystem characteristics of the site, such as persistent vegetation in depressions that could slow floodwaters and settle solids. *Social significance* criteria focused on the site's ability to benefit society (Figs. 2 and 3).

The Duck-Pensaukee Watershed Approach is a pilot effort designed to be applied in other watersheds, given the availability of data and technical personnel. Once components of this plan are implemented, the monitoring of outcomes, such as water quality at strategic locations, will serve both science and practice. Additional knowledge would accrue if subwatersheds were paired (with and without restoration efforts) or if several subwatersheds with varying amounts of restored wetland were compared as direct tests of the effectiveness of restoration at the landscape scale.

Fig. 2 A floodplain of the Pecatonica River in southwest Wisconsin, restored by the Wisconsin Department of Natural Resources and The Nature Conservancy. The stream segment was cleared of brush and sediments that had eroded from agricultural fields (**a**), with rich topsoil stockpiled on site and removed voluntarily when advertised as free for the taking. The creek bed was left intact, intended for fish use, and bare banks were stabilized using mulch-matting (**b**). A cover crop and native species were sown on the new floodplain, and woody debris was added to enhance fish habitat (**c**). Additional restoration took place upstream when the site and funding became available (October 2006 photos by J. Zedler)

Future Challenges

Now that watershed planning is underway, the challenge is to implement wetland restoration in ways that improve both the science and practice of wetland restoration. A landscape designed to maximize wetland services (e.g., support biodiversity, abate flooding, sustain clean water, and store carbon) would have at least these components:

1. Remnant wetlands left intact, buffered, and protected from damaging inflows (although "dry deposition" of airborne N and dust would still occur).
2. Wetlands restored in strategic locations to provide the above services.
3. Sufficient habitat patches with sufficient connectivity to support dispersal of plants and animals across landscapes as they respond to climate changes (cf. Vos et al. 2010).

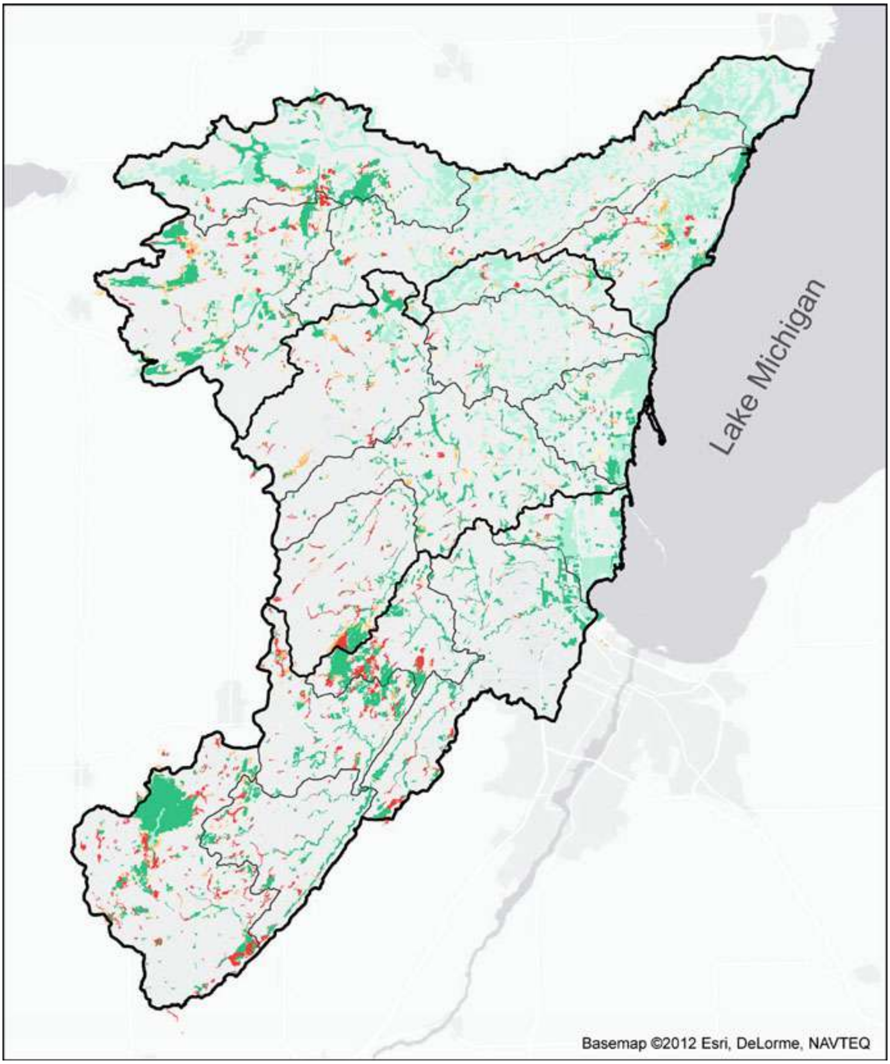

Fig. 3 Relative potential for wetland sites to improve water quality in the Duck-Pensaukee Watershed, Wisconsin, based on an assessment of watershed position, land-use context, and site characteristics. Darker shades indicate greater potential. Potential restoration sites (former wetlands) are indicated in *red* and existing wetlands are indicated in *green*

4. A populace dedicated to minimizing the discharge of water and contaminants, i.e., increasing infiltration (porous asphalt driveways, rain gardens, rain barrels), minimizing the use of pesticides and fertilizers, and practicing no-till agriculture and crop rotation (with N-fixing legumes) wherever possible. Reducing runoff and minimizing the contaminant loads upstream would reduce the workload for downstream wetlands.

5. Paired subwatersheds that compare landscapes with and without restored wetlands.
6. Broad upland buffers (free of cultivation, grazing and urbanization) would surround wetlands and reduce contamination.
7. Where wetland protection and restoration are no longer able to provide necessary ecosystem services, stormwater management facilities (constructed from uplands) would collect runoff from agricultural fields and urban developments, use native vegetation to stabilize exposed sediment, and remove both N and P, as well as other contaminants.

Wetland restoration is poised to improve ecosystem services at both site and watershed scales. Recent comparisons of watersheds in North Carolina indicate large discharges of N and P with only small losses of wetland area (Flanagan and Richardson 2010). It follows that small gains in wetland area could measurably improve nutrient removal, especially if projects are located strategically and implemented skillfully, following a watershed plan. At the same time, services that are compatible with nutrient removal would be augmented; potential examples are flood abatement (via slowing water flow and enhancing infiltration) and carbon storage (via burial of organic matter under sediments).

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Wetland Heterogeneity

24

Daniel J. Larkin

Contents

Abstract

Heterogeneity is a key feature of wetland ecosystems, which vary physically, chemically, and biologically over space and time. Today, longstanding interest in how heterogeneity influences fundamental ecological processes is coupled with concern that anthropogenic changes are reducing the heterogeneity of wetland and other ecosystems. Spatial heterogeneity is apparent in wetlands at landscape, habitat, and micro scales. Variation can be gradual along environmental gradients or abrupt; even relatively small changes in wetlands can exert large influences through interactions with hydrology. Temporal heterogeneity is an important feature of wetlands, from the decadal or longer scales associated with vegetation succession to diurnal variability in water levels. Origins, effects, and applications of heterogeneity to wetland management are discussed, with a focus on the spatial dimension of wetland heterogeneity at the habitat scale.

Keywords

Hydrology · Landscape · Restoration · Spatial · Temporal · Topography

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Introduction

Heterogeneity is a broad concept used to describe one of the key features of wetland ecosystems: their variability in structure and function over space and/or time. Ecologists have a longstanding interest in understanding how spatial variability influences the distribution of organisms, interactions among species, and community change through time. Today, this interest is coupled with widespread concern that anthropogenic global change is reducing the heterogeneity (i.e., the ecological distinctiveness) of the planet's ecosystems (McKinney and Lockwood 1999).

Wetland heterogeneity occurs across multiple dimensions and scales. At the landscape scale, there is heterogeneity in the types of wetland habitats that occur in a region (Fig. 1a). For example, a northern-temperate landscape may be dotted with marsh, sedge-meadow, fen, shrub-carr, and bog systems that vary in size, density, and proximity. At the habitat scale, wetlands are heterogeneous over two-dimensional space, e.g., in the interspersion of vegetation with open water, bare ground, and other cover types (Fig. 1b). This change can be gradual along environmental gradients (Squires and van der Valk 1992) or abrupt, as when interactions between organisms and their environments create sharp boundaries (Pennings and Callaway 1992). A third spatial dimension is vertical heterogeneity.

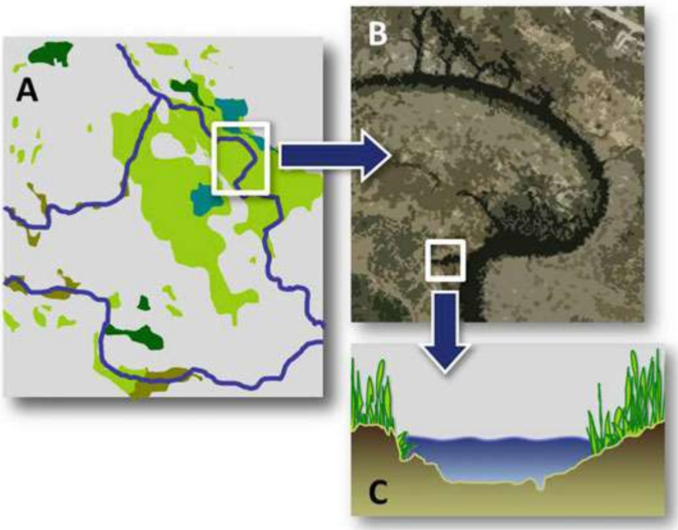

Fig. 1 Spatial heterogeneity influences wetland structure and function at multiple scales. (**a**) Diversity of wetland types (different-colored patches) imparts landscape-scale heterogeneity. (**b**) Within a wetland, changes in plant-community composition and water-vegetation interspersion are key components of habitat heterogeneity. (**c**) Topographic heterogeneity, depicted in this cross-section of a creek channel, provides distinct microenvironments

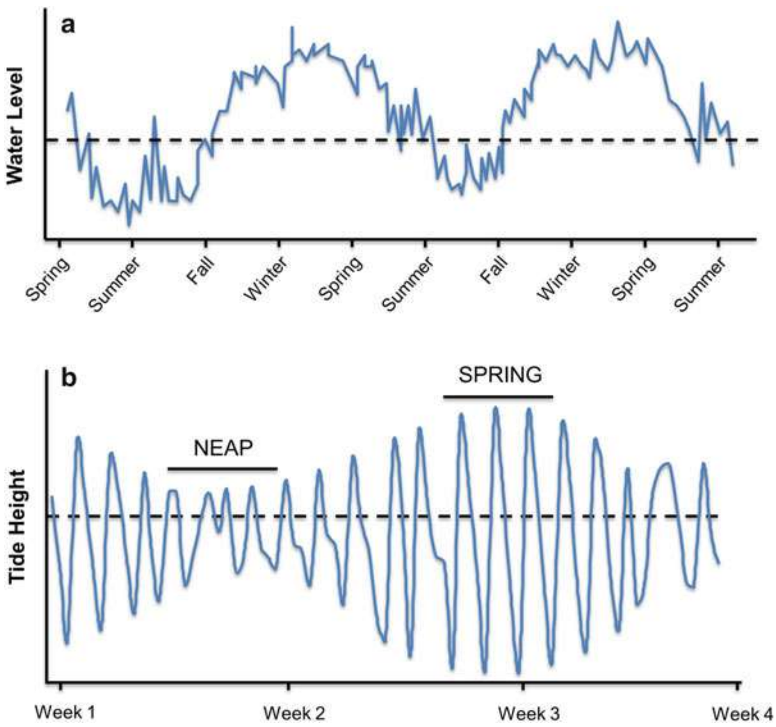

Fig. 2 Wetlands are strongly influenced by temporal heterogeneity. (**a**) In this hypothetical hydrograph of an inland, temperate wetland, the growing season spans from spring flooding to a mid-summer drawdown to fall flooding (elevation of the marsh surface shown with a dashed line). (**b**) Inundation of tidal wetlands changes frequently and predictably. Water depth varies on diurnal (high and low tides), lunar (spring and neap tides), and seasonal cycles (periods of overall higher vs. lower high tides)

Although wetlands lack the dramatic elevational changes of many terrestrial ecosystems, subtle differences in topography can be important features. Through mediation of wetland hydrology and edaphic conditions, such topographic heterogeneity (Fig. 1c) can greatly influence wetland structure and function (Larkin et al. 2006). Change over time is a fourth dimension of heterogeneity. Temporal heterogeneity includes the multidecadal scales associated with vegetation succession; interannual changes in weather and hydrology; seasonal differences in temperature, precipitation, and biotic activity; and diurnal variability in water levels and tidal inundation (Fig. 2).

The following sections introduce aspects of the origins, effects, and research and management challenges associated with wetland heterogeneity, with a focus on the spatial dimension of wetland heterogeneity at the habitat scale.

Sources of Heterogeneity

Spatial heterogeneity arises from geomorphological, hydrological, and biotic processes – and their interactions – and is a defining feature of certain wetland types. In coastal wetlands, tidal flows alter patterns of sediment distribution, creating complex tidal-creek networks (Williams et al. 2002). Movement of water also strongly influences the structure of riverine wetlands (Brinson 1993). Hummocks formed by *Sphagnum* mosses in peatlands can impede perpendicular water flow, yielding "flark-and-string" patterning over large spatial scales (Swanson and Grigal 1988). In salt marshes, waterlogging and salinity stress plants, altering the balance between competition and facilitation in ways that can cause stark zonation of dominant vegetation (Bertness 1991; Pennings and Callaway 1992). The sedge *Carex stricta* engineers ecosystems; its growth form produces complexes of elevated tussocks hosting numerous other plant species, interspersed with lower, often unvegetated inter-tussock spaces (Werner and Zedler 2002; Crain and Bertness 2005).

Effects of Heterogeneity

As in other ecosystem types, habitat heterogeneity of wetlands exerts strong effects on environmental/community structure and ecosystem functioning. In both natural and experimental systems, heterogeneity in wetland topography/hydroperiod has been shown to increase plant diversity (Vivian-Smith 1997). In a study of isolated wetlands in Germany, hydroperiod heterogeneity explained plant diversity better than the classic habitat-area relationship (Brose 2001). In Atlantic USA coastal wetlands, declines in macroinvertebrate diversity and fish-support functions accompanied the loss of marsh-surface heterogeneity that followed *Phragmites australis* (common reed) invasion (Angradi et al. 2001; Able et al. 2003). Abundance of several secretive marsh bird species (American Bittern, Least Bittern, Sora Rail, and Virginia Rail) was best predicted by interspersion of water and vegetation in a New York study, indicating that managing for complex spatial patterning could be beneficial for wetland-dependent birds (Rehm and Baldassarre 2007).

In addition to these effects on plant and animal communities, heterogeneity can also influence ecosystem functioning of wetlands. Wetland functions include valuable ecosystem services, such as carbon sequestration and denitrification. These and other biogeochemical processes in wetlands exhibit "hot spots and hot moments" (i.e., unusually high reaction rates in discrete patches or periods of time) (McClain et al. 2003) driven by spatial and temporal heterogeneity. Consideration of how hydrology, topography, and substrate and nutrient availability interact to produce heterogeneity in ecosystem processes can improve predictive models and facilitate targeting for specific ecosystem services (McClain et al. 2003; Wolf et al. 2011).

Future Challenges

While there has been substantial progress in understanding the effects of wetland heterogeneity, key challenges remain pertaining to its science and management applications. In some cases, the momentum to incorporate heterogeneity into restoration practice has outpaced its scientific justification, leading to high costs and poor outcomes (Palmer 2009). Further research is needed to separate the effects of habitat heterogeneity itself from potentially confounding factors like habitat area or the rarity of a given habitat within a landscape (Palmer 2009).

A key avenue for advancing the science of wetland heterogeneity is more rigorous testing of the effects of heterogeneity in restoration contexts. Recent work indicates that increased heterogeneity can enhance species diversity, food-web functions, and water-quality services in restored wetlands (Able et al. 2003; Larkin et al. 2008; Wolf et al. 2011). Wetland scientists and managers have a growing capacity to restore key components of habitat heterogeneity, e.g., through an improved understanding of how to restore for hydrogeomorphic complexity (Williams et al. 2002) or to jumpstart the work of facilitative ecosystem engineers like *Carex stricta* or *Triglochin maritima* (seaside arrowgrass) (Fogel et al. 2004; Lawrence and Zedler 2011). But the long-term effectiveness of these and other approaches employing hydrologic and biotic feedbacks to create and maintain heterogeneity in wetlands is not well known. More work is needed to critically evaluate the long-term trajectories of restored heterogeneity, the influence of heterogeneity across broader suites of taxa and desired functions, and its sustainability in the face of widespread watershed degradation that is having homogenizing effects on wetland environments and biota.

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Landscape Genetics: Wetlands

25

Stephen F. Spear

Contents

Abstract

Landscape genetics is defined as research that explicitly quantifies how landscape variables (such as configuration and matrix quality) affect patterns of genetic variation and gene flow. Landscape genetic questions are typically focused on recent gene flow and landscape changes, and therefore, landscape genetic studies are often used to address ecological and conservation questions (e.g., barriers and corridors, source-sink dynamics, influence of landscape change) that are difficult to answer with more traditional demographic methods. Landscape genetics uses genetic data as the dependent variable and typically attempts to correlate genetic relationships with several independent variables representing landscape or environmental data, usually from a geographical information systems (GIS) computer environment that allows the landscape data to be visualized and analyzed using spatial statistics.

Keywords

Connectivity · Genetic variation · Gene flow · Landscape genetics

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Introduction

Landscape genetics is defined as research that explicitly quantifies how landscape variables (such as configuration and matrix quality) affect patterns of genetic variation and gene flow (Storfer et al. 2007). Landscape genetic questions are typically focused on recent gene flow and landscape changes, and therefore, landscape genetic studies are often used to address ecological and conservation questions (e.g., barriers and corridors, source-sink dynamics, influence of landscape change) that are difficult to answer with more traditional demographic methods. Landscape genetics uses genetic data as the dependent variable and typically attempts to correlate genetic relationships with several independent variables representing landscape or environmental data, usually from a Geographical Information Systems (GIS) computer environment that allows the landscape data to be visualized and analyzed using spatial statistics.

The term "landscape genetics" was first coined by Manel et al. (2003) and was in response to advances in technology in both population genetics and landscape ecology that resulted in the availability of high resolution genetic and landscape data at a relatively low cost. These advances allowed researchers to be able to examine how contemporary genetic structure was affected by both natural and anthropogenic landscape features. A few studies had addressed landscape genetic questions before 2003 (without using that term), but the number of publications in the field began to rapidly increase after 2005 (Storfer et al. 2010), resulting in many empirical studies, another major review paper (Storfer et al. 2007), and special issues devoted to landscape genetics in journals such as *Landscape Ecology*, *Molecular Ecology*, and *Conservation Genetics*.

Organisms inhabiting wetland systems in a variety of environments have been the focus of several landscape genetic studies and have provided much insight into the ecology and connectivity of wetland systems. Wetlands serve as excellent study systems for landscape genetics because they can be easily modeled as patches that are connected by paths along which environmental variables are measured. This allows for traditional population genetic analyses to be melded with landscape data. Furthermore, many wetland organisms are limited by environmental factors due to their reliance on the discrete habitat type that many wetlands represent, and therefore, genetic variation is highly correlated with the surrounding environment. Wetland organisms that have been modeled in a landscape genetic framework include zooplankton (Michels et al. 2001), frogs (Murphy et al. 2010a, b; Goldberg and Waits 2010), salamanders (Spear et al. 2005; Giordano et al. 2007), turtles (Mockford et al. 2007; Howeth et al. 2008), snails (Wilmer et al. 2008), and fish (Dominguez-Dominguez et al. 2007). Of these taxa, amphibian species have been most widely studied, and there is a noticeable lack of studies investigating wetland plant species.

Estimating Genetic Diversity and Connectivity

Genetic diversity and connectivity in landscape genetic studies is most often assessed using microsatellite DNA markers, although SNPs are a future direction (Storfer et al. 2010). Microsatellites are noncoding regions of the genome characterized by multiple base pair repeats and are thus measured by the length of the repeat region (Fig. 1). Because they represent noncoding sections of DNA, they are not under natural selection and are thus considered neutral genetic markers (Schlötterer 2000). **Neutral markers** allow for inference of gene flow between sites because they are only subject to **genetic drift** and not the potentially homogenizing force of natural selection. Microsatellites typically mutate rapidly and it is thought this is likely due to misalignment of repeat sequences during DNA replication; such "slippage" generates additional repeat units (Schlötterer 2000). Therefore, microsatellites tend to have many alleles (or different forms of a gene), which make them ideal for assessing changes in genetic variation due to recent events. A SNP is characterized by variation in a single nucleotide and therefore can have at most four alleles. However, thousands of SNPs can be identified across the genome, and therefore, using SNPs can provide a better representation of what factors are influencing the entire genome (Kohn et al. 2006). Despite this, broad-scale SNP markers have only been developed for a few nonmodel species, and at the current time, microsatellites are still the most common marker used in landscape genetic studies.

Regardless of the genetic marker used, the raw data consists of a genotype (the identity of the specific alleles for each individual). With neutral markers, the specific genotype is not informative independently; rather investigators are interested in how that genotype relates to other individuals' genotype. In the case of wetland species, individuals are often clustered together, and a method that summarizes the genetic data across populations is typically used. The summary measure can either represent the genetic diversity within a single population or the genetic distance between two populations. Genetic diversity can be represented by metrics such as number of alleles or average **heterozygosity**. In landscape genetics, genetic diversity would be correlated with at-site environmental variables. Far more common in landscape genetics is to correlate genetic distances with among-site landscape variables. There are many types of genetic distances that can be estimated, the assumptions of which are beyond the scope of this article. However, in landscape genetics,

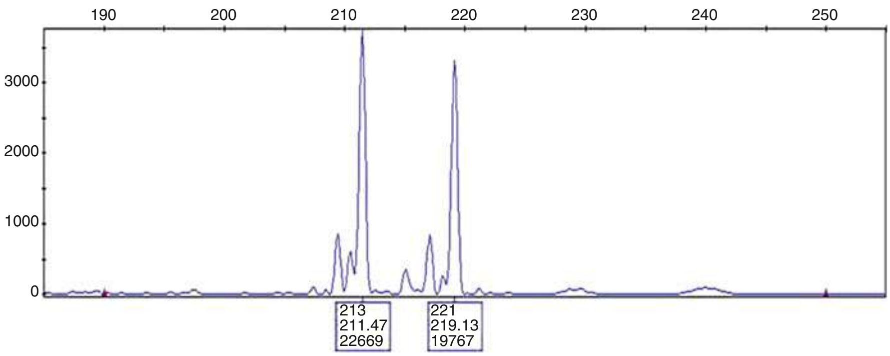

Fig. 1 Visualization of a microsatellite genotype. Each labeled peak represents a unique microsatellite allele that is distinguished by the length of the genetic fragment. Labels represent the number of base pairs and can be used to infer number of repeat units

popular types of genetic distances include F_{ST}, chord distance, Nei's distance, and proportion of shared alleles (Storfer et al. 2010).

Modeling Landscape Connectivity

One of the most important decisions when developing a landscape genetics study is how to model the landscape so as to represent the pattern of how genetic connectivity occurs across the landscape. This essentially is comparable to developing a hypothesis of landscape influence on genetic structure and then modeling that hypothesis. There are three approaches that so far have been most commonly used for identifying landscape influence on genetic connectivity. One method is to create a straight-line network among sites (connecting either all sites or a subset of sites), and values of landscape variables along this network are correlated with genetic distance. Murphy et al. (2010a) used this technique to demonstrate the importance of land cover, topography, and climate on western toad (*Anaxyrus boreas*) gene flow in Yellowstone National Park and Goldberg and Waits (2010) discovered that the importance of land cover type differed between two amphibian species (Columbia spotted frogs (*Rana luteiventris*) and long-toed salamanders (*Ambystoma macrodactylum*) inhabiting the same agricultural landscape. A buffer can also be applied to the straight-line network to incorporate a broader range of landscape variables among sites and test scale effects (Murphy et al. 2010a; Emaresi et al. 2011). This approach has the advantage of being a simple model with a limited number of assumptions. However, if dispersal movements leading to gene flow deviate strongly from a straight line, then this method will not be representative of actual gene flow. Pond-breeding amphibians such as toads may not violate this assumption because they can move over land between wetlands, but wetland species that are more closely tied to the aquatic environment are likely unsuitable for straight-line modeling.

A second common approach is to model connectivity among sites using least cost paths based on a resistance surface (Spear et al. 2010) (Figs. 2 and 3). A resistance surface represents the value of some variable thought to restrict gene flow, and least cost paths model the path of least resistance through this landscape. For instance, modeling gene flow as following a least cost path that maximized flow rate among ponds best explained genetic connectivity in the zooplankton *Daphnia ambigua* (Michels et al. 2001). Such least cost paths have the advantage of allowing researchers to model more realistic movement paths than straight lines, but are difficult to implement when the actual resistance of a variable is unknown. Furthermore, least cost path approaches assume a single path between populations, but if there are multiple suitable paths, the least cost path approach may not approximate landscape influence on gene flow well. A solution that has been proposed is to use electrical circuit theory to model how landscape features affect gene flow (McRae 2006). Circuit theory uses the same resistance surface as used for least cost paths but calculates resistances across all possible paths between sites. Therefore, this method takes into account redundant paths and likely is a more realistic approximation of gene flow. However, this method is not suitable for wetland-breeding species, as it

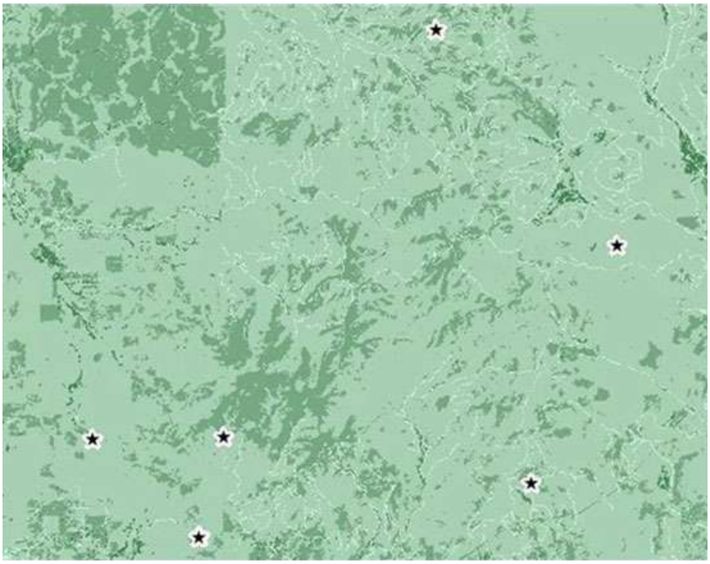

Fig. 2 Example resistance surface for a landscape genetic analysis. Stars represent genetic sampling sites. Background is a land cover map that has been coded to represent resistance. The darker color represents greater resistance to gene flow

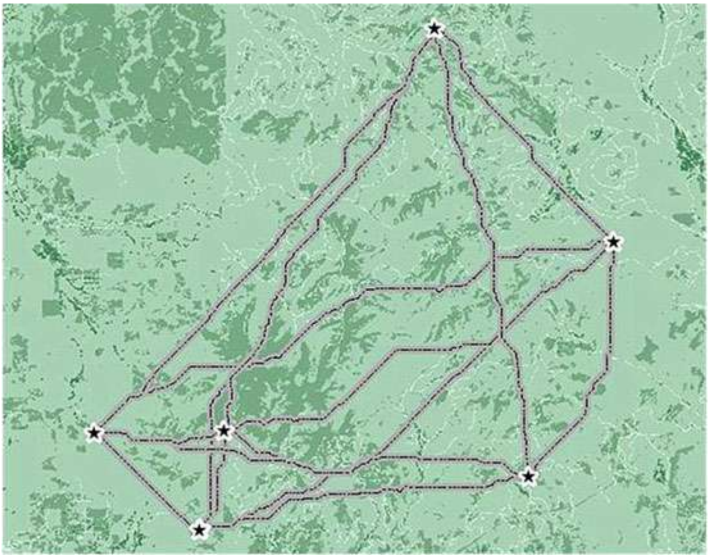

Fig. 3 Example of a least cost path connecting sites across the resistance surface in Fig. 2. Notice that dark areas are avoided by the least cost path

assumes that gene flow could take place at any location across the landscape. A new application of circuit theory for network patches has been developed, though not yet implemented in any empirical studies (McRae and Shah 2009). However, this network circuit theory may hold promise for wetland species due to the discrete nature of this habitat.

One disadvantage of either of the above approaches to landscape genetics in wetland systems is that both are measuring genetic connectivity as a function of the environment between wetlands and not at wetland sites. Therefore, this could limit insight into the characteristics of wetlands themselves that influence genetic connectivity. However, Murphy et al. (2010b) recently applied gravity models to landscape genetics. Gravity models allow researchers to model variables both at and between sites. Murphy et al. (2010b) used these models to demonstrate that wetland depth and heat load at wetlands were both important variables structuring a metapopulation of Columbia spotted frogs. Thus, this methodology may prove to be an important future direction in wetland landscape genetics.

Future Challenges

Landscape genetics is a relatively young and emerging field, and therefore, there are a number of future challenges and opportunities (Balkenhol et al. 2009a). The appropriate statistical analysis technique used to relate genetic and landscape data remains an important issue in landscape genetics. Landscape genetic studies typically use a regression framework, yet usually have pairwise genetic distances. This violates assumptions of nonindependence required for most regression analyses. This complicates both assessment of significance in regression models and typical model selection techniques such as the Akaike information criterion (AIC; Akaike 1974; Burnham and Anderson 2002). Permutation approaches using Mantel tests or multiple regression on distance matrices are one way to assess significance without violating assumptions, but some of these permutation techniques are controversial as to their efficacy (Balkenhol et al. 2009b). Furthermore, these permutation approaches do not address the problem of selection among alternative models. Some alternative methods have been recently proposed (Goldberg and Waits 2010; Van Strien et al. 2012), but have yet to be widely incorporated in empirical studies.

Another promising future application for landscape genetics is the incorporation of adaptive genetic markers in addition to neutral loci (Manel et al. 2010). This would allow researchers to distinguish between environmental factors that affect gene flow by affecting dispersal to those environmental factors that are leading to natural selection on organismal traits. The advances in sequencing large portions of genomic data have facilitated interest in adaptive landscape genetics, as it allows researchers to more easily identify regions of the genome under selection. For nonmodel species (such as most wetland species), there is still much work to be done to identify genetic markers of adaptive significance, but this avenue is especially interesting for predicting the effect of future climate change of species' ability to adapt and persist in novel environments.

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Concepts in Landscape Genetics

26

Stephen F. Spear

Contents

Abstract

There are several concepts that are integral to an understanding of landscape genetics. These include concepts within the field of population genetics, including genetic drift, heterozygosity and neutral loci, which are described in this chapter. Additionally, I discuss issues related to statistical models used in landscape genetics. Overall, the definitions provided in this chapter should better allow a reader to understand terms in other chapters of this book in this volume.

Keywords

Landscape genetics · Population genetics · Statistical models

Concepts

Definition

Genetic Drift

Genetic drift is a process within the field of population genetics that describes how allele frequencies change due to random chance or sampling error (Hartl and Clark 2006). Genetic drift occurs most strongly in the absence of gene

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flow or natural selection and in small populations. The easiest analogy to consider for genetic drift is a coin flip. If two people flip a coin only a few times, it is unlikely that they will end up with exactly the same number of heads and tails. However, if the two people each flip a coin hundreds or thousands of times, it is much more likely that both will have a 1:1 ratio of heads to tails. Similarly, even if two populations are genetically identical initially, but only a few individuals reproduce, it is likely that they will begin to become different due to different alleles being passed on. Population genetic theory also demonstrates that genetic drift leads to loss of genetic diversity, with the rate of loss determined in large part by population size (Hartl and Clark 2006). This is why small populations are a concern from a genetic as well as demographic perspective, since loss of genetic diversity is associated with a reduced ability to adapt to environmental changes as well as inbreeding depression. In landscape genetics, a common assumption is that if populations are diverged genetically and the divergence is correlated with some landscape feature, then genetic drift is occurring due to reduced movement and consequent reduced gene flow between populations. Theoretical population genetic work demonstrates that as few as one successful migrant between populations is sufficient to stall the effects of genetic drift, so a landscape feature that is correlated with genetic divergence likely is quite strong as an isolating mechanism. Furthermore, if the landscape feature is indeed leading to genetic drift, it suggests that the population may already be small and of conservation concern, as isolation is much less likely to lead to detectable genetic change in large populations. Thus, landscape genetics can be a very effective conservation tool through its ability to detect environmental associations with genetic drift.

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Definition

Heterozygosity

Heterozygosity describes the situation when a diploid individual has two different alleles present at the same gene or locus. The opposite of

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heterozygous is homozygous, in which an individual has two copies of the same allele at the same location. Heterozygosity, along with the number of alleles, is a primary metric used to assess the amount of genetic diversity in a population (Hartl and Clark 2006), and is usually presented as a population-level metric. In other words, heterozygosity represents the proportion of individuals within a population that has two different alleles at the same locus. It can be a useful metric of genetic diversity because the more alleles present in a population (and thus, the more diversity), the more likely that individuals will be heterozygous. From a functional perspective, high heterozygosity eliminates the effect of harmful alleles that are recessive. These harmful recessives will only be expressed in homozygous individuals, and the existence of these harmful recessives is one reason why high genetic diversity is considered important for conservation.

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Definition

Neutral Loci

A neutral locus (plural: loci) is a gene that is not under natural selection, and therefore the specific form of that gene has no effect on individual fitness (Hartl and Clark 2006). Thus, neutral loci can only change due to random effects of genetic drift, gene flow, or mutation. Neutral loci are often regions of the genome that do not contain a sequence that codes for an amino acid, or they possibly could be a coding region in which there is no difference among all the alternative alleles with respect to individual fitness. An example of a type of genetic locus that is typically neutral and is commonly used in population genetics is a microsatellite; microsatellites are a sequence of many short repeats and thus are noncoding (Schlötterer 2000). Neutral loci are important in landscape genetics because we often wish to infer how the landscape or environment affects movement between populations, with the assumption that movement equates to gene flow. In contrast, a locus that was under natural selection could be similar in two isolated populations if the populations were under the same degree of natural selection. A drawback of using neutral loci to

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infer conservation status is that genetic diversity is not as important for neutral loci. Therefore, genetic studies that use neutral loci to infer conservation status are assuming that the diversity of neutral loci is representative of genetic diversity in general, which may not necessarily be the case (Reed and Frankham 2001).

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Definition

Population Genetics

Population genetics is the study of how the frequencies of alleles (the various forms of a gene) change within and among populations (Hartl and Clark 2006). A population is typically defined as a group of individuals that are connected by random mating among them, and therefore tend to be genetically similar to one another, although in practice this can sometimes be difficult to assess. Population geneticists are interested in several evolutionary processes such as gene flow, genetic drift (see Genetic Drift), natural selection, and mutation. These processes can all change the genetic composition of populations, which has implications for evolution, ecology, and conservation. For instance, natural selection in different environments can lead to unique adaptations that may eventually lead to speciation. In contrast, the process of gene flow among populations tends to reduce the chance that populations will become different, but it may also provide beneficial genetic diversity that could prevent declines in small populations. In the context of landscape genetics, researchers are most interested in the relationship between genetic drift and gene flow (Storfer et al. 2007). For instance, if landscape or environmental features lead to increased similarity due to gene flow, we typically infer that these landscape features have facilitated gene flow among populations.

(*continued*)

Thus, we assume that individuals are able to physically move through these environments. On the other hand, if populations appear to be genetically different, then genetic drift due to population isolation is inferred, which suggests that individuals are unable to move across the landscape features. An emerging approach is to use landscape genetics to identify environmental features important for natural selection and adaptation. This type of investigation focuses on finding loci that have allele frequencies that are outliers when compared to an expectation of neutral genetic drift. The assumption is that the allele is selected for or against, and that is why it has a higher or lower frequency than expected by chance. These outlier loci are then correlated with landscape or climatic variables that might be agents of natural selection (Manel et al. 2010). These examples highlight how population genetics theory and assumptions form the backbone of more applied fields such as landscape genetics and conservation genetics.

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Definition

Statistics for Landscape Genetics

One of the major considerations in conducting landscape genetic analyses is the appropriate statistical analysis. Many, but not all, landscape genetic studies are interested in correlating the levels of genetic differences or genetic diversity with landscape or environmental variables. The most common approach to such correlative approaches is multiple linear regression, but the problem with

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this approach is that genetic differences are expressed as pairwise distances in which the same population is used in multiple comparisons (Storfer et al. 2007). Clearly, in this situation, the genetic differences are not completely independent. Furthermore, there might be violations of independence simply due to the fact that populations that are close to each other in space are similar because they are neighbors, and as such cannot be considered completely independent. Thus, a major emphasis in landscape genetics is to identify statistical techniques that are not sensitive to violations of statistical independence. Some of the first possibilities were Mantel and partial Mantel tests. Mantel tests are regressions, but use a randomization approach to assess significance that was believed to deal with the nonindependence problem, and Mantel tests are currently the most widely used landscape genetic statistical analysis (Storfer et al. 2010). However, multiple studies have shown that Mantel tests often do not provide reliable results and may be biased (Raufaste and Rousset 2001, Balkenhol et al. 2009, Guillot and Rousset 2013, Graves et al. 2013). Other possibilities are models that include a regression coefficient that directly accounts for the nonindependence (such as a spatial term) (Spear and Storfer 2010), but such analyses have not been rigorously assessed in landscape genetics to this point. Analyses based on transportation flow, called gravity models, allow analysts to include both between-site landscape variables and landscape variables at the site to test their effects on genetic differences (Murphy et al. 2010). So far, gravity models seem to be reliable and provide the additional information of how habitat factors at a population influence genetic divergence. There are other methods that have been proposed as well, but in general, an appropriate statistical method is one of the grand challenges of landscape genetics.

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Section IV

Hydrology to Wetlands: Importance

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Wetland Hydrology

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Todd C. Rasmussen, James B. Deemy, and S. Lynsey Long

Contents

Abstract

Hydrology profoundly affects wetland habitats and recruitment for both plants and animals, as well as in adjacent aquatic and terrestrial environments. Water storage in wetlands promotes flood attenuation and sediment retention; while wetland soil saturation and dissolved oxygen concentrations promote unique biogeochemical reactions that foster nutrient and contaminant removal. Wetland

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water inputs and outputs are a function of hydrologic exchanges with atmospheric, surface, subsurface, and tidal systems. These exchanges are described using water budgets, hydropatterns, and residence times, which characterize the unique behavior of individual wetlands. Future challenges to wetland hydrologic functioning include natural and human alteration of wetlands and the surrounding landscape.

Keywords
Hydropattern · Water budget · Residence time · Hydrologic exchanges · Future challenges · Human alteration

Definitions

Nymphoides indica, a rooted floating–leaved macrophyte of monsoonal wetlands in Asia (Photo by Beth Middleton)

Definition: macrophyte
Author: Beth Middleton
A macrophyte is a type of vascular plant, which grows in wetlands, and includes emergent, floating, or submersed species.
Definition: zonation pattern
Author: Beth Middleton

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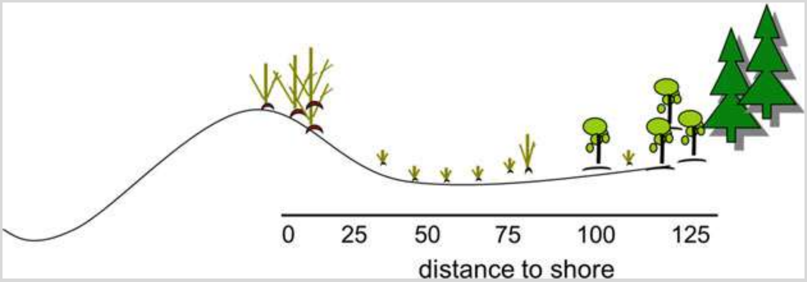

Zonation patterns of vegetation follow salinity and water depth patterns from shoreline to inland for the southeastern United States (e.g., salt marsh to freshwater bald cypress swamp)

Zonation patterns are apparent in vegetation, which form along environmental gradients. In wetlands, the underlying environment gradients are related to water depth, salinity, and other biogeochemical factors, and in the changes of dominant vegetation along the gradient.

Introduction

Wetlands are a unique landscape feature, which occupy the transition zone between permanently saturated aquatic environments (e.g., lakes, rivers, estuaries, oceans), and terrestrial environments with a root zone free of saturation during the growing season. Wetlands create unique biogeochemical conditions that differ from aquatic and terrestrial environments due to the low dissolved oxygen concentrations found in wetland soils. Anaerobic conditions are more likely to develop in saturated than in unsaturated soils due to the low oxygen solubility, slower rates of water advection, and oxygen diffusion in water compared to air.

Soil saturation thus serves as an indicator of the dissolved oxygen state of the soil-water system; wetter systems have higher water levels and lower soil dissolved oxygen concentrations, while drier systems have lower water levels and higher dissolved oxygen concentration. While anaerobic conditions promote many biogeochemical transformations, they also create adverse conditions for root survival and growth for many plant species. Because of these low oxygen concentrations, wetlands are among the most biologically productive ecosystems on the landscape, supporting diverse assemblages of species with special morphological and physiological adaptations, which enable them to survive and prosper in these otherwise harsh conditions.

Wetlands are generally found in low-energy environments, that is, on flat areas or on shallow slopes where water lies at or near the land surface, either above or below. Wetlands tend to form where surface water and groundwater accumulate within topographic depressions, such as along floodplains, within kettles, potholes, bogs, fens, lime sinks, pocosins, Carolina Bays, vernal pools, ciénegas, pantanos, tenajas, and playas, and behind dunes, levees, and glacial moraines. *Seepage* wetlands form where groundwater discharges on slopes, as well as near the shore of streams, lakes, and oceans. *Fringe* wetlands form along shorelines, with periodic inundation caused by water exchanges with adjacent waterbodies, such as floods and tidal action. *Perched* wetlands form in areas above low-permeability substrates where infiltration is restricted, such as clay, rock, or permafrost.

Because wetlands lie in low-energy environments, water velocities are slow and surface areas expand and contract as the water stage changes, promoting water storage. Wetlands therefore moderate hydrologic variability, storing flood flows and reducing flow velocities during wet weather, and releasing water slowly during dry weather. This water storage capacity is also related to the ability of wetlands to trap nutrients and sediments.

Wetland hydrology is integrally linked to other ecosystem processes (e.g., biology, soils, geochemistry) by complex processes with many potential feedbacks. For example, wetland microtopography can induce heterogeneous subsurface flows that create biogeochemical *hotspots*, which can vary over both space and time (Frei et al. 2012). Also, biogeochemical processes in wetlands linked to estuarine and marine systems demonstrate nonlinear relationships within nutrient cycles as a function of hydrologic behavior (Howarth et al. 2011). Understanding the coupled mechanics of wetland systems is important for understanding their influence on local, regional, and global ecosystem dynamics.

Hydrologic Descriptors

An important hydrologic descriptor of wetlands is the general elevation of water levels relative to the soil surface. Deep-water areas with few, if any, emergent macrophytes are known as *open water*. Vegetation present in these areas is usually not attached to the wetland bottom but may be floating on the water surface. However, tall woody species occasionally recruit during severe drawdown events, such as those caused by abnormally long drought, and may persist when water levels return to normal (e.g., *Taxodium distichum* and *Nyssa* spp. in southeastern swamps). Shallow areas containing substantial quantities of emergent macrophytic vegetation, either living or dead, are called *emergent zones*. Wetlands may also have large areas that are not flooded, but have a near-surface water table with macrophytic vegetation. Cowardin et al. (1979) proposes a general relationship between wetland water levels and hydric states.

Staff gages are used to monitor wetland water levels when the water level is above the ground surface (Fig. 1). The staff gage is a vertical scale that serves to indicate the elevation of water, or *stage*, with respect to a reference elevation. In some wetlands after a flood event, a water line can be observed on periodically submerged

Fig. 1 Staff gages are used to monitor the surface water level (stage). Multiple staff gages are used to monitor large stage variations, with gages more distant from the center of the wetland used during wet periods and gages in the deepest part of the wetland used during dry periods (Photo by the Illinois State Water Survey)

vegetation, or where floating debris (e.g., leaves, trash, branches) has been deposited on the shoreline or lodged in bushes and trees. In other cases, the normal water level can be observed as a horizontal stain on woody vegetation.

Piezometers are used to find the water level below the ground surface. A piezometer is a small-diameter perforated tube, which is installed within the soil to monitor groundwater levels over a specific interval of time. The perforated zone of the tube should be narrow to minimize the depth of soil monitored, and placed within a unique hydrogeologic unit, such as a soil horizon or geologic layer. Depth-to-water probes are commonly used to detect the belowground water level. For continuous measurements, automated dataloggers can be used with *pressure transducers*, *vibrating wire sensors*, or *capacitance sensors* to monitor above- and belowground water levels. Soil saturation can be measured using *time-domain reflectometry*, which determines the water content using the electromagnetic properties of a wave pulse passing through probes placed in the soil.

Hydropattern

The wetland *hydropattern* incorporates the timing, duration, and distribution of wetland inundation by quantifying the duration and frequency of water-level fluctuations. Spatial and temporal changes in water levels are important determinants of habitat suitability for many aquatic flora and fauna. The reproductive success of these wetland species can be adversely affected when fluctuations are not correctly synchronized with their developmental stages. The hydropattern of some systems, such as tidal marshes, fluctuates dramatically over short periods of time; other systems, such as seasonally flooded bottomland hardwood swamps, fluctuate more slowly over time. Other wetland systems are more static and may not display substantial short- or long-term variability. The wetland hydropattern is a function of the landscape topography and the net difference between inflows and outflows from atmospheric, surface, subsurface, and marine sources.

The *hydrograph* relates the wetland stage as a function of time (Fig. 2a). Ideally, the hydrograph should have sufficient duration and frequency so that it captures the range of possible hydrologic variability. Interannual, seasonal, event, and daily water-level fluctuations in a wetland should then be apparent. Water levels typically rise in response to precipitation, and then decline slowly over time. The *time to peak* is the time between the peak precipitation and peak stage. Times to peak are short in urban areas with high amounts of impervious surfaces and channels that have been modified to increase stream velocities. Times to peak are longer in forested areas with few impervious surfaces and high channel complexity.

The *stage-duration* relationship provides a descriptive summary that indicates how long a typical flood lasts for each stage (Fig. 2b). Lower stages typically show longer durations than higher stages. This approach is useful for characterizing water level variability using the duration that a specified water level is exceeded and can also be used to determine the seasonal nature of inundations by segregating the data into specific time frames. While the stage-duration approach successfully captures the duration of time that the system is flooded, it does not indicate the frequency of flooding.

The *stage-frequency* relationship is an alternative approach that quantifies the frequency in time that the wetland is observed to exceed a range of specified stages (Fig. 2c). This approach provides a cumulative frequency plot (or table) that can be used to calculate exceedance probabilities. The mean, median, and extreme stages (e.g., 1, 10, 50, 90, and 99 percentile probability) can be estimated using the stage-frequency relationship.

The two previous relationships can be combined to yield the *stage-duration-frequency* relationship, which is used to determine the hydrologic behavior for a range of probabilities and durations. That is, a system whose water levels vary slowly over time displays a different duration and frequency than rapidly fluctuating systems. Partitioning or stratifying data sets into seasonal or other periods may improve the characterization of water level conditions (Mitsch and Gosselink 2007).

Water Budgets

A water budget accounts for the gains and losses of water storage in a wetland system. When hydrologic inputs exceed outputs, the surplus water increases the water storage within the wetland, causing water levels to rise. For example, if rainfall inputs exceed evaporation, seepage, and discharge outputs, then the total effect is to

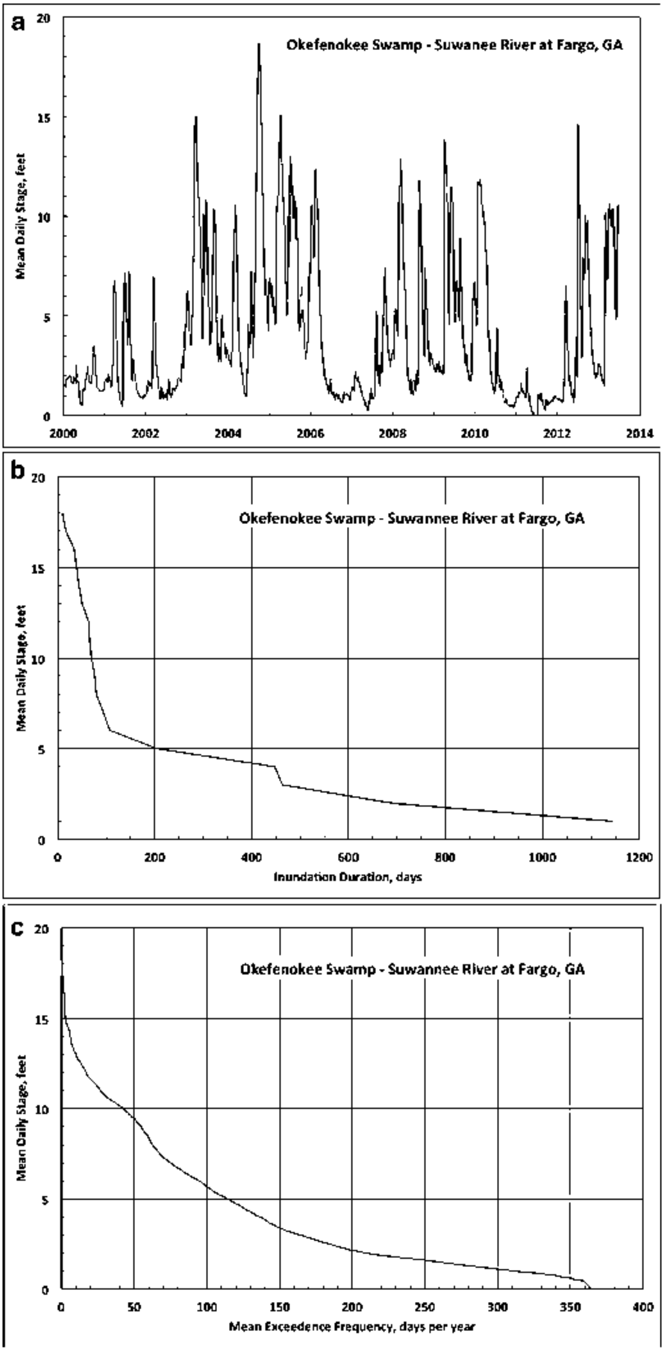

Fig. 2 Hydropattern example for Okefenokee Swamp showing (**a**) hydrograph, (**b**) stage-duration relationship, and (**c**) stage-frequency relationship

raise water levels within the wetland. Hydrologic inputs and outputs are expressed in units of volume, $\mathbf{V}$, or depth, $\mathbf{D} = \mathbf{V/A}$, (where $\mathbf{A}$ is the wetland area) per unit time, $\mathbf{\Delta V/\Delta t}$ and $\mathbf{\Delta D/\Delta t}$, respectively. These water level and volume changes result from accumulated inputs and outputs over space and time. Wetlands attenuate downstream water-level fluctuations by storing water during wet periods and releasing them during dry periods.

Subsurface water storage depends on the specific yield (i.e., the drainable porosity) of the organic and mineral sediments in the underlying soil. The specific yield, $\mathbf{S_y} = \mathbf{(\Delta V/\Delta D)/\Delta A}$, is the volume of water released, $\mathbf{\Delta V}$, per unit decline in water level, $\mathbf{\Delta D}$, per unit area, $\mathbf{\Delta A}$. In many cases, organic and mineral sediments may remain at or near saturation as water levels fall. In these cases, only a small volume of water is released from the sediments during drainage. Specific yields of sediments are strongly influenced by their particle size distribution and chemical composition; sands have larger specific yields, while clays and organic soils have lower specific yields.

Residence Time

The hydrologic residence time within a wetland provides a metric that can be used to indicate the rate of water exchange (or turnover rate) within the system. Some wetlands exchange water quickly, having a short duration of time that water remains within the wetland, while water may travel very slowly through other wetland systems.

The mean residence time, $\boldsymbol{\tau} = \mathbf{V/Q}$, is the ratio of the volume of water within the wetland, $\mathbf{V}$, to the average flow through the wetland, $\mathbf{Q}$. Short residence times occur when the flow through the wetland is large compared to its volume, and longer residence times occur when the flow is small compared to its volume. The residence time of a wetland is often related to its hydropattern; wetlands with large water-level fluctuations may have shorter residence times, such as in tidal marshes. On the other hand, some wetlands may fluctuate rapidly due to large changes in inflow; yet have very long residence times due to slow loss rates.

Different parts of a wetland may exhibit different residence times; water in an active, flow-through portion of a wetland may have shorter residence times than water in an inactive, isolated portion of a wetland. While each section of a wetland may have identical hydropatterns, the flow is concentrated in one area, leaving other areas with stagnant conditions. The same equation can be applied regardless of the situation; each section would be characterized using the volume and flow rate in the section of interest. Residence times for dynamic systems are more complex than steady flow systems. In these cases, the residence time is not a constant, with longer residence times during periods when outflows exceed inflows, and shorter residence times inflows exceed outflows. This situation results as a consequence of the fact that residence time is influenced by the rate of adding new water as well as removing old water from the wetland.

Hydrologic Exchanges

Wetland hydropatterns, water budgets, and residence times are influenced and controlled by water exchanges (inflows and outflows) with the surrounding environment. The four general types of wetland water exchanges include atmospheric, surface, subsurface, and tidal, and are discussed in the following sections. Brinson (1993) characterized wetlands based on the type of hydrologic inflow, noting that wetlands have different types of inflow and outflow patterns. That is, some wetlands have simple exchanges with adjacent waterbodies, such as when a wetland receives water from a flooding river during the rising stage and returns water to the river during the falling stage.

Another type of wetland hydropattern is due to tidal exchange along the coast, where water moves in and out of wetlands to its original source. In other cases, exchanges may be between different types of waterbodies and have a hybrid character, so that inflows may be of one type (e.g., subsurface inputs) and outflows may be of another (e.g., evapotranspiration). Identifying and characterizing the wetland hydrologic exchanges is critical for understanding hydrology and ultimately in managing wetland systems.

Atmospheric Exchanges

Atmospheric exchanges include *precipitation inputs* (e.g., rain, snow, hail, sleet, freezing rain, fog drip, dew, frost) and *evapotranspiration* (e.g., evaporation, transpiration, interception). Water levels in wetlands that are dependent on atmospheric exchanges tend to be more affected by climatic signals than those dependent on groundwater sources (Orme 1990). Precipitation generally occurs as discrete events, characterized by the intensity, duration, frequency, and areal extent. In aggregate, these precipitation events can be described using monthly and seasonal averages, along with longer-term variability associated with decadal and climatic fluctuations.

The loss of water from a wetland by evapotranspiration (process by which water is lost from the water surface as well as from plant leaf and stem surfaces) can have large effects on water levels. Evaporation dominates when open water is present. Saturated soils may lose nearly as much as open water, but not if a litter or mulch layer is present. Transpiration dominates in systems with little open water and large coverage of living vegetation. Evapotranspiration rates are affected by leaf and stem area, air, water, and plant temperatures, atmospheric humidity, wind speed, and the water potential of exposed soils.

Precipitation and evaporation can be readily measured using raingages and evaporation pans, respectively. These tools are relatively inexpensive and provide reliable estimates of daily atmospheric exchanges. A single raingage is usually sufficient for small wetlands (e.g., smaller than 100 ha), but multiple raingages may be required for larger wetlands, especially if significant spatial variation in rainfall is present. Pan evaporation rates can be used to estimate evapotranspiration rates, with a single pan usually being sufficient for all but the largest wetlands.

However, local effects of shading and wind shelter can adversely affect the accuracy of the measurements.

Surface Exchanges

Surface water exchanges result from a large number of mechanisms, including overland (or sheet) flow, direct exchange when the channel of a river or stream flows through the wetland, overbank flooding during wet weather when the channel is separated from the wetland by a levee or floodplain, and along the edges of lakes, estuaries, and the ocean. These surface exchanges result in either constant or episodic hydrologic communication between the surface water and the wetland. Streamflow can be divided into two types, *baseflow* and *stormflow*. Baseflow is that component of flow found during low flow periods, while stormflow refers to the response to precipitation events. If a stream was flowing before the rainfall (a typical situation), stormflow is the flow attributable to the rainfall and occurs in addition to the baseflow.

Surface water flows can be estimated using flow measurement control devices, such as *weirs* (which require a pool upstream and are not satisfactory when elevated sediment concentrations are present), *flumes* (which tend to flush sediments more effectively than weirs), and *culverts* (which are less accurate for surface water flow measurement). The relationship between stage and streamflow discharge is called the *rating curve*. The stream discharge is readily found using the observed stage and the rating curve.

Subsurface Exchanges

Subsurface inflows to wetlands result from shallow, topographically induced drainage from nearby uplands, or from discharges of regional, confined aquifers. Subsurface outflows from wetlands result from downward (or lateral) flow from the wetland to underlying surficial aquifer or to deeper, confined aquifers. Shallow inflows may result from perched drainage (also called interflow) on top of lower-permeability units such as clay beds, soil horizons, or even permafrost. Shallow subsurface inflows may also arise when the water surface within the wetland lies below the water table in the underlying surficial (unconfined) aquifer.

Subsurface inflows from deeper sources may arise when confined aquifers discharge into the wetland, either at discrete points (such as springs) or as diffuse upward leakage through the underlying surficial aquifer. Shallow inflows may respond more rapidly to individual storm events, as well as to seasonal and climatic changes. This rapid response is because interflow and water levels in shallow aquifers tend to be more sensitive to net changes in atmospheric flux (precipitation less evapotranspiration) in nearby upland areas.

Shallow subsurface outflows may occur if a layer of low permeability underlies the wetlands and causes the water to perch. In these cases, a low point on the perimeter of the wetland allows water to exit the wetland as overland flow, channel flow, or interflow. When wetland water levels are contiguous with water levels within the surficial aquifer, flow through the surficial aquifer may be affected by the wetland.

Normally, surficial aquifer water levels dip in the direction of water flow, while the general water level elevation in the subsurface layers of the wetland are more uniform (Fig. 3a). Thus, wetland water levels are lower in the upgradient direction and higher in the downgradient direction. As a result, aquifer discharge conditions are present at the upstream end of the wetland, and recharge occurs at the downstream. This type of flow-through wetland may account for most of the flux of water through a wetland that has no readily apparent inflows or outflows. Finally, recharge to deeper, confined aquifers may occur when subsidence or collapse has breached the confining layer that isolates the aquifer from the surface. In this case, water level increases in the wetland during wet-weather periods may cause direct recharge to the deeper aquifer (Mitsch and Gosselink 2007).

Groundwater flow normally varies over both space and time, so that multiple vertical and horizontal measurements at a sufficiently frequent time interval may be needed to characterize the system. Spatial variability can be determined by placing an array of piezometers near the wetland (Fig. 3b). Multiple piezometers (called a *nest*) can be placed at different depths at each location to evaluate the magnitude of vertical flow (Fig. 3c).

Water quality sampling can be used as an independent method to determine the source of water within wetlands. For example, if shallow groundwater is moving laterally into the wetland, and groundwater is also moving upward into the wetland from a deeper aquifer, then the geochemical signature of each source can be used to evaluate the relative magnitude of each inflow relative to the total.

Tidal Exchanges

Coastal wetlands are similar in many ways to freshwater wetlands, except that they are transitional between marine, freshwater, and terrestrial environments. Coastal wetlands occupy similar landscape positions as lacustrine wetlands, except that water quality can have greater variation due to mixing of marine and freshwater. Besides water-quality variation, water levels in *tidal* wetlands vary with the regular ebb and flow of ocean water levels, which can affect wetlands many miles inland. The hydrodynamic conditions imposed by tidal fluctuations can cause rivers to slow, or even reverse flow direction, many miles from the coast. The magnitude of this effect usually diminishes with distance upstream, and is a function of local channel features. The combination of water level and water quality changes promotes ecologic diversity and productivity in these wetlands.

Many coastal wetlands are affected by nearby freshwater inputs, especially in estuarine environments. In these cases, occasional, large stormwater inflows can cause rapid changes in the salinity, temperature, dissolved oxygen, and sediment

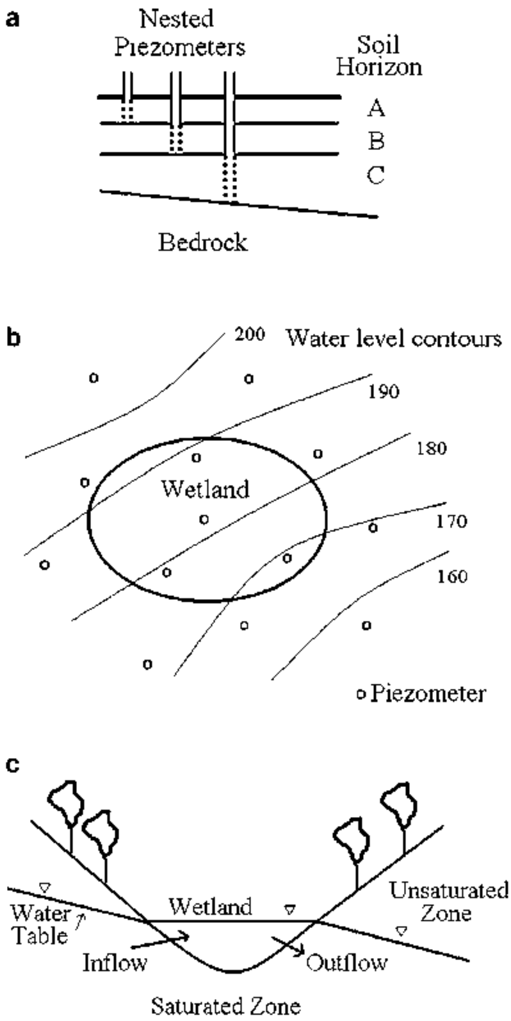

Fig. 3 Piezometers are used to monitor subsurface water levels: (**a**) Multiple piezometers (called a *nest*) can be installed at different depths if vertical flow is suspected; (**b**) Network of piezometers are used to map water levels near a wetland and contours determined by interpolation between piezometers; and (**c**) Subsurface water movement in the vicinity of the wetland inferred from piezometer data

concentration within the wetland. These inputs can be beneficial in some cases, such as historical sediment deposition in the Mississippi River Delta of southern Louisiana, in contrast to the current practice of diverting stormflows away from coastal wetlands, which has led to regional subsidence and saltwater intrusion. In other cases, these inputs can be detrimental, as when increased urban wastewater and stormwater inputs to coastal estuaries alter natural conditions.

Differentiating marine from freshwater (both surface and subsurface) inputs can be achieved using geochemical information. Marine water has high concentrations of sodium chloride type water, while freshwater inputs normally have markedly lower specific conductivities and total dissolved solids. Depending upon location, subsurface inputs are intermediate, with possibly distinct geochemical signatures. For example, a calcium signal may be present in regions with carbonate aquifers. The degree of tidal flushing can thus be monitored using water quality data to characterize the residence times of water within the system. Fluxes can then be estimated using a water budget approach.

Future Challenges

Wetlands change over time due to both natural processes (e.g., beaver activity, natural erosion, and eutrophication) and those caused by humans (e.g., upstream development, wetland removal, cultural erosion, and eutrophication). Some of the many factors that cause wetland hydrologic functions to change over time are discussed here, especially those that compromise the important ecosystem services those wetlands provide.

Natural Change in Wetlands

Wetlands result from many natural processes. Rivers create floodplains, which provide a landscape position that promotes wetland development. Glaciers scour the landscape, leaving behind features that also promote wetlands. Tectonic uplift and subsidence create depressional features that are favorable to wetland formation. Carbonate aquifers dissolve over time, leaving behind depressions where wetlands can form. Erosion transports sediment out of natural channels, leading to downcutting and deepening of channels, which leads to a lowering of riparian water tables and the reduction of overland flows, both of which alter wetland saturation.

Wetlands can also modify their environment as they mature; peats may substantially modify the original landscape by filling in the depression that originally created them. Other biological forces also promote wetland formation. Some examples include beaver activity, which creates impoundments that form natural wetlands in habitats that are favorable to their needs, and large woody debris dams that impound shallow wetlands. Wetlands age over time, slowly filling in with external sources of materials, such as sediments from upland erosion, as well as with detrital materials from wetland vegetation. Rates of deposition of these materials can be slow, such as in oligotrophic systems with small upstream catchment areas. Alternatively changes can be rapid, such as in nutrient-rich areas with large upstream locations prone to extensive natural erosion.

Wetlands in a natural setting are constantly being formed and lost, so that wetland hydrology also changes over time. Reducing wetland water storage decreases the residence time and alters the hydropattern. The dynamic nature of hydrology (especially when applied to wetlands) means that wetlands cannot be investigated apart from their regional environment.

Human Alteration

Humans have substantially increased hydrologic disturbances within watersheds (Azous and Horner 2001). These changes generally cause increased sediment production and transport, as well as increases in nutrient concentrations and loads. Such increases cause reductions in wetland water storage due to sediment trapping and nutrient uptake with subsequent deposition of organic sediments.

Routing stormwater runoff from urban, industrial, and agricultural areas can increase surface inflows to wetlands. Inflows are also altered when hydraulic structures, such as reservoirs, canals, levees, dikes, revetments, and jetties obstruct or alter natural hydrologic patterns. Many of these alterations resulted from efforts to drain wetlands. Outflows from wetlands were increased by the construction of drainage ditches, channels, and canals, or the removal of natural barriers, such as vegetation, and by straightening streams.

Other efforts to drain wetlands used groundwater extraction techniques, such as underground tile drains and pumping wells that lower groundwater levels. Lowering of water tables can affect wetlands by increasing subsurface drainage from the wetland to the point of groundwater extraction. Ditches and tile drains increase the discharge of shallow groundwater, thus lowering water tables in their vicinity. Tile drains promote shallow groundwater flow, thus favoring drier conditions within the wetland.

Drainage of agricultural fields may reduce surface water inflows, lower water tables, and reduce the seasonal period of soil saturation. Groundwater pumping in the vicinity of the wetland can lead to a reduction in shallow aquifer water levels, while irrigation may increase water levels, resulting in either decreases or increases in wetland water levels, respectively. The effects of regional groundwater pumping tend to manifest themselves as slow (and in some cases, rapid) declines in regional, confined aquifer levels. These declines in regional water availability are then transmitted to changes in wetland hydrology by reductions in diffuse upward leakage or direct connections to wetlands, resulting in the lowering of water levels in wetlands.

While beaver-created wetlands once dotted the landscape, trapping and eradication efforts have substantially reduced the populations of these animals. Also, the clearing of large woody debris from rivers may also reduce wetland formation. Harvesting of riparian vegetation, particularly the larger diameter trees could result in poorer recruitment of large woody debris. While wetlands clearly affect vegetation, fish, and wildlife, it is also true that these biological factors affect wetlands.

The construction of surface-water obstructions (e.g., levees, road and bridge embankments) may deprive the wetland of natural flows. In some cases, obstructions may not substantially alter total wetland outflows. Obstructions may just alter the stage-discharge relationship, requiring a higher water level in order to pass an equivalent discharge. This change in discharge rate may have both positive and negative effects on wetlands. Increased water levels can alter the natural storage ability (an adverse effect for flood mitigation), but may increase the residence time (a positive effect for water quality enhancement).

Alteration of the coastal morphology by altering the landscape (e.g., channel dredging and canal construction) can adversely affect natural wetlands by increasing saltwater intrusion rates. Density-dependent stratification of estuarine waters may alter saltwater mixing in coastal areas. Construction of deepwater navigation channels may allow saltwater to migrate inland, resulting in increased salinities.

Coastal groundwater pumping affects coastal wetlands by reducing artesian pressures in underlying confined aquifers, which may then cause a reduction in discrete and diffuse upward leakage. Groundwater pumping may also cause coastal subsidence, resulting in the effective lowering of the ground surface relative to the sea level, causing the intrusion of saline water into coastal wetlands. Pumping from shallow aquifers can also lower coastal zone water levels, causing local dewatering of coastal wetlands. Shallow disposal of septic wastes can alter local groundwater quality by increasing organic and nutrient loading and decreasing dissolved oxygen concentrations, which may subsequently affect local wetlands.

Restoration

Efforts toward restoration of the hydrologic function of compromised wetlands are expanding (Means and Hinchee 1999). Efforts are also being undertaken to create artificial wetlands that take advantage of natural functions that wetlands provide (Kadlec and Wallace 2001). For example, efforts to mitigate stormwater runoff include designing and constructing artificial wetlands that mimic the beneficial hydrologic effects of natural wetlands. These efforts seek to achieve specific management goals that require reductions in nutrients, sediments, and peak flows. It is also apparent that increased stormwater loads, along with decreased retention times, substantially decreases the effectiveness of wetlands in storing water during flood periods and subsequently releasing them during dry periods. The resulting hydrologic performance of affected wetlands may be compromised.

Regardless of whether impaired wetlands are being restored, or new wetlands are being created, the intent is to recreate the hydrologic behavior that maximizes ecosystem services. The emphasis in these cases is the design and evaluation of alternative strategies for wetland restoration. Manipulating the stage-discharge relationship can control water levels in wetlands. Changing the elevation of an outflow structure (e.g., raising the base of an outlet weir elevation) changes the wetland water levels and flooded areas. The base elevation along with the rate of change in discharge with elevation can be adjusted using outflow structures of different sizes and shapes, depending upon the desired outflow characteristics. Other hydrologic alteration possibilities include closing of ditches and drains, thus reducing wetland outflows. Removing artificial obstructions, such as roads and berms can also improve flow through the wetland by recreating natural hydrologic communication with neighboring waterbodies.

These principles apply not just to freshwater wetlands but also apply to the restoration of tidal wetlands, which requires the recreation or simulation of natural hydraulic conditions (Zedler 2001). Weirs and ditch plugs are devices used in tidal marshlands to maintain minimum water levels. Weirs are useful because the bottom elevation of the weir controls the minimum elevation on the upstream side, but allows higher flows to pass unaffected. Ditch plugs provide the same control, but are more susceptible to destruction during high flows.

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Hydrology of Coastal Wetlands

28

Ralph W. Tiner

Contents

Abstract

Hydrology – the frequency, duration, and timing of inundation and/or soil saturation - is the lifeblood of wetlands. The presence of water strongly influences the physical and chemical environment, biota, productivity, and connectivity between wetlands and waterbodies and among habitats within wetland complexes. Wetlands occurring along the shorelines of the world's oceans, estuaries, and tidal rivers are called coastal wetlands or tidal wetlands. Their hydrology is

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largely influenced by the tides and factors that affect the tides such as the position of the moon and sun relative to the Earth's surface, local weather conditions, and river discharge. Consequently, site wetness is dynamic, changing daily. For example, some portions of coastal wetlands are inundated nearly every day by the tides, while other portions are infrequently flooded.

Keywords
Coastal wetland · Tidal wetland · Salt marsh · Brackish marsh · Mangrove · Tidal swamp · Tides · Hydrology

Introduction

Hydrology – the frequency, duration, and timing of inundation and/or soil saturation – is the lifeblood of wetlands. Hydrology strongly influences the physical and chemical environment, biota, productivity, and connectivity of habitats within wetland complexes and between wetlands and waterbodies. Therefore, hydrology has a powerful influence on the usage of wetlands by humankind and other animals. Wetland hydrology (e.g., prolonged flooding and/or saturation) typically creates anaerobic soil conditions that have a great effect on plant and animal life. The degree of wetness separates wetlands from terrestrial habitats, produces biological diversity among wetlands, and provides wetlands with unique properties and accompanying functions, which make them highly valued natural resources. Wetlands typically form wherever there is a seasonal excess of water for a few weeks in most years during the time when temperatures are warm enough to initiate and support plant growth.

Wetlands occurring along the shorelines of the world's oceans, estuaries, and tidal rivers are called coastal wetlands or tidal wetlands. Their hydrology is largely influenced by the tides and factors that affect the tides such as the position of the moon and sun relative to the earth surface, local weather conditions, and river discharge. While tidal beaches and rocky shores along the world's oceanic shoreline have been included as tidal wetlands, the majority of the world's tidal wetlands are marshes, tidal flats, and mangrove swamps. These types have formed in places where ocean currents and wave action have been reduced allowing sediments to accumulate, creating intertidal environments. Coastal wetlands are typically found behind protective features such as spits or barrier islands, in sheltered (semi-enclosed) embayments, or along coastal rivers. While wetlands along large freshwater lakes, most notably North America's Great Lakes, have been described as coastal wetlands, the effect of tides on these coastal wetlands is minimal so that these are not considered tidal wetlands. The focus of this summary will be on the hydrology of tidal wetlands in North America. The discussion will not address the effects of hydrology on water quality or biota. Much of the text has been adapted from *Tidal Wetlands Primer: An Introduction to their Ecology, Natural History, Status, and Conservation* (Tiner 2013).

Salt marsh in southern Maine, USA (Photo by Ralph Tiner)

Water Budget

The degree of wetness in a wetland at any point in time is a net result of water inputs versus outputs. Inflows increase site wetness with water coming from four sources: (1) precipitation (P) – rain, snow, sleet, hail, or fog, (2) surface water inflow (Si) – rivers, streams, and surface water runoff from the land, (3) groundwater inflow (Gi) – water discharged to the wetland surface (seepage), and (4) flood tides (Ti). Outflows reflect water losses from: (1) evapotranspiration (ET) – the combination of evaporation and plant uptake of water and loss through transpiration, (2) surface water outflow (So) – water draining to rivers or streams, (3) groundwater outflow (Go) – water recharging underground aquifers, and (4) ebb tides (To), which are falling tides that drain water off the surface and lower water tables. The water budget is expressed by a simple formula that compares inflows versus outflows and yields a net change in volume (ΔV):

$$\Delta V = [P + Si + Gi + Ti] - [ET + So + Go + To]$$

When the inputs exceed the outputs, the wetland stores water either in its soil or both in the soil and on its surface (inundation). When the opposite occurs, there is a net loss of water from the wetland and there is no surface water and the water table drops below the surface. The water table and presence of surface water are dynamic characteristics of wetlands. The main difference between the water budget equations for a tidal versus a nontidal wetland is thus the tidal input and output variables. While the tides are just one factor affecting site wetness, tides are the dominant factor in wetness for most tidal wetlands, especially those subject to frequent tidal flooding. Along the upland border, however, groundwater discharge and surface water runoff may play significant roles in site wetness as illustrated by the presence of seepage areas with distinctly different vegetation (i.e., species with more freshwater affinities) than found in the contiguous salt marsh. For freshwater tidal river swamps, surface water inflow during spring freshets is a major factor contributing to the annual water budget, especially for those infrequently flooded by tides. High evapotranspiration coupled with infrequent tidal inundation has an enormous effect on site wetness in semiarid and arid regions or areas with Mediterranean climates. Given the significance of the tides as a driver of coastal wetland hydrology, an introduction to the tides is critical for understanding the hydrology of coastal wetlands.

The Nature and Variability of Tides

Observed tides are the result of the gravitational forces of the moon and the sun exerted upon the water on the Earth as modified locally by weather conditions (e.g., wind and precipitation). According to Newton's law of gravity, the force of attraction between two bodies is proportional to the product of their masses and inversely proportional to the square of the distance between their centers. So even though the moon is only 1/400th the size of the sun, the moon has a stronger influence on the tides because it is closer to the Earth. The moon's influence on the Earth is about 2.25 times greater than the sun's (U.S. Army Corps of Engineers 1991). Given orbits of the moon around the Earth and the Earth around the sun, the interaction of lunar and solar effects on the tides varies daily, seasonally, and annually. The moon's orbit is elliptical, so once a month the moon is closest to the Earth (at perigee) and the tides will be higher than usual. Two weeks later it will be furthest away (at apogee) and tides will be lower. The Earth's orbit around the sun is also elliptical so that the Earth is closest to the sun around January 2 (perihelion) and furthest away around July 2 (aphelion) with corresponding effects on the tides. When the perigee and perihelion coincide, the highest tides occur.

Types of Tides

A tidal cycle consists of one high tide and one low tide. While recognizing that no two tide cycles are exactly alike, tides have been classified into three types: diurnal, semidiurnal, and mixed (Fig. 1). *Semidiurnal tides* occur when two tidal cycles take place each day, with high or low tides of more or less equal height. *Mixed tides* are characterized by two daily tides of distinctly unequal height: typically, a "higher high

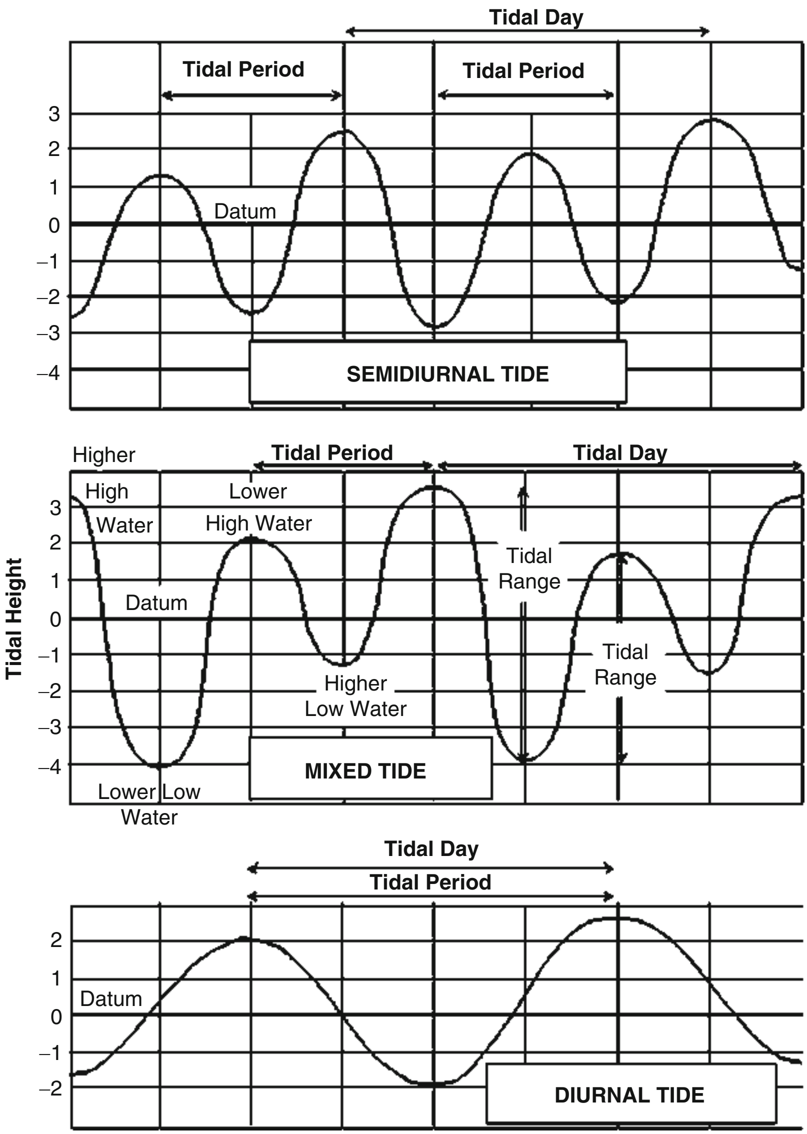

Fig. 1 Three main types of tides that occur over a lunar day– 24-h and 50 min, which is the time it takes for the moon to circle the Earth (Zevenbergen et al. 2004; US government publicaton)

tide" and "lower high tide" and a "higher low tide" and "lower low tide." Where only one tidal cycle occurs during a twenty-four-hour period, the tides are called *diurnal*.

Depending on the position of the moon relative to the sun, other tides occur at 2-week intervals. On new and full moons when the sun and moon are aligned in a straight line ("syzngy"), the highest astronomic tides – "spring tides" – are produced

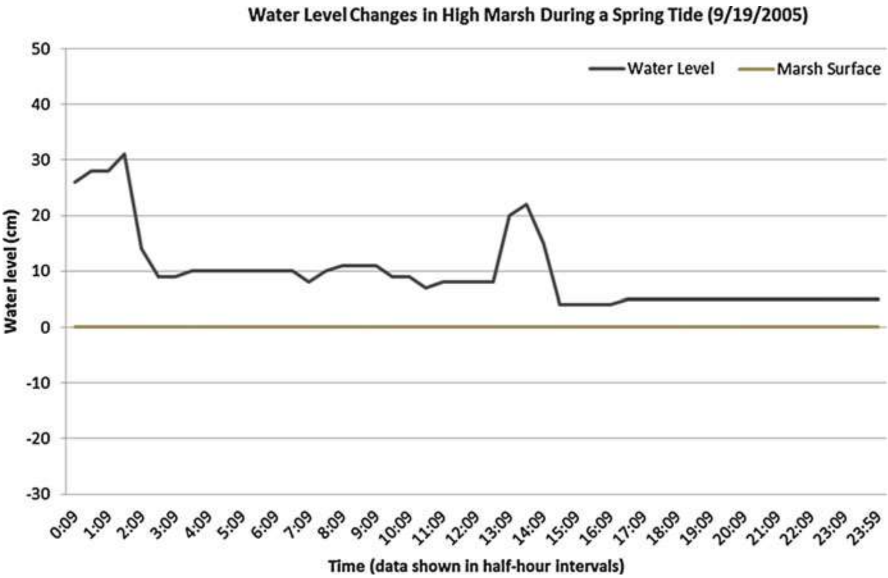

Fig. 2 Most if not all of the high marsh is flooded by spring tides. This site was flooded continuously during a tidal day (2 high and 2 low tides) (Source: Robert Vincent, used with permission). The marsh surface is at 0 cm (Source: Robert Vincent, used with permission)

by the combined gravitational pull of the moon and sun. These tides will flood the uppermost portions of salt marshes at high tide (Fig. 2) and expose the lowest levels of the intertidal zone (e.g., tidal flats) at low tide, resulting in the highest tidal range. About a week after the spring tides, the sun and moon are 90° apart in relation to the Earth (i.e., moon's first and third quarters) so their gravitational forces work against each other, minimizing the pull on the Earth's oceans. These conditions create moderated tides called "neap tides" and produce the lowest tidal range. These tides often do not flood the higher elevations of marshes, but they produce fluctuations in the water table of the high marsh (Fig. 3). The highest spring tides occur during full moons and the lowest neap tides happen during the first quarter after the new moon (R. Vincent, pers. comm. 2013). Neap tides can be at a relatively high level (relative to mean sea level) during third quarter after the full moon and inundate portions of the high marsh despite having a low tide range.

While the lunar cycle of the spring and neap tides repeats itself roughly every 30 days, it takes the moon 18.6 years to repeat its four phases at the same day of the solar year. At some point during this cycle (metonic cycle), the highest and lowest astronomic tides occur. The number of extreme high tides appears related to the declination of the moon (angle of the moon relative to the equator) over the 18.6-year period; about every 9.3 years the number of extreme high tides increases around the time of high-maximum lunar declination, whereas 9.3 years later or earlier, the lowest number of extreme high tides occur during the low-maximum phase (Wood 2001).

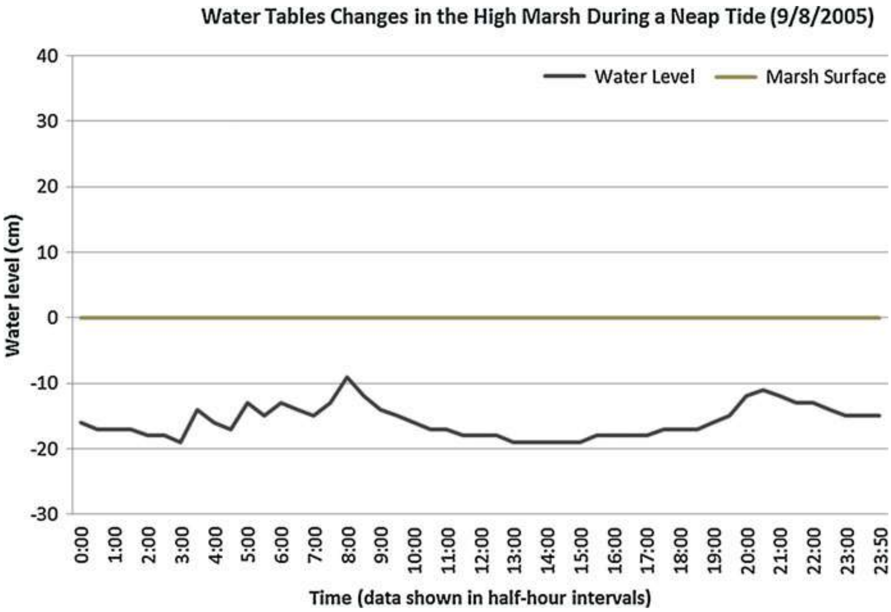

Fig. 3 Tidal signal in water table of high marsh during neap tides (Source: Robert Vincent, used with permission) The marsh surface is at 0 cm

Other Factors Affecting Tides and Coastal Wetland Water Levels

While charts of predicted tides show a repeating cycle of high and low tides, the amplitude of tides and the height of the tide at any given time and the duration of flooding of coastal wetlands are affected by other factors including the morphology of the estuary, river discharge, local weather conditions, and tsunamis (high waves created by earthquakes on the sea floor). The tidal effect in the marshes and swamps is further complicated by elevation, soil properties, and other factors, so the further one gets from the waterbody, the more other factors may influence site wetness in a tidal wetland.

Geomorphology

The behavior of the tides in an estuary is largely governed by the size and shape of the estuary and offshore bottom characteristics. As a result, some estuaries have higher tides further away from the sea, while others do not. Some estuaries have diurnal tides, while others nearby have semidiurnal tides.

The Bay of Fundy has a rare geomorphology that creates extraordinary tides that are among the highest tides in the world, reaching heights of >16 m (Parkes et al. 1999). The basin's funnel shape and gradual decrease in depth cause a great rise in tides from a 6 m tide at its mouth to a 16 m tide at its upper end. These huge tides are also the result of a so-called “bathtub effect” where water sloshes back and forth within this marine basin. This sloshing effect is nearly in sync with the lunar tides so these forces act in harmony to produce tides of enormous proportions. Storm tides can generate tides of 21 m or more. Peak tides occur at three cycles: 7 months, 4.53 years, and 18.03 years, with the latter generating the highest tides (Desplanque and Mossman 1998).

Local Weather and Climate

Local climatic conditions, especially winds and storms (e.g., hurricanes), influence the actual height, duration, and frequency of tidal flooding. Actual tides therefore may be different than the predicted tides. Storms generate the highest tides. High-pressure air masses (fair weather) lower sea level and tides, whereas low pressures raise sea level and tides. Wind effects may be most pronounced in microtidal estuaries (i.e., tidal range less than 2 m). Winds blowing onshore produce higher water levels (setup), while offshore winds generate the lowest water levels (setdown). In these estuaries, tide levels often do not exhibit a regular or predictable pattern. The wind effect on the tides is related to the orientation of the water body relative to the direction of the wind and wind strength and persistence. Strong winds may keep water in the estuaries for days and flood the coastal marshes for longer periods than one would expect from predicted tides (Fig. 4). Of course, hurricanes with over 160.9 km/h winds generate the highest tides (“storm surges”) in local areas, e.g., Katrina – up to 10 m along the Mississippi coast (Fritz et al. 2007). Storm surges move coastal waters inland for considerable distances; Katrina’s surge penetrated at least 10 km along the coast and up to 20 km in Mississippi bays and rivers. The longer the water remains upstream, the greater the ecological impact of salt water intrusion and prolonged inundation on coastal wetlands.

Wind tides and river discharge play a more significant role influencing wetland hydrology of Mississippi Delta marshes than do astronomical tides. These deltaic marshes are flooded at different frequencies than are typical salt marshes. Inland portions are inundated for an average of 16–27 h as much as 263 times per year, while streamside levee marshes (5.1–7.6 cm higher in elevation) are flooded only 160 times for an average duration of 6.6 h (Gosselink 1984). These marshes are flooded for 50% and 12% of the year, respectively, and for more than 80% of the time in September and October.

River discharge significantly affects water levels and salinity patterns in estuaries. Heavy runoff from contributing watersheds can raise water levels in estuaries and move the salt wedge seaward while droughts and low discharge have the opposite effect and introduce salt water into tidal freshwater wetlands. High discharge occurs in spring during peak runoff and in late summer during the hurricane season. These discharges temporarily reduce the daily tidal range in local areas as their flows overwhelm the tides. Peak discharges by the world’s major rivers like the Mississippi

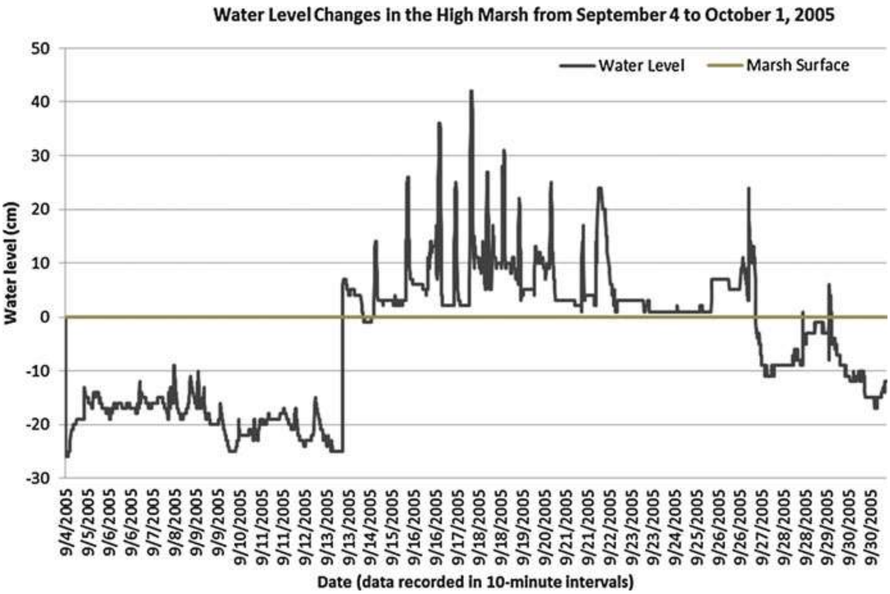

Fig. 4 Water levels in the high marsh are affected by astronomic tides modified by local weather conditions and other factors. This hydrograph shows changes in water levels in a Massachusetts high salt marsh during the month of September 2005. Notice prolonged inundation from 9/13 to 9/27. The plant community was dominated by *Spartina patens* and *Juncus gerardii* (Source: Robert Vincent, used with permission)

River may eliminate tidal influence along its waterway. Precipitation brought about by hurricanes can quickly raise water levels especially in coastal rivers and in estuaries with low tidal ranges that receive freshwater inflow from major river basins. The combined effect of three hurricanes (Dennis, Floyd, and Irene) in September and October 1999 raised river levels significantly, causing almost 2 months of flooding in most of eastern North Carolina (Bales et al. 2000). In areas of Mediterranean climates, winter precipitation may contribute to higher tidal levels during those times.

Global climatological events related to warming and cooling of the tropical Pacific Ocean (El Niño/La Niña-Southern Oscillation, ENSO) also affect the hydrology of coastal wetlands in some places. Pacific warming (El Niño) brings more precipitation to central-southern California, the Southwest, and the Southeast, thereby increasing river discharge and marsh flooding. The cooling event (La Niña) produces drier conditions in the Southwest and Southeast with less frequent and shorter duration flooding of their coastal wetlands (Childers et al. 1990). Interestingly, La Niña brings wetter conditions and higher water levels to the Pacific Northwest. These warming or cooling periods may last for 7–9 months or more and occur on average every 5 years.

Changes in Sea Level

Seasonal and annual shifts in sea level also play a role in tidal wetland flooding. The lowest levels occur in winter and highest levels in summer or early fall due to a combination of factors including thermal heating (decreases water density thereby increasing water volume), precipitation, and increased discharge of freshwater (spring to summer). Monthly sea levels also experience interannual variability.

Tidal Influence in Freshwater Rivers and Their Wetlands

In coastal rivers with large contributing watersheds and high discharge, eventually an upstream location is reached where the water is strictly fresh, with no trace of ocean salts, yet the water levels still fluctuate with the tides. This point marks the beginning of the zone where tidal freshwater wetlands are found. Flood tides cause water levels to rise in these areas. A directional change in the water flow occurs in the lower freshwater tidal areas with water moving upstream during the flood tide and downstream during ebb tide. This area is called the "point of tide reversal." Tidal influence, however, extends farther upstream, where downstream flow continues while water levels temporarily rise due to high tide conditions downstream. Upstream of this area where the water levels are not affected by the tides marks the beginning of the nontidal reach of the river. The National Ocean Service considers the point where the mean tide range is less than 6 cm to be the "head of tide" (Hicks et al. 2000).

Flooding of tidal swamps may be more complicated in the lower reaches (closer to the estuary) since their topography often consists of a mosaic of hummocks (mounds) and hollows (depressions). In contrast, tidal swamps near the upper limits of tidal influence typically have a relatively flat floodplain form. The hollows are frequently inundated as much as once or twice daily and when not flooded remain saturated near the surface, largely due to the short duration between inundations (Rheinhardt 2007). As expected, hummocks are flooded less often, by spring tides and other high-water events, but have water tables close to the surface. The upstream floodplain tidal swamps are flooded by spring freshets, spring tides, and storm surges. Peak river discharges – spring freshets – may eliminate any tidal signal in the water levels and inundate the wetlands for extended periods. Water levels in some swamps can be higher than that of adjacent rivers due to rainfall, residual floodwater, or groundwater contributions (Light et al. 2002). For a detailed description of the hydrology of southern tidal swamps see Day et al. (2007).

Soil Saturation

Water brought in by tides moves both as "sheet flow" across the wetland surface after the tide overtops creek or ditch banks and as "subsurface flow" through the soil saturating aerated portions of the soil. The position of the water table and duration of saturation vary due to many factors including tide stage, duration between flooding events, soil type, topography, geomorphology, groundwater interactions, evapotranspiration, precipitation patterns, and distance from tidal creeks. In Bay of Fundy macrotidal marshes, many of these elements have been reported to be more important than tide height as drivers of subsurface hydrology (Byers and Chmura 2012).

Tides have a significant effect on subsurface hydrology of marshes near the creek banks. Interstitial water (water stored in marsh soils during high tide) seeps from creek banks and drains via tidal creeks at low tide (Jordan and Correll 1985). This soil water is replaced by incoming tides and then lost again with ebb tide, resulting in a fluctuating water table in the high marsh, especially areas in close proximity to the creek banks. This type of seepage ceases during high tide when the marsh is submerged and hydraulic pressure is equal between the creek banks and marsh.

Soil saturation of nonflooded portions of the marsh is affected by the tides to some degree. In a Massachusetts salt marsh, the tidal signal from the semidiurnal tides was observed as water fluctuation in the water table within 2.5–15.0 m of a tidal creek; howevever, beyond 15 m, there was no horizontal movement of water (Nuttle 1988). Similar results were found in a New York brackish marsh where the high tide signal could still be detected in the water table up to 12 m from the creek (Montalto et al. 2006). The water table remained high in the interior marsh longer than in the creekside marsh. At a distance of 6 m from the creek, the soil was saturated to within 20 cm of the surface more than 90% of the time during a lunar month, whereas the water table of the marsh closer to the creek had a water table at this depth for only half of the time. Beyond 24 m from the creek, the interior marsh was saturated to the surface for more than 90% of the time.

Groundwater Interactions

Groundwater may enter coastal wetlands from unconfined aquifers through the marsh substrate or tidal creeks that breech the confining layer. Along the upland border, groundwater discharging to the surface may create seepage areas where freshwater species may occur. Groundwater discharge may also affect salinities further into the marsh. In a South Carolina marsh, less saline conditions (5–10 ppt) were found in areas receiving considerable groundwater (Gardner et al. 2002). Vegetation shifted from glassworts (*Salicornia* spp.) in the neighboring saline areas (25–50 ppt) to black needlerush (*Juncus roemerianus*) in the groundwater-affected marsh.

Wetland Zonation Driven by Hydrology

Tidal wetlands are intertidal habitats, which are alternately flooded and exposed to air. Given the variable levels of flooding by daily, spring, neap, and storm tides, the elevation within the wetland largely determines the frequency and duration of flooding. Several "water regimes" have been used to describe the hydrologic zones of tidal wetlands for classification and inventory purposes (Cowardin et al. 1979). For marine and estuarine wetlands where tides are the main source of water, three water regimes are applied including irregularly exposed, regularly flooded, and irregularly flooded (Fig. 5). The irregularly exposed zone is the area that is nearly permanently flooded but exposed during spring low tides (from mean low water – MLW – to extreme low water of spring tides). The irregularly exposed zone represents the lowest portion of tidal flats. The regularly flooded zone is typically flooded at least once daily by the tide. It extends from MLW to the mean high tide mark (MHW) and is occupied by the low marsh and the majority of tidal flats. The irregularly flooded zone is flooded less than once per day and occurs from MHW to extreme high water of spring tides. This zone has the most varied hydrology ranging from almost daily flooding to areas flooded only by spring or storm tides. This zone is the high marsh where a diversity of plant communities may form depending largely on variations in hydrologic and salinity conditions (Table 1). Depressions in the high marsh are flooded by spring and storm tides and water will persist for variable periods, with pools being inundated for the longest periods and pannes for shorter periods. In semiarid, arid, and tropical areas, barren areas ("salinas") may form within the irregularly flooded zone. Along the US coast, salt marshes are mostly irregularly flooded (high marsh) types, yet regularly flooded types predominate the marshes behind the sea islands of South Carolina and Georgia and in coastal Louisiana where significant subsidence is occurring.

For tidal freshwater wetlands, overbank flow from rivers plays a significant role in their hydrology. While some of these wetlands are regularly flooded like their estuarine counterparts, the focus of others is on river overflow. The duration of flooding in tidal freshwater wetlands depends on both tidal and inland water influences and are described as: semipermanently flooded-tidal (flooded throughout

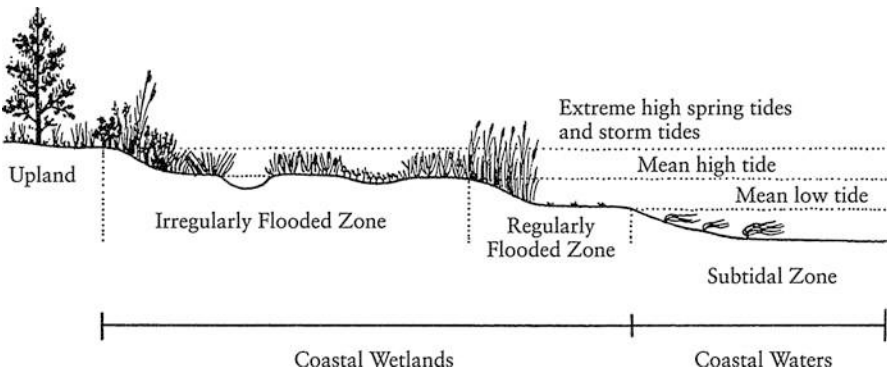

Fig. 5 Two main zones of coastal wetlands created by flooding frequency: regularly flooded and irregularly flooded. The irregularly exposed zone (not shown) occurs between the regularly flooded zone and subtidal zone where spring tides expose shallow bottoms but can be exposed by strong offshore winds (Copyright: Tiner 2013)

Table 1 Flood frequency and salt marsh plant communities (Sources: RI – Bertness and Ellison 1987; GA – Antlfinger and Dunn 1979; CA – Cahoon et al. 1996)

Plant community	State	Average days/month subject to flooding
Spartina alterniflora – tall form	RI	30.5
S. alterniflora – short form	RI	23.5
Spartina patens	RI	17.3
Juncus gerardii	RI	8.7
S. alterniflora – tall form	GA	24.4–30.5
S. alterniflora – short form	GA	12.2–24.4
Barren zone	GA	1.5–3.1
Salicornia spp. – *Batis maritima*	GA	1.2–2.4
Juncus roemerianus – *Borrichia frutescens*	GA	0.6–1.5
Spartina foliosa	CA	30.5
Salicornia depressa	CA	5.0–6.0

the growing season in most years), seasonally flooded-tidal (flooded for extended periods during the growing season), and temporarily flooded-tidal (flooded for brief periods usually a week or less during the growing season) (Cowardin et al. 1979).

Impacts to Coastal Wetland Hydrology

While many tidal wetlands are freely connected to the ocean or estuaries, the hydrology of many others has been altered either directly or indirectly. Direct alterations involve onsite impacts including ditching, diking, and filling, whereas indirect effects are produced by activities offsite (see Tiner 2013 for details). Many tidal marshes have been ditched, first to improve drainage for salt hay farming or livestock usage, then later for mosquito control. Building causeways across marshes for roads and railroads has restricted both sheet flow and tidal inflow to the marsh by the installation of culverts, tide gates, or narrow bridges, which may reduce the volume of water entering the wetland complex. Impoundments have been constructed in tidal wetlands for waterfowl management, mosquito control, or other purposes. Water-control structures (e.g., tide gates or weirs) are installed to regulate water levels within these diked wetlands. Tidal wetlands have been fragmented by development (e.g., harbor facilities, airports, residential and commercial real estate), which in addition to reducing marsh extent has altered the quality of many wetlands through disrupting tidal flows and pollution. Offsite alterations of estuarine hydrology have significant impacts on tidal wetlands. For example, deepening of rivers for navigation has increased the volume of water entering estuaries, elevated tidal ranges, heightened tidal current velocities, and allowed salt water to migrate further upstream. These changes have increased bank erosion and impacted wetland plants and animals. Freshwater diversion projects including heavy use of fresh water for industrial purposes and damming rivers for water supplies have had similar impacts. Coastal wetland restoration efforts typically focus on restoring hydrology because many of the problems they seek to address (e.g., invasive species, restricted access to marshes by fish and aquatic invertebrates) are the result of altered hydrology.

Future Challenges

Rising sea-levels will undoubtedly have an enormous impact on coastal wetlands depending on if the marshes can accrete their marsh platform to support plants. Increased air temperature and carbon dioxide availability and changes in precipitation patterns will also affect vegetation growth. Much has been written on the predicted fate of tidal and other wetlands in response to climate change (see Cahoon et al. 2006; Lovelock and Ellison 2007; Kirwan et al. 2010; Parker et al. 2011). Recent observations have already witnessed changes from low marsh to tidal flats or subtidal bottoms, high marsh to low marsh, and salt marsh migration into low-lying forests as a result of changes in the frequency and duration of tidal flooding. Similar changes in mangroves are expected.

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Hydrologic Modeling of Wetlands

29

Elias Getahun and Misganaw Demissie

Contents

Abstract

This chapter provides a brief compilation of reference works on hydrologic modeling of wetlands. Hydrology, which is the study of water circulation and its constituents through a water cycle on or near the land surface, is the primary driving force determining the structure and functions of wetlands. Developing a robust wetland model requires understanding the main hydrologic processes occurring in its drainage area. Succinct discussions of hydrologic model types, simulation models for wetland hydrology, and key modeling considerations are presented here.

Keywords

Hydrology · Wetland water budget · Hydrologic modeling

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Definitions

Definition: Riverine Wetland Hydrology

Author: Beth Middleton

Riverine wetland hydrology is characterized and driven by seasonal changes in precipitation, snow melt, groundwater input/output, and evapotranspiration. River hydrology also reflects geomorphology (e.g., depth to groundwater/bedrock and constraints by channel walls). Rivers in more arid parts of the world tend to have "flashier" flow during rainy season, but little streamflow during the drier season of the year.

Porsuk River near Eskisehir, Turkey, which is a semiarid river lined by the tree, *Populus alba*. Pictured is Harun Böcük. Photo by Beth Middleton

Definition: Water Monitoring

Author: Beth Middleton

(*continued*)

Water monitoring is the measurement of water level changes in wetlands using instrumentation such as water gages. In conjunction with a nearby water gage, a Sediment Elevation Table (SET; pictured) is useful to construct a simple hydrograph of a research site. During major flooding, regional sites at the same elevation are inundated to similar depths by the floodwater sheet. This idea can be used to tie recorded water depth fluctuation at water gages near SETs in research sites. A SET establishes a stationary monument, which consists of a series of connected rods driven into the ground to the point of refusal.

Construction of a Sediment Elevation Table, which is useful for understanding elevation and hydrologic changes in wetlands. Pictured are Guodong Wang and Evelyn Anemaet. Photo by Beth Middleton

Introduction

Hydrology is a science concerned with the circulation of water and its constituents through a water cycle, including precipitation, evaporation, runoff, infiltration, groundwater flow, streamflow, and the transport of substances dissolved or suspended in the flowing water (Maidment 1992). It specifically deals with water on or near the land surface. In wetlands, hydrology is the primary driving force, affecting their structure and functions. The existence of wetlands is governed by a continuous or recurrent presence of surface or subsurface water, occupying transition zones between predominantly wet and dry environments and thereby sharing aspects of both aquatic and terrestrial ecosystems (USEPA 2008). Wetland restoration or creation efforts are attempting to mimic this necessary hydrologic environment for the desired wetland type (Demissie et al. 1997).

Wetlands can be broadly categorized into two classes based on topographical location in the landscape and hydrologic factors: nontidal or inland wetlands (e.g., riverine wetlands) and tidal or coastal wetlands (e.g., fringe wetlands). The focus of this chapter is on modeling the hydrology of nontidal wetlands, which requires understanding hydrologic processes occurring in the wetland drainage area as well. Its main objective is to provide a concise compilation of reference works on hydrologic modeling of wetlands. In doing so, types of hydrologic models, models used for simulation of wetland hydrology and important modeling considerations are succinctly discussed.

Modeling of Wetland Hydrology

Hydrology is one of the most important features used in defining wetland structure and function, in addition to soil and vegetation. Inland wetlands predominantly occur in regions where precipitation is in excess of evapotranspiration, exhibiting seasonal fluctuation of water surface levels dependent on weather and antecedent moisture conditions. Based on the recurrence of flooding, wetlands could be intermittently, seasonally, or permanently flooded (William and Gosselink 2000). The hydrologic water budget of a wetland comprises wetland storage, inflows and outflows, and mathematically can be expressed as

$$\frac{\Delta S}{\Delta t} = I - O\ ;\ \text{where } I = P + SR + GD \quad \text{and} \quad O = ET + SO + GR$$

Where I is defined as inflows to the wetland such as precipitation (P), surface runoff (SR), and groundwater discharge (GD); O is defined as outflows from the wetlands including evapotranspiration (ET), surface outflow (SO), and groundwater recharge (GO); and ΔS is defined as water storage in the wetland at a given time (Δt). Figure 1 illustrates the major components of the wetland water budget, including precipitation, surface water inflows and outflows, groundwater, and evapotranspiration. The precipitation pattern varies with climate regions where there exists distinct periods of dry and wet seasons. Surface water inflows and outflows can be channelized or overland flows, and often match with precipitation patterns with the exception of spring thaw conditions. Evapotranspiration depends on the meteorological, physical, and biological conditions in the wetlands and has a seasonal pattern with highs in summer and lows in winter (William and Gosselink 2000).

Understanding wetland hydrology requires gathering all data that are important to describe components of the hydrologic water budget as accurately as possible. Precipitation including both rainfall and snow is usually a point measurement from which areal coverage is estimated. Point measurements are subject to errors as a result of

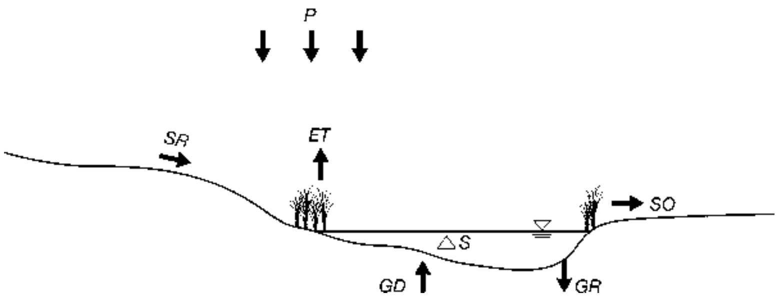

Fig. 1 Components of wetland water budget with inflows including precipitation (*P*), surface runoff (*SR*), and groundwater discharge (*GD*) outflows including evapotranspiration (*ET*), surface outflow (*SO*), and groundwater recharge(*GO*). ΔS is water storage in the wetland at a given time

wind eddies created by the gage and estimated areal precipitation values could introduce additional uncertainties. Surface inflows can be estimated as a function of hydrologic processes in the wetland drainage area, including surface and subsurface runoffs. Appropriate characterization of the wetland drainage area is required for accurate runoff simulations. Surface outflows may be estimated as a function of wetland storage levels and thus requires adequate collection of data to establish wetland storage-outflow relationships. Groundwater inflows or outflows depend on the relative water level in the wetland and the surrounding groundwater table. Groundwater flows into a discharge wetland when the surrounding water table is higher and the reverse is true for a recharge wetland. Data collection and analysis are crucial steps for accurate representation and thereby modeling of wetland hydrology.

Important Modeling Considerations

Ideally, modeling of wetland hydrology should include the ability to simulate precipitation, surface runoff, infiltration, groundwater flow, wetland storage, outflows, and routing overland flow, streamflow, and subsurface flow (USACE 1988). The importance of hydrologic simulation components may vary depending on the purpose of the modeling. When the focus of interest is in the wetland hydrologic budget and functions, it is necessary to perform detailed modeling of all hydrologic components. In contrast, hydrologic processes such as evapotranspiration, infiltration, and groundwater flow may not be significant in the model during flood simulations. Therefore, identifying important component processes is central to effective modeling exercises. For example, in their modeling effort to evaluate the cumulative hydrologic impacts of wetlands in watersheds, Demissie et al. (1997) first identified important component processes to be considered, which included interception, evapotranspiration, infiltration, drainage, overland flow, groundwater flow, and channel routing.

Before selecting a model to simulate wetland hydrology, understanding its characteristic features such as model structure and process components, spatial and temporal representation, and application extent are all very important. A modular programming structure consisting of components for subprocesses, data input and response output facilitates simpler model application, potential coupling and/or modification without affecting the model structure. A model with distributed-parameter representation provides the capability to account for spatially varied hydrologic responses with in a watershed. The ability to perform continuous model simulation is desirable as it enables simulating changes in soil moisture storage. Such capabilities could also be critical in assessing water availability for wetland recharges during low-flow conditions and estimating initial soil moisture conditions in cases of single-event simulations. A model's application extent should include the possibility of simulating watersheds with varying landscapes such as farmland and forest where wetlands mostly occur, and also urban areas. Finally, detailed model documentation describing process components and program structure is necessary for an effective modeling exercise.

Mathematical Hydrologic Model Types

Types of hydrologic models can be distinguished by the way in which the component processes are modeled. Major classification of mathematical hydrologic models generally includes four categories: deterministic or stochastic models; empirical, physically based, or conceptual models; lumped-parameter or distributed-parameter models; and continuous-time or event-based models. A deterministic model entails defined initial conditions that are parameterized while excluding randomness. In contrast, a stochastic model accounts for randomness resulting from uncertainties in model parameters. Based on a representation of concerned physical processes, a model can be classified as "empirical" when it is derived from experimentation or dependent correlations of observed hydrologic outputs to inputs, "physically based" when it is based on governing laws of physics including conservation of mass, momentum, and energy, and "conceptual" when it represents the hydrologic system in simplified conceptual elements. A hydrologic model can be categorized as "lumped" when it involves spatially aggregating parameters of the hydrologic system to create uniformity and as distributed when it accounts for spatial and temporal variability of both input and output of hydrologic variables. Hydrologic models can also be classified as "event-based" if they are designed to simulate a single event from a couple of minutes to days, or as "continuous-time" models if they are capable of performing long-term simulations.

Hydrologic Models Used for Wetland Simulation

A number of hydrologic models have been developed and applied to simulate wetland hydrology. Comprehensive reviews of mathematical hydrologic models as applied to the simulation of wetland hydrology were discussed by several researchers, providing insights into model strengths and weakness in terms of simulating hydrologic processes and functions (Bengston and Padmanabhan 1999; Demissie et al. 1997). An overview of selected hydrologic models that were used for simulation of wetland hydrology is provided here.

The Iowa State University Hydrologic Watershed Model (ISUHM; Haan and Johnson 1968), the Minnesota Model for Depressional Watersheds (MMDW; Moore and Larson 1979), and DRAINMOD (Skaggs 1980) were among the first of these models that were specifically developed to analyze wetlands and agricultural drainage. These physically based models are capable of simulating hydrologic processes including precipitation, surface runoff, evapotranspiration, infiltration, and streamflows. Depressional storage and routing are used in ISUHM and MMDW to simulate the effect of wetlands. In contrast, DRAINMOD simulates wetland effects by filling available depression storage volume before initiating surface runoff and DRAINMOD also allows seepage of water stored in the depression. ISUHM, an event-based model, was later modified to enable continuous simulation (Campbell and Johnson 1975). DRAINMOD was also upgraded to perform watershed modeling by incorporating an unsteady-flow, channel-routing procedure based on the dynamic wave method (Broadhead and Skaggs 1984).

Walton et al. (1996) developed the Wetlands Dynamic Water Budget Model to evaluate hydrologic and hydraulic processes in various types of wetlands. The model consists of modules for simulating surface water, ponding and infiltration, and ground water processes. Application of the model to Cache River wetlands in Arkansas confirmed that backwater effects of downstream constrictions resulted in an inundation of upstream wetlands.

In an effort to simulate the effect of wetlands on flooding, Ogawa and Male (1986) applied HEC-1 (Feldman 1995) and employed its reservoir routing option, representing wetlands as reservoirs. Nichols and Timpe (1985) applied Hydrologic Simulation Program-Fortran (HSPF; Donigan et al. 1995) to simulate wetland hydrology in their study of phosphorus dynamics in floodplain wetlands in South Florida. The HSPF model was modified to better simulate wetlands by including a flow routing algorithm based on depth-discharge relationships for the wetlands. Arnold et al. (2001) developed a water budget model for wetland simulation, incorporating it into the hydrologic model, Soil and Water Assessment Tool (SWAT; Arnold et al. 1998). Modification to SWAT model allowed effecting limited interaction of water stored in the wetland with the soil profile and the shallow aquifer. Bekele et al. (2011) applied SWAT to simulate hydrologic and water quality functions of constructed wetlands in an attempt to identify optimal wetland locations in the tributary watersheds of Mackinaw River in Illinois that are predominantly agricultural. In that study, the constructed wetlands were considered as best management practices for controlling nonpoint source pollution.

Sun et al. (1998) developed FLATWOODS model to simulate the hydrology of cypress wetland-pine upland landscape by coupling a 2-D groundwater model, a surface flow model based on a variable source hydrology, an evapotranspiration model, and an unsaturated water flow model. The model is capable of simulating daily groundwater table depth, evapotranspiration, and soil moisture content distributions in a watershed and it is applied to pine flatwoods research watersheds in northern Florida. The FLATWOODS model may be used to predict hydrologic impacts of various forest management practices in coastal regions. The FLATWOOD model, however, is limited in its description surface water-groundwater interaction because of its large spatial scales and model structure (Mansell et al. 2000).

WETLANDS (Mansell et al. 2000) is a model developed to simulate wetland hydrology, and it is a modified version of USGS's Variably Saturated Two-Dimensional Transport model (VS2DT; Healy 1990). Major modifications to VS2DT include incorporating capabilities to simulate a surface pond with seasonally dynamic water levels over inundated areas, estimate evapotranspiration using limited data, and simulate surface water-ground water interaction. The modified model was used to successfully simulate local hydrology for a cypress pond located in a coastal plain, pine forest landscape, describing temporal patterns of daily pond water and groundwater table elevations. Even in cases in which available empirical data for a given site are minimal, WETLANDS has the potential to be utilized as a predictive tool for wetland hydrology provided that appropriate assumptions were made (Mansell et al. 2000).

Coupling components from various models and/or modification of process routines within a model have become commonplace when a single suitable model is unavailable. In this regard, Yergeau (2010) developed a hydrologic model, which coupled the Storm Water Management Model (SWMM; Rossman 2005) with MODFLOW (McDonald and Harbaugh 1988) to simulate the hydrologic water budget for Kearny Marsh to understand the dynamics of urban wetland hydrology for potential restoration efforts. The coupling was done through an exchange of evapotranspiration and infiltration data between the models. Thompson et al. (2004) developed a coupled hydrologic-hydraulic model using MIKE SHE (Refsgaard and Storm 1995) and MIKE 11 (Havnø et al. 1995) modeling system for lowland wet grassland, the Elmley Marshes, in southeast England. The marshes have characteristic features of wetlands commonly found in Europe with a complex ditch network and water control structures. The coupled model was able to simulate seasonal fluctuation of groundwater and ditch water levels, and it could be used to explore the impact of alternative water level management (Thompson et al. 2004). Demissie et al. (1997) developed an integrated model for evaluating cumulative hydrologic effect of wetlands in Cypress Creek watershed in the Cache River basin in southern Illinois. This wetland hydrologic model was built on Areal Nonpoint Source Watershed Environment Response Simulation (ANSWERS; Beasley and Huggins 1982), which is a physically based, distributed-parameter, event-based hydrologic model, by incorporating dynamic wave channel-routing component.

Challenges

Hydrologic modeling of wetlands involves the process of selecting and/or developing models that are appropriate for a particular application of interest. This process should take into account the tradeoff between model complexity and accuracy. After reviewing complexity, accuracy, and effectiveness of mathematical models applied to freshwater wetlands, Costanza and Sklar (1985) concluded that there is an optimum model complexity beyond which the benefits of additional articulation are outweighed by declining accuracy. Therefore, the challenge for modelers is to strike a balance between model complexity and accuracy in developing and/or selecting the right model for simulation of wetland hydrology.

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Hydrologic and Treatment Performance of Constructed Wetlands: The Everglades Stormwater Treatment Areas

30

Wossenu Abtew, Tracey Piccone, Kathleen Pietro, and Shi Kui Xue

Contents

Abstract

Constructed wetlands are now widely used around the world for water treatment. The Everglades Stormwater Treatment Areas (STAs) are currently the largest constructed wetlands and are reducing Total Phosphorus (TP) concentrations in agricultural and urban runoff that flows into the Everglades. Average inflow flow-weighted TP concentration is 140 ppb and an average of 73 percent removal efficiency is observed from all the STAs. Total area of the STAs is 27,530 ha and target water depth is 38-46 cm.

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Keywords
Constructed wetlands · Everglades · Runoff treatment · Stormwater treatment areas · Phosphorus removal

Definitions
Definition: constructed wetland

Author: Beth Middleton

A constructed wetland is an artificial wetland built for a specific purpose, e.g., a wetland created to improve the water quality of sewage water. Such wetlands are often planted with emergent species such as *Typha* spp. or *Phragmites australis*.

Introduction

Constructed freshwater wetlands are being used to reduce phosphorus (P) concentrations in surface waters that flow into the Everglades, a 900,000 ha subtropical freshwater marsh located in the southern end of Florida, USA. (Fig. 1).

Emergent species are often planted in constructed wetlands. Photo by Beth Middleton

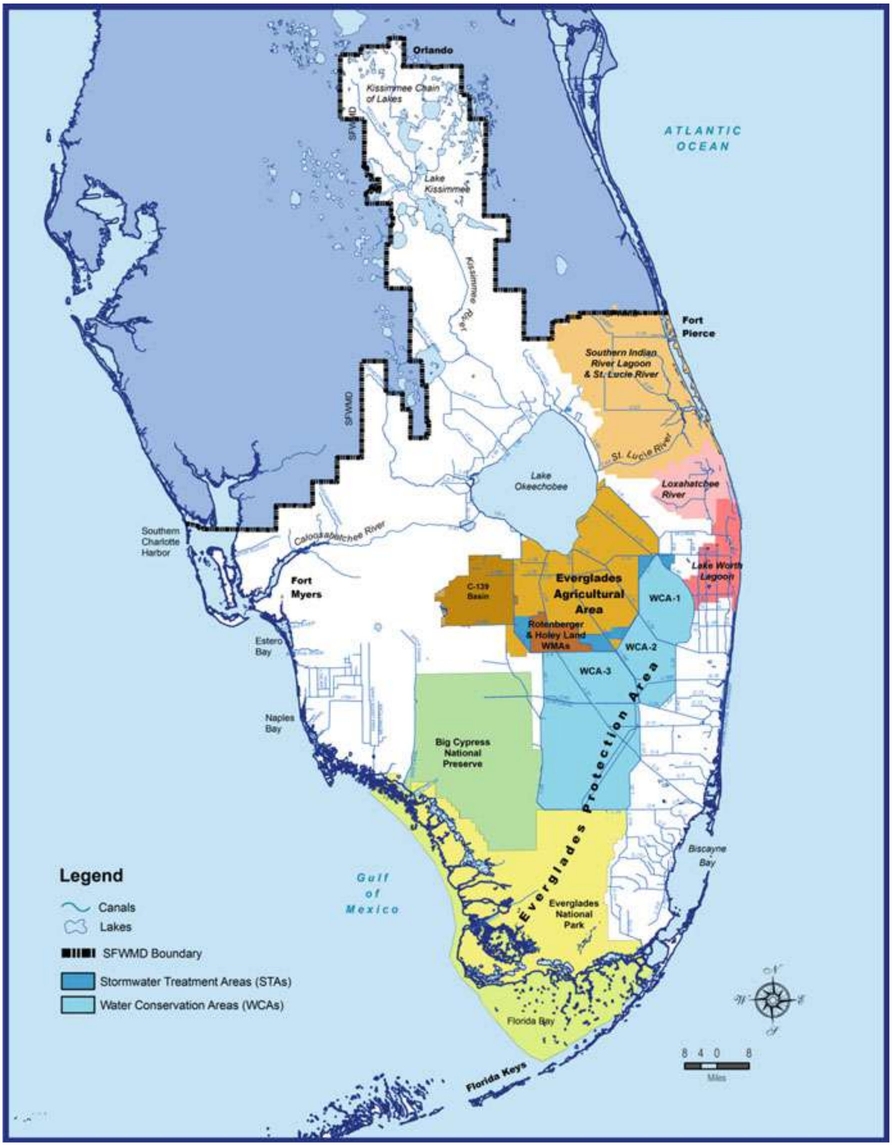


Fig. 1 The Everglades Agricultural Area, Stormwater Treatment Areas, and the Everglades Protection Area

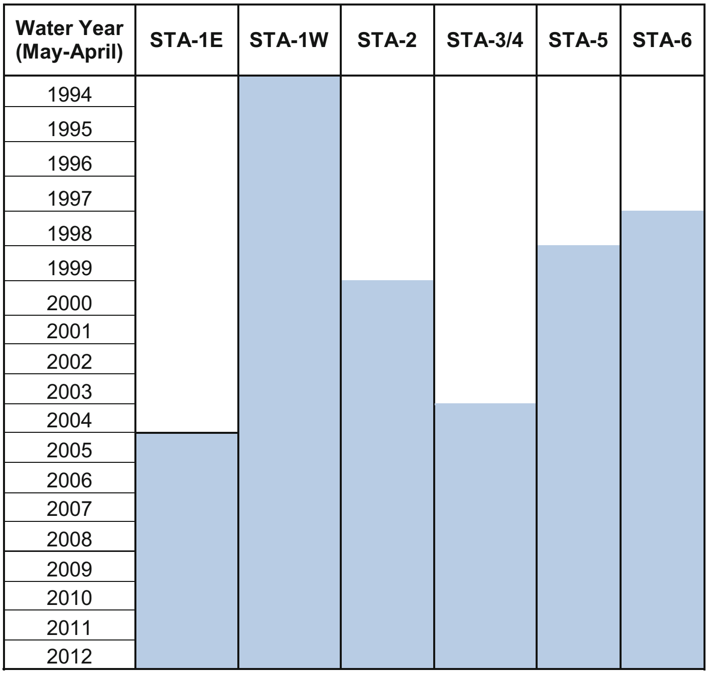
The Everglades ecosystem has been impacted by both anthropogenic and natural factors through the years. Observed changes in flora and fauna are attributed to increase in nutrients in inflow waters and hydroperiod changes (Davis 1991; Koch and Reddy 1992; Noe et al. 2001; Ramesh and DeLaune 2008). In the twentieth century, the northern portions of the Everglades swamps were drained with an

Fig. 2 Stormwater Treatment Areas, STA-1E, 1-W, 2, 3/4, 5, and 6 showing emergent aquatic vegetation (EAV) and submerged aquatic vegetation (SAV) cells

intricate canal network and pumps, to create a very productive agricultural basin, called the Everglades Agricultural Area (EAA). Currently, the EAA is a 224,000 ha irrigation/drainage basin upstream of the remaining Everglades (Fig. 2). The main crop in the EAA is sugarcane as well as vegetables, rice, and sod. The EAA basin currently has a lower elevation than the surrounding areas due to subsidence of muck (organic) soil as a result of drainage, with a loss of as much as 1.7 m in 80 years (1924–2003; Snyder 2004). Irrigation water to the EAA is supplied from Lake Okeechobee. Excess rainfall is drained from the agricultural fields as runoff/drainage and is mostly pumped to the south, with agriculturally influenced water quality with a high nutrient content. For the past two decades, the Everglades Stormwater Treatment Areas (STAs) (Fig. 2) in combination with agricultural Best Management Practices (BMPs) have performed an important role in improving water quality in discharges to the Everglades (Baker et al. 2012).

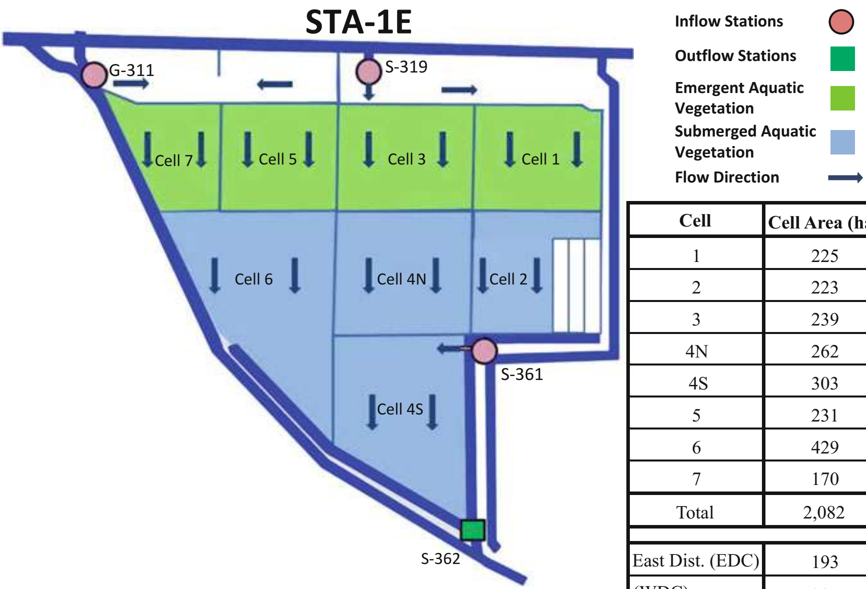
Currently, six constructed wetlands with a total area of 27,530 ha are operational. The objective of this chapter is to present hydrologic and phosphorus removal performance of these Everglades STAs.

Table 1 Operational timeline for the Everglades STAs



The Everglades Stormwater Treatment Areas

Natural wetlands have been used to treat water for at least a century. Construction of wetlands for water treatment started in the 1950s. Constructed wetlands are now widely used around the world to remove pollutants from water (Kadlec and Knight 1996; Kadlec and Wallace 2009), including the construction of the Everglades STAs in the past two decades. These STAs with a total area of 27,530 ha are currently the largest manmade wetlands in the world. These wetlands are designed and operated to discharge low TP concentrations, originally designed for 0.05 mg L^{-1} outflow TP concentration and currently achieving lower concentrations. The first of these six STAs started operation in 1994, and the others have come on line since then (Table 1). Each STA is compartmentalized into cells and in most cases the cells are aligned serially to form treatment paths called flow-ways. The six original STAs, STA-1E, STA-1 W, STA-2, STA-3/4, STA-5, and STA-6, are shown in their current states in Figs. 3, 4, 5, 6, and 7. (Note that STA-5 and STA-6 have recently been combined, but their performance records to date are separated for reporting). The cell



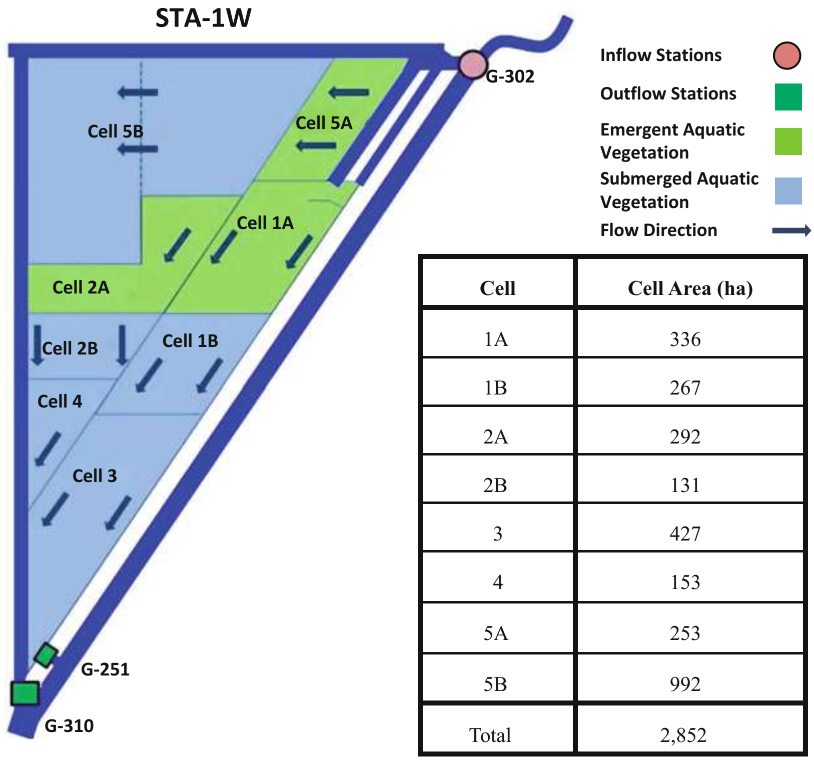
Cell	Cell Area (ha)
1	225
2	223
3	239
4N	262
4S	303
5	231
6	429
7	170
Total	2,082
East Dist. (EDC)	193
(WDC)	235

Fig. 3 Stormwater Treatment Area 1 East (STA-1E) with cells, flow-ways, water control structures, and vegetation type

areas shown on the STA maps were calculated from levee centerlines and do not include inflow canals. The area of most of the STAs was increased in later years. The hydraulics of the STAs commonly require supply canals, inflow and outflow pumps, flow distribution canals, divider levees, gated culverts, and flow collection canals. The vegetation type of each cell is related to its position along the treatment path (Figs. 3, 4, 5, 6, and 7).

The cells are populated with a mixture of plant species. In general, upstream cells which receive influent water with high TP are dominated by emergent aquatic vegetation (EAV) species. Downstream cells are usually dominated by submerged aquatic vegetation (SAV), which provide further TP reduction. The common vegetation species in the STAs are listed in Tables 2, 3, and 4. Vegetation varies among the cells and each cell undergoes temporal changes in vegetation coverage and composition. Figure 8 depicts vegetation in four STA cells to show variations in species coverage.

From 1996 to 2012, these constructed wetlands retained approximately 1,560 metric tons (mt) of TP, reducing inflow loads by 73%. Over this period, the inflow flow-weighted TP concentration of 140 parts per billion (ppb) was reduced to an outflow flow-weighted concentration of 37 ppb (Table 5). The TP removal rate ranges from 50% to 90% varying over the period of operation from one STA to another and from year-to-year in each STA (Pietro 2012). Sources of variation include STA site characteristics such as soils, vegetation, topography, hydraulic



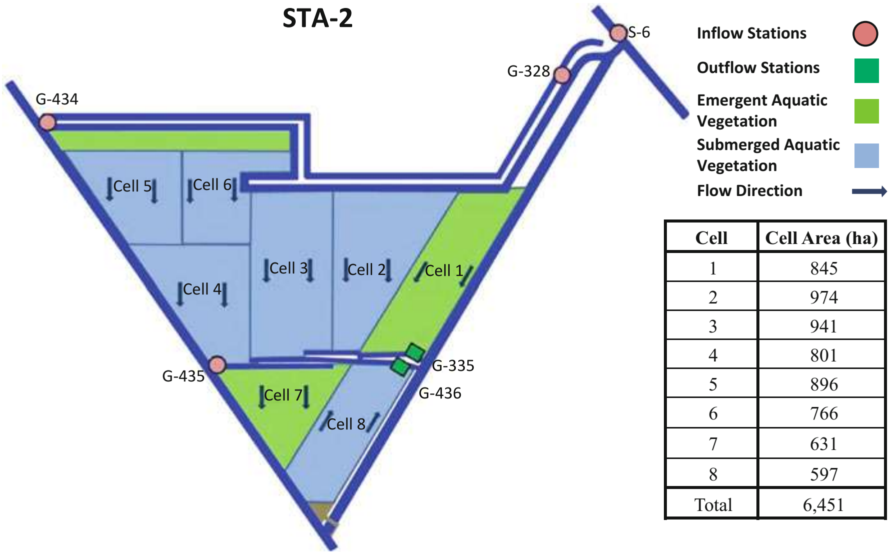
Cell	Cell Area (ha)
1A	336
1B	267
2A	292
2B	131
3	427
4	153
5A	253
5B	992
Total	2,852

Fig. 4 Stormwater Treatment Area 1 West (STA-1W) with cells, flow-ways, water control structures, and vegetation type

and TP loading rates, site geometry, cell configuration, and operational history. Internal P retention and cycling processes are likely to be important and research on internal processes is being planned.

Stormwater Treatment Area Design

Wetlands remove phosphorus via sediment accretion, leaching, vegetation growth and decay, and microbial activities (Kadlec and Knight 1996; Ivanoff et al. 2013). The ability of wetlands to assimilate and store phosphorus is the basis of their utility for treating surface water delivered to the Everglades. Important parameters for designing and operating constructed wetland treatment systems include inflow TP concentration (C_i), inflow rate (Q), desired outflow TP concentration (C_o), outflow rate (Q_o), settling rate (k), depth (d), and available area of land. The settling rate (k) is developed from existing treatment wetlands (Walker 1995). In developing the design basis for the Everglades STAs, Walker (1995) assumed steady-state flow with sheet-flow hydraulics and first-order kinetics. Phosphorus removal in wetlands is



Cell	Cell Area (ha)
1	845
2	974
3	941
4	801
5	896
6	766
7	631
8	597
Total	6,451

Fig. 5 Stormwater Treatment Area 2 (STA-2) with cells, flow-ways, water control structures, and vegetation type

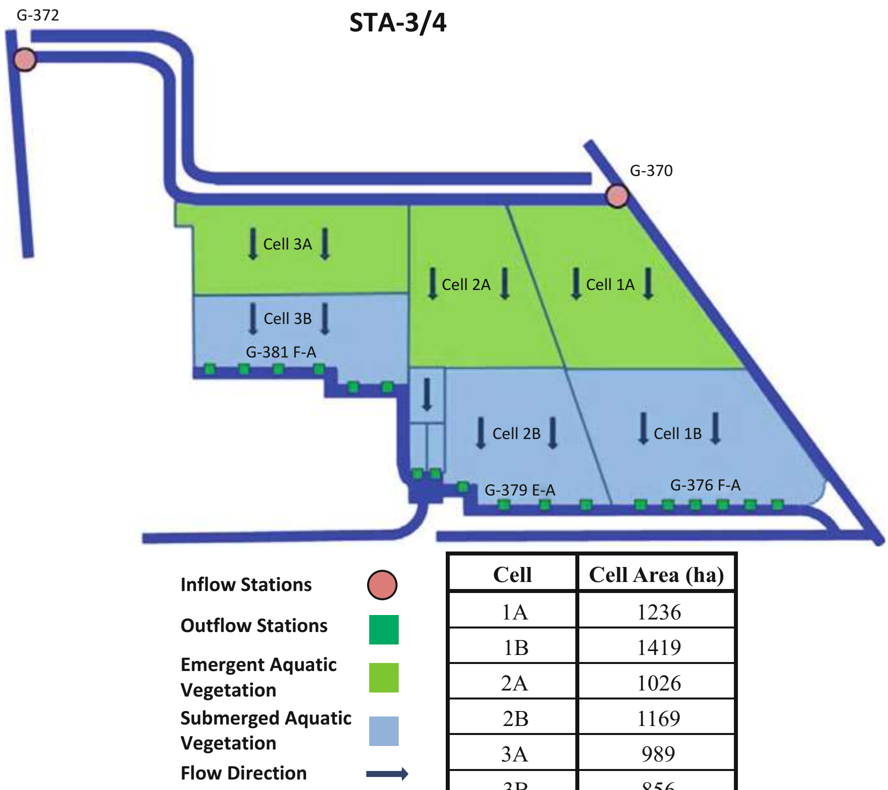
postulated to be a first-order process where the rate of removal is directly proportional to the amount of TP present as expressed by Eq. 1 (Kadlec and Knight 1996) although the process is complex (Fig. 9).

$$C_o = C^* + (C_i - C^*)\exp\frac{-kA}{0.0365Q} \tag{1}$$

Where C_o and C_i are outflow and inflow TP concentrations in mg L^{-1}, C^* is wetland background TP concentration in mg L^{-1}, k is settling rate in m year^{-1}, A is area in ha, and Q is flow in m^3 d^{-1}. The amount of water a given area of wetland can treat depends on the inflow rate, TP load (inflows and concentrations), and desired outflow TP concentration. The required area (A) for a target outflow concentration for a given inflow and wetland background concentrations and for a given inflow (Q) can be computed from Eq. 1. Area (A) can be derived in a few iterations in a spreadsheet.

Hydrology of the Everglades Stormwater Treatment Areas

The hydrology of the Everglades STAs is dominated by the amount and distribution of basin runoff. Direct rainfall to and evapotranspiration from the wetlands is a small fraction of the mass balance accounting for only about 11% and 12% of total inflows and outflows, respectively. The wetlands are enclosed with earthen levee and some seepage in and out occurs through the levees and vertically from the wetlands.



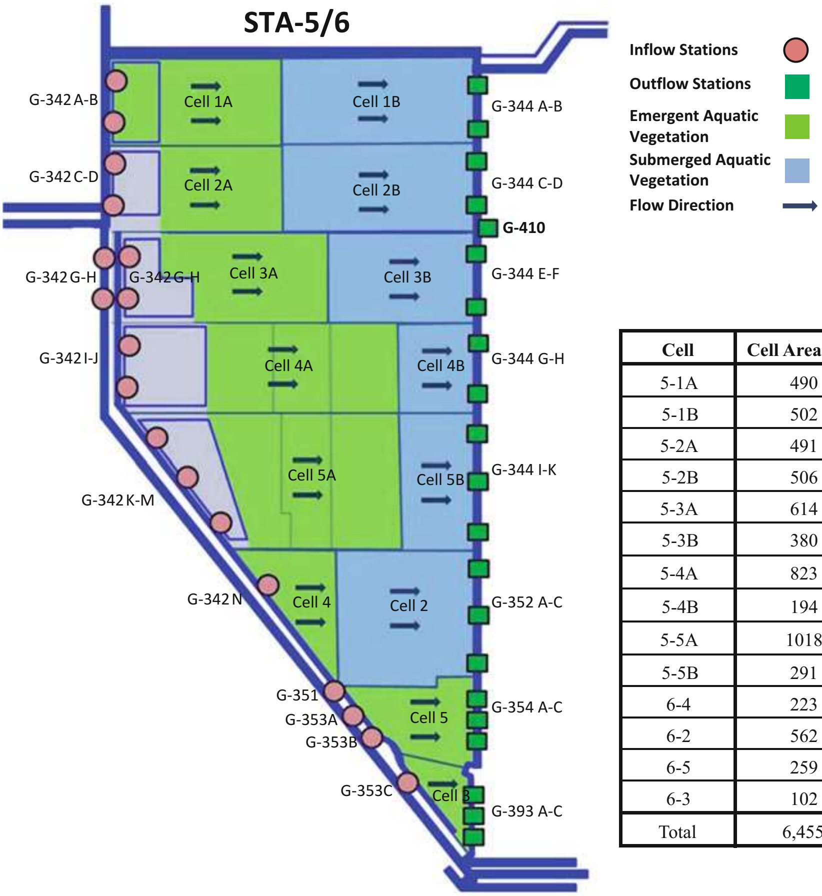
Cell	Cell Area (ha)
1A	1236
1B	1419
2A	1026
2B	1169
3A	989
3B	856
Total	6,694

Fig. 6 Stormwater Treatment Area 3 and 4 (STA-3/4) with cells, flow-ways, water control structures, and vegetation type

Hydrologic characteristics of the Everglades STAs are illustrated by Fig. 10, which shows the monthly hydrology of STA-1W from 1999 to 2012. As shown in the figure, inflows and outflows are similar in most cases, so that differentiating between the two lines in the figure is difficult. The water budget of the wetlands is expressed by Eq. 2 and hydrologic parameters are shown in Fig. 9. Equation two is:

$$\Delta S = \text{Inflow} + \text{Rainfall} - \text{Outflow} - \text{ET} \pm \text{Seepage} \pm \varepsilon_{t,} \qquad (2)$$

where ΔS is the change in storage in the wetland from the beginning day to the last day of the period of water budget analysis and ε_t is total error, which is a cumulative term of measurement errors for all the hydrologic parameters and ungauged sources and sinks. The water budget for STA-1W from 1999 to 2012 is depicted in Fig. 11. The change in storage, seepage, and error terms are lumped together and account for only 3.5% of the water budget. South Florida is relatively a wet region with a mean



Cell	Cell Area (ha)
5-1A	490
5-1B	502
5-2A	491
5-2B	506
5-3A	614
5-3B	380
5-4A	823
5-4B	194
5-5A	1018
5-5B	291
6-4	223
6-2	562
6-5	259
6-3	102
Total	6,455

Fig. 7 Stormwater Treatment Area 5 and 6 (STA-5/6) with cells, flow-ways, water control structures, and vegetation type

Table 2 Common emergent aquatic vegetation (EAV) species

Common name	Scientific name
Arrowhead	*Sagittaria latifolia*
Bulrushes	*Scirpus* spp.
Cattail	*Typha latifolia and/or Typha domingensis*
Duck potato	*Sagittaria lancifolia*
Maidencane	*Panicum hemitomon*
Pickerelweed	*Pontederia cordata*
Sawgrass	*Cladium jamaicense*

Table 3 Common submerged aquatic vegetation (SAV) species

Common name	Scientific name
Southern naiad	*Najas guadalupensis*
Coontail	*Ceratophyllum demersum*
Muskgrass	*Chara* spp.
Hydrilla	*Hydrilla verticillata*

Table 4 Floating aquatic vegetation species

Common name	Scientific name
Water lettuce	*Pista stratiotes*
Water hyacinth	*Eichhornia crassipes*
Duck weed	*Lemna* spp.

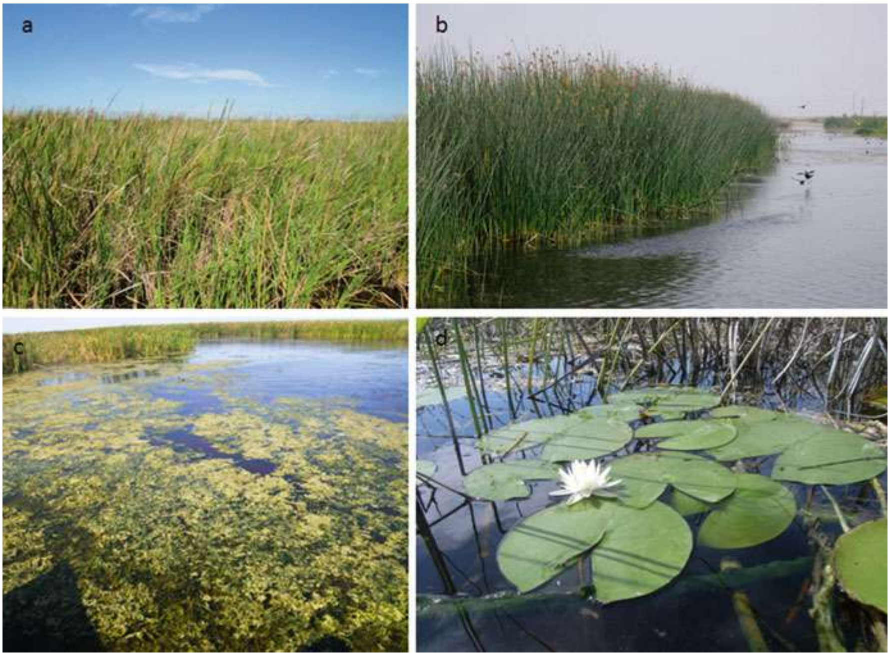


Fig. 8 (**a**) Cattails in STA-1W, (**b**) bulrush in STA-1W, (**c**) SAV with periphyton in STA-3/4, and (**d**) water lily in STA-1W (Photos by the South Florida Water Management District)

annual rainfall of 1,340 mm. Drought frequency has increased in the last decade with several years with below average rainfall (2000, 2001, 2006, 2007, 2008, 2011, and 2012).

Precipitation is well-monitored in STA-1 W with six rain gauges through 2010 and currently five rain gauges. Daily rainfall has high spatial variation in the region,

Table 5 Performance of STAs from 1994 to 2012

	STA-1E	STA-1 W	STA-2	STA-3/4	STA-5	STA-6	All STAs
Start date	Sep-04	Oct-93	Jun-99	Oct-03	Oct-99	Oct-97	***1994-2012***
Inflow volume (million m^3)	783	4,008	3,401	4,577	1,509	846	15,124
TP inflow to date (ppb)	172	171	102	114	225	100	140
Standard deviation TP inflow (ppb)	*58*	*54*	*43*	*31*	*56*	*39*	*26*
TP inflow load to date (mt)	135	689	349	522	341	85	2,120
Outflow volume (million m^3)	765	4,086	3,706	4,648	1,386	574	15,165
TP outflow to date (ppb)	57	51	22	18	93	34	37
Standard deviation TP outflow (ppb)	*121*	*32*	*9*	*4*	*46*	*23*	*13*
Outflow TP load (mt) to date	44	209	81	82	129	20	564
TP retained to date (mt)	91	480	269	440	212	66	1,556

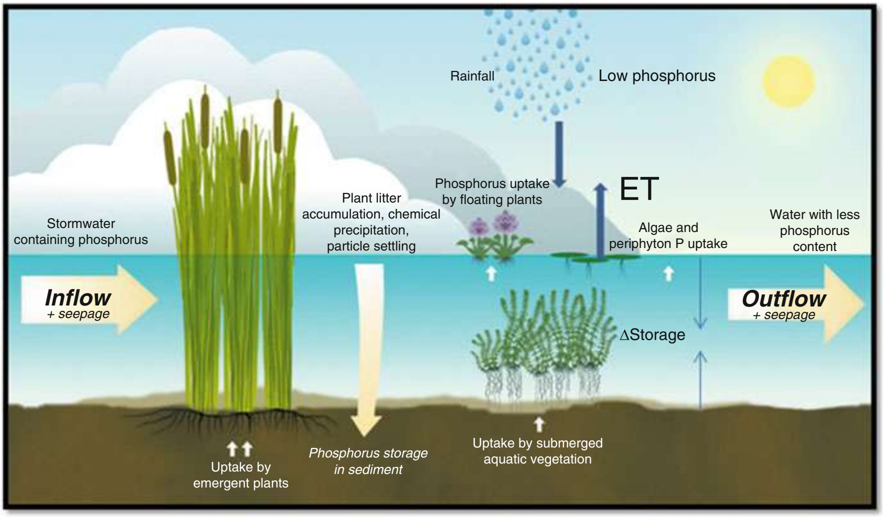


Fig. 9 Hydrologic and phosphorus removal processes of a constructed wetland

but this relatively dense rain gauge network provides good measurements (Abtew et al. 1995). The wet season from June through October contributes 66% of the annual rainfall, and most of the inflows to the STAs are during this season. During dry periods, there were times when treatment cells dried out so that supplemental water from a lake source was needed to keep the STAs hydrated. When treatment cells dry out and later rehydrate, stored phosphorus can be released to the water column resulting in a temporary decrease in the performance efficiency of the STA.

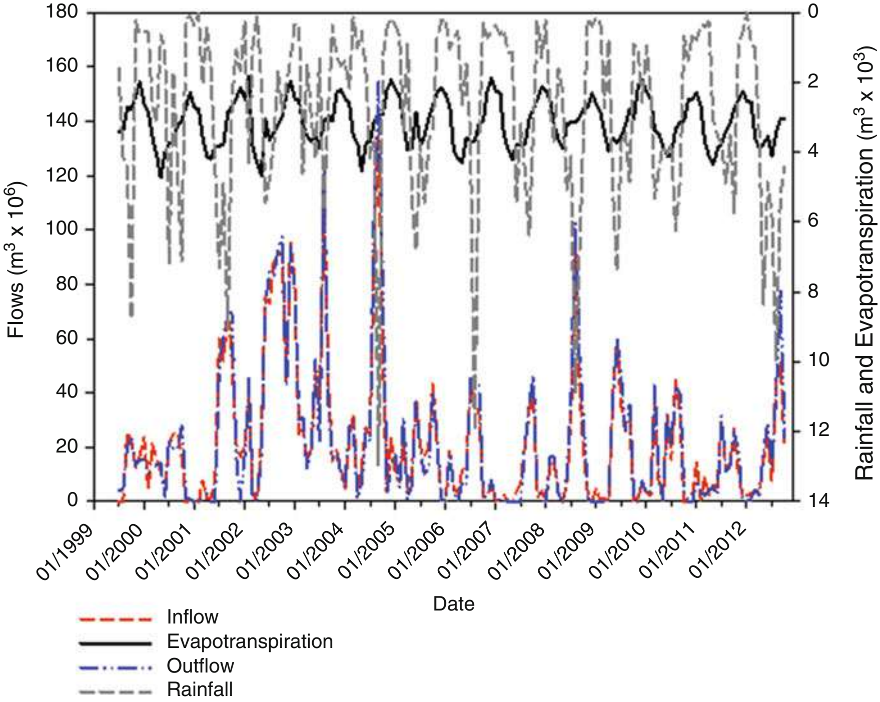


Fig. 10 Monthly hydrologic variation in STA-1W (1999–2012)

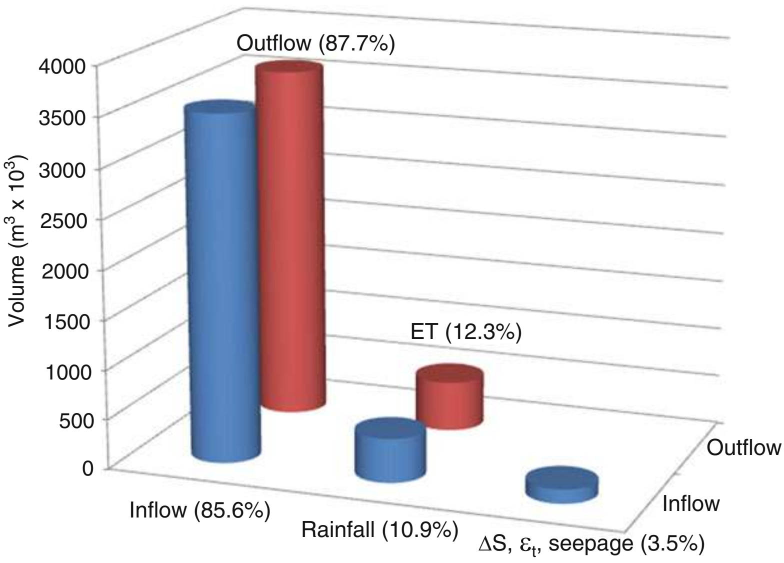


Fig. 11 STA-1W water budget (1999–2012)

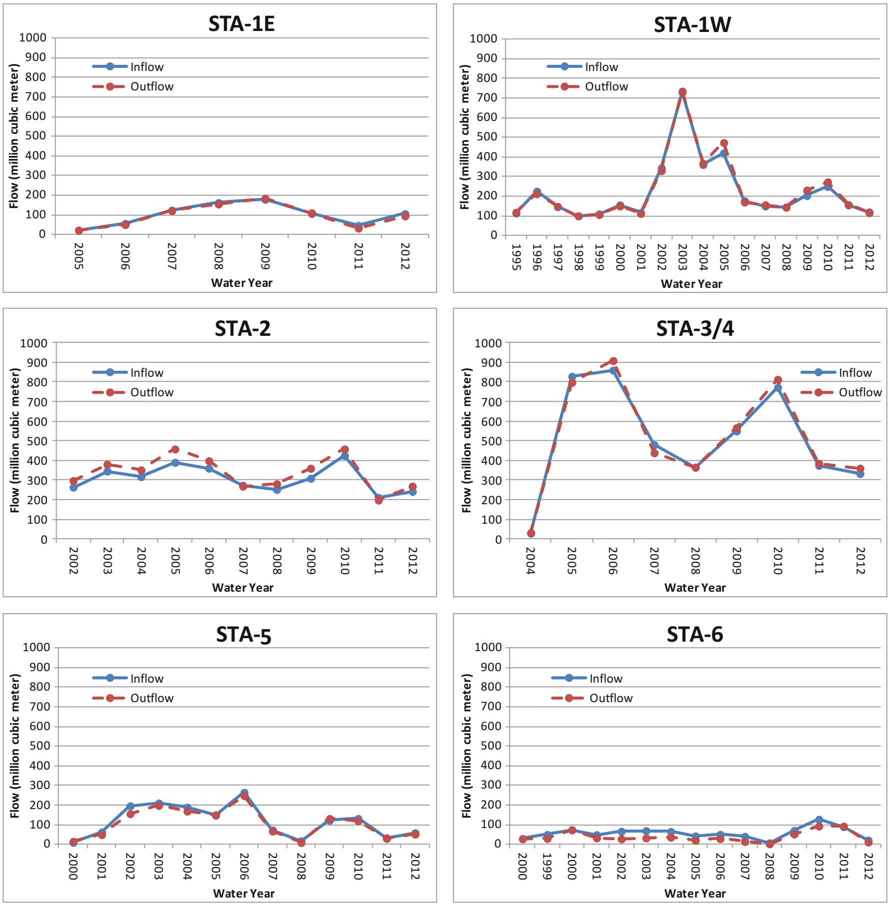


Fig. 12 Annual inflows and outflows for each STA

Wetland evapotranspiration is quantified using an equation that was developed from lysimeter studies in the wetlands (Abtew 1996; Abtew and Melesse 2012). Three different types of lysimeters were installed in cells dominated by cattails, mixed marsh, or open water cells. The study was conducted for 2 years.

Figure 12 depicts annual inflow and outflow from each STA showing that inflow and outflow closely match. Rainfall is a fraction of the water budget, but runoff generated from the rainfall in the upstream basin drives the treatment process. Figure 13 shows the relationship between monthly rainfall in the basin and inflow into STA-1W. Both variables are standardized to a mean of zero and standard deviation of 1 to remove visual distortion of the differences in magnitude and range between the variables.

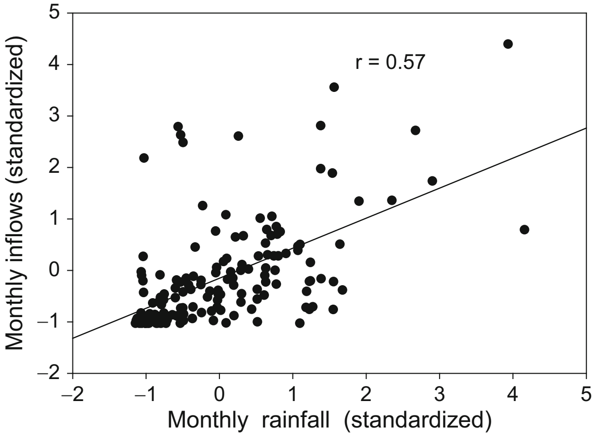


Fig. 13 Correlation between rainfall in the basin and inflows into STA-1W

Stormwater Treatment Areas Operation and Performance

As a major component of Everglades restoration, the STAs were built and are operated to reduce the amount of total phosphorus in surface waters that flow into the Everglades Protection Area (Pietro 2012). The South Florida Water Management District operates the system of STAs with continuous evaluation of their performance and adaptation of operations. The STAs are flow-through systems approximating a plug flow type process rather than a completely mixed reactor type process. STA managers use an adaptive approach and weekly data analyses to make operational decisions. Current water depths in the STAs, outflow TP concentrations, and phosphorus loading rates are considered when making operational decisions for each STA and flow-way. Figure 14 compares inflow and outflow TP concentrations to depict the performance of each STA for its period of operation. The difference in source water TP concentration and the phosphorus reduction efficiency of the STAs is discernible. Tables 5 and 6 depict detail performance of each STA and all STAs combined for period of operation and the 2012 water year (May 1, 2011 to April 30, 2012), respectively. The adjusted effective treatment area is the actual treatment area in water year 2012. These areas vary annually due to many reasons, for example, a cell may be off line for construction or maintenance reasons.

Hydraulic loading rate (HLR) is computed using Eq. 3 as:

$$\text{HLR} = \frac{Qx100}{A}, \tag{3}$$

where HLR is hydraulic loading rate in cm d^{-1}, Q is flow in $m^3\ d^{-1}$, and A is STA area in m^2. Because the length of time that a unit of water resides in the marsh is a factor in TP reduction, hydraulic residence or retention time (HRT) is an important design and operation parameter. It is calculated using Eq. 4 as:.

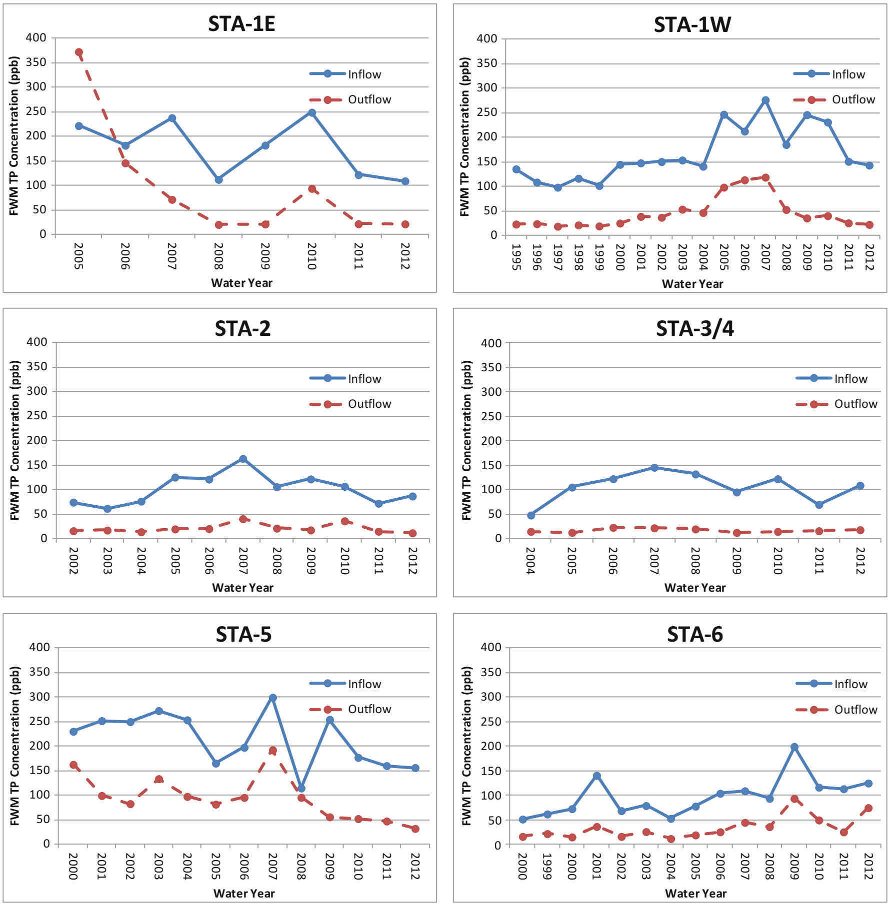


Fig. 14 Inflow and outflow TP concentrations from each STA

$$\mathrm{HRT} = \frac{Ad}{Q^*}, \tag{4}$$

where HRT is in days, A is area in m^2, d is depth in m, and Q^* is average of inflow and outflow in $m^3\ d^{-1}$. TP loading rate is computed as:

$$\mathrm{PLR} = \frac{Q_i C_i}{A}, \tag{5}$$

where PLR is phosphorus loading rate in g m^{-2} $year^{-1}$, C_i is inflow TP concentration in g m^{-3}, Q_i is flow in m^3 $year^{-1}$, and A is area in m^2. Detailed performance parameters for each STA and all STAs for water year 2012 are shown in Table 6. The average TP load reduction for all STAs for water year 2012 was 83%.

Table 6 Performance of STAs (May 1, 2011 to April 30, 2012)

	STA-1E	STA-1W	STA-2	STA-3/4	STA-5	STA-6	All STAs
Effective treatment area (ha)	2,077	2,699	3,335	6,695	2,467	913	18,185
Adjusted effective treatment area (ha)	2,063	2,699	2,565	6,695	2,467	338	16,827
Water year 2012 inflow							
Total inflow volume (million m^3)	105	119	241	332	58	21	877
Total inflow TP load (mt)	11.504	17.105	21.064	36.327	9.157	2.585	97.742
Flow-weighted mean inflow TP (ppb)	109	143	87	109	156	123	111
Hydraulic loading rate (HLR) (cm/day)	1.40	1.21	2.58	1.36	0.65	1.70	1.43
TP loading rate (PLR) (g/m^2/year)	0.56	0.63	0.82	0.54	0.37	0.76	0.58
Water year 2012 outflow							
Total outflow volume (million m^3)	94	116	268	359	51	11	899
Total outflow TP load (mt)	2.010	2.598	3.278	6.675	1.659	0.833	17.054
Flow-weighted mean outflow TP (ppb)	21	22	12	19	32	75	19
Hydraulic residence time (days)	15	41	19	31	46	3	
TP retained (mt)	9.49	14.51	17.79	29.65	7.50	1.75	80.69
TP removal rate (g/m^2/year)	0.46	0.54	0.69	0.44	0.30	0.52	0.48
Load reduction (%)	83%	85%	84%	82%	82%	68%	83%

STA Monitoring and Maintenance

Intensive hydrologic, hydraulic, and water quality monitoring is conducted at each STA to make operational decisions, to characterize treatment efficiency, and fulfill requirements for regulatory permit reporting. Rainfall is measured with a network of gauges and also estimated from radar. Weather stations in the vicinity of each STA provide input data for the evapotranspiration estimation model. Continuous inflows and outflows are measured from automated and mostly remote controlled water control structures such as pumps, spillways, and culverts (Figs. 3, 4, 5, 6, and 7). Generally, the maximum depth of operation for the STAs is 122 cm. Target depth varies slightly cell by cell but is generally 38–46 cm deep. A large network of water stage recorders is instrumented in each STA for operational decision-making, flow computations through structures, and water depth calculations.

Water quality at the inflow and outflow structures is monitored using flow-proportional composite auto-samplers and grab samples are collected weekly. In addition to TP concentrations, many more water quality parameters are measured weekly or bi-weekly as required by permit or for performance monitoring. These parameters are alkalinity, calcium, chloride, dissolved oxygen, dissolved phosphate, hardness, magnesium, ortho phosphate, pH, potassium, sodium, specific conductance, water temperature, total Kjeldahl nitrogen, ammoniacal nitrogen, nitrate + nitrite, total suspended solids, temperature, and sulfate. Mercury and turbidity are also monitored in accordance with specific monitoring plans.

In an ongoing effort to improve performance, various levels of STA maintenance and enhancements are implemented. Short circuiting of flow due to the development of preferential flow paths limits the use of all available treatment area. Structural measures such as building internal levees, planting vegetation, and grading are implemented to improve flow distribution. Hurricanes and other extreme weather events can result in structural damage and vegetation uprooting. An STA cell that requires major rehabilitation work may need months to reach prestorm performance

Fig. 15 Cell 5 at STA-1E with flow short circuiting (*left*) and on the (*right*), after successful establishment of planted bulrush (dark green vegetation) and floating aquatic plants trapped by bulrush stands (Photos by South Florida Water Management District)

levels. Changes in vegetation require maintenance to keep the preferred type of coverage. Figure 15 depicts an STA cell where an opening in the vegetation created short circuiting (left) and was rectified by planting macrophytes to reduce short circuiting (right).

Summary

Over 18 years of operation, the Everglades STAs have demonstrated that constructed wetlands can effectively reduce TP concentrations in surface water discharges to the Everglades. An average of 73% removal efficiency is observed from all the STAs. The hydrology of the wetlands has been studied extensively and operational criteria have been continuously improving based on experience. Each STA is different due to soil type, antecedent land use, inflow rate and TP concentrations, area, cell configuration, topography, and type of vegetation cover. Therefore, constructed wetlands require tailored operations and maintenance for their specific conditions. Long-term operation of these constructed wetlands will provide data to further improve the design, operation, and treatment performance of large-scale constructed treatment wetlands.

Future Challenges

Current estimates of sediment accrual rates suggest that sediment management may be needed in the constructed wetlands after 25 or more years. The STA optimization research program continues in support of the effort to achieve further reduction of outflow TP concentrations. Understanding the factors controlling long-term performance also continues as the STAs are operated. Ongoing activities to understand and improve performance include implementation of vegetation management strategies, pre- and postrehabilitation monitoring, poststorm monitoring, and post–dry out monitoring.

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Section V

Ecological Processes and Biogeochemistry

C. Max Finlayson

Microbially Mediated Chemical Transformations in Wetlands

31

Darren S. Baldwin

Contents

Abstract

This chapter discusses how the microbiota mediate the cycling of carbon, nitrogen, phosphorus and sulfur in wetlands. Wetlands have a number of characteristics that make them hot-spots for microbially-mediated transformation of constituents. Abundant plant life, adapted to aquatic environments represents a source of carbon (energy) to fuel microbial processes. Wetlands are sites of deposition of sediments, and often interact with ground-water, both of which are source of electron-acceptors for microbial processing. Biofilms growing on emergent and submerged plants, as well the root zone of aquatic plants, are active sites of reactions that require both oxic and anoxic zones such as coupled nitrification-denitrification. Finally, many wetlands are ephemeral, meaning the sediments undergo periodic wetting and drying, which can strongly influence nutrient cycling.

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Keywords
Carbon · Nitrogen phosphorus · Sulfur · Nutrient cycling · Wetting and drying

Introduction

The microbiota, particularly bacteria and fungi, play a major role in the biogeochemical cycling of materials. Thus, these microbes are responsible for the degradation of organic matter and the cycling of nutrients and are crucial controllers of plant productivity and environmental resilience of ecosystems to disturbance. In short, microbial processes are critical for many aspects of ecosystem structure and function. Although wetlands represent only a small proportion (2–6%) of the Earth's land surface, they play a disproportionally large role in the cycling and transformation of chemical constituents on a global scale (Kayranli et al. 2010). The combination of soil saturation and specifically adapted plant species in wetlands is one of the main reasons that wetlands are important hotspots for chemical transformations.

The abundant plant life often associated with wetlands leads to a large input of organic carbon to the sediments when the plants die, and the fate of this material regulates the nature of chemical transformations in wetlands. This organic carbon is decomposed by microorganisms. During decomposition of the plant material, the bacteria also consume oxygen. Oxygen is poorly soluble in water – at 20 °C water that is fully saturated contains only about nine parts per million of oxygen. Because the soils usually are saturated with water, oxygen can be rapidly depleted so that environments ultimately may become anaerobic (i.e., contain no oxygen). Anaerobic zones are common in wetlands, particularly in the sediment profile. However, the loss of oxygen does not mean that life cannot occur in wetlands. There are a multitude of bacteria that can use other chemical compounds for respiration in the same way that we use oxygen. Nitrogen, carbon, and sulfur compounds as well as metals such as iron and manganese can all be used by various microorganisms in anaerobic respiration. During anaerobic respiration these compounds (known as electron acceptors) undergo reduction (gain electrons), while an organic carbon molecule is oxidized (loses electrons). Although reduction reactions are important in the cycling of many elements (including C, N, P, Fe, and S), the anaerobic zones, where these reactions can occur, are not common in the terrestrial landscape, hence the disproportionate importance of wetlands in the cycling of these elements.

Wetlands often have a number of other unique characteristics that make them hotspots for microbially mediated chemical transformations (sensu McClain et al. 2003):

Sites of sediment deposition: Wetlands are often found at the terminus of a flow pathway. Consequently, wetlands are characterized by generally having low flows relative to the watercourses that feed them. As such, terminal wetlands act as zones where fine suspended materials (including suspended sediments and particulate organic matter) are deposited, a process enhanced by the presence of aquatic vegetation such as emergent reeds, which slow the velocity of water. Therefore, the bed sediments of wetlands are often very fine and rich in organic matter. This combination of reduced flow, lack of sediment porosity, and high organic matter load results in sediments that tend to be devoid of oxygen.

Interception of groundwater: It is not uncommon for wetlands found in low areas of the landscape to intercept shallow groundwater. Groundwater can be a significant source of some electron acceptors (e.g., sulfate and nitrate) for a wetland (Baldwin and Fraser 2009). Groundwater often flows within its aquifer. Groundwater flow can be an important pathway for the export of dissolved material from a wetland.

Biofilms growing on aquatic macrophytes: Biofilms are complex assemblages of algae and bacteria that form on solid substratum and are often important sites for chemical transformations, particularly in aquatic ecosystems. One characteristic of wetlands is the presence of plant species that are adapted to partial or complete inundation for long periods of time. Substantial quantities of biofilms can form in wetlands on the submerged stems and leaves of these macrophytes.

The rhizosphere: The rhizosphere is the area immediately adjacent to the roots of plants. Although wetland sediments tend to be anoxic, wetlands support a variety of submerged and emergent macrophytes whose root systems extend into the anoxic muds. To survive, these root systems require a supply of oxygen. Many wetland macrophytes have evolved both active and passive mechanisms to transport oxygen to their root systems (e.g., Sorrell 2004). Oxygen leaking from the roots creates a zone of oxic sediment close to the root surface. Therefore, the sediment under stands of macrophytes will have zones of oxidized sediment threading through the predominantly anoxic sediment. The juxtaposition of aerobic and anaerobic zones favors a number of reactions that require a supply of an oxidized component to the anaerobic zone, e.g., coupled nitrification–denitrification (see under section "Nitrogen Cycling"). Bioavailable carbon compounds are also exuded from aquatic macrophyte roots, enhancing microbial activity in the sediment.

Wetting and drying: Many wetlands are ephemeral – undergoing periodic drying. The impact of drying on chemical transformation in wetlands is quite complex (Baldwin and Mitchell 2000). One important process is the Birch effect – named after the soil scientist H.F. Birch who first observed the phenomenon in African soils. When soils and sediments that have been dried out are rewetted, there is a pulse of nutrients (particularly C and N) released into the overlying water column. Most likely, these nutrients were previously locked up in microbial biomass, so that nutrients are released after the sediments dry.

Chemical Transformations in Wetlands

The following section discusses in more detail some of the more important microbially mediated biogeochemical cycles within wetlands. Although each cycle is dealt with separately, the cycles are not independent, and processes that occur in one cycle may impact on the cycling of another element, e.g., the anaerobic oxidation of methane during sulfate reduction.

Carbon Cycling

Carbon decomposition, either through respiration or fermentation, is the principal source of energy for non-photosynthetic organisms. Hence understanding the cycling of carbon is important in understanding ecosystem functioning. Somewhat paradoxically, wetlands are both hotspots for carbon decomposition (including the production of the greenhouse gases methane and carbon dioxide) and, simultaneously, one of the most important ecosystem types for carbon sequestration in the landscape. While wetlands only occupy a small proportion of the world's land surface area, they produce about 80% of all methane produced from natural sources (Anderson et al. 2010) yet also contain about one third of the organic matter stored in the world's soils (Kayranli et al. 2010). Much of the carbon found in wetlands is originally derived from aquatic and riparian plant material. As this material decomposes, carbon is respired as carbon dioxide or methane, incorporated into new biomass, and/or sequestered in wetland sediments. The fate of carbon will depend ultimately on the chemical structure of the carbon and the organisms responsible for its decomposition.

Most of the carbon found in plant material is incorporated into large, sometimes complex, molecules including cellulose, hemicelluloses, and lignin. Before these large molecules can be used in fermentation or respiration, they first must be broken down into smaller molecules.

Cellulose and hemicelluloses are important structural polymers made up of repeating sugar units. In cellulose, there is only one type of sugar present (glucose), while hemicelluloses may contain a number of different types of sugars. Hemicellulose can account for up to 50% of plant biomass. The actual chemical process responsible for the breakdown of cellulose and hemicellulose into their component sugars is called hydrolysis, which is the chemical breakdown of a molecule by water. However, cellulose is not very soluble in water. In nature, these polymers are broken down into their component sugars with the help of enzymes excreted by a variety of microorganisms, including fungi, as well as both aerobic and anaerobic bacteria. Therefore, cellulose degradation will occur in both anaerobic and aerobic zones, as long as there is free water available. Hence cellulose degradation will occur more rapidly in moist to wet environments such as those found in wetlands.

The other major structural polymer found in plant material is lignin. Lignin is a relatively complex molecule containing phenolic linkages (Fig. 1). Lignin is broken down principally by a group of enzymes known as phenol oxidases. These enzymes are mostly produced by fungi, rather than bacteria, under aerobic conditions (Krauss et al. 2011). Under waterlogged, and therefore mostly anaerobic, conditions often found in wetland sediment, lignin is very slow to decompose. In fact, lignin often reacts with other chemical compounds to form even more poorly soluble and unreactive molecules – a process known as humification. Humification is a process that stabilizes organic carbon molecules and is the process by which wetlands are able to sequester carbon and other nutrients (Pant and Reddy 2001).

Smaller molecules can be further decomposed by the process known as fermentation. During fermentation sugars, peptides, and amino acids are converted to alcohols and low-molecular-weight volatile fatty acids like acetate, as well as

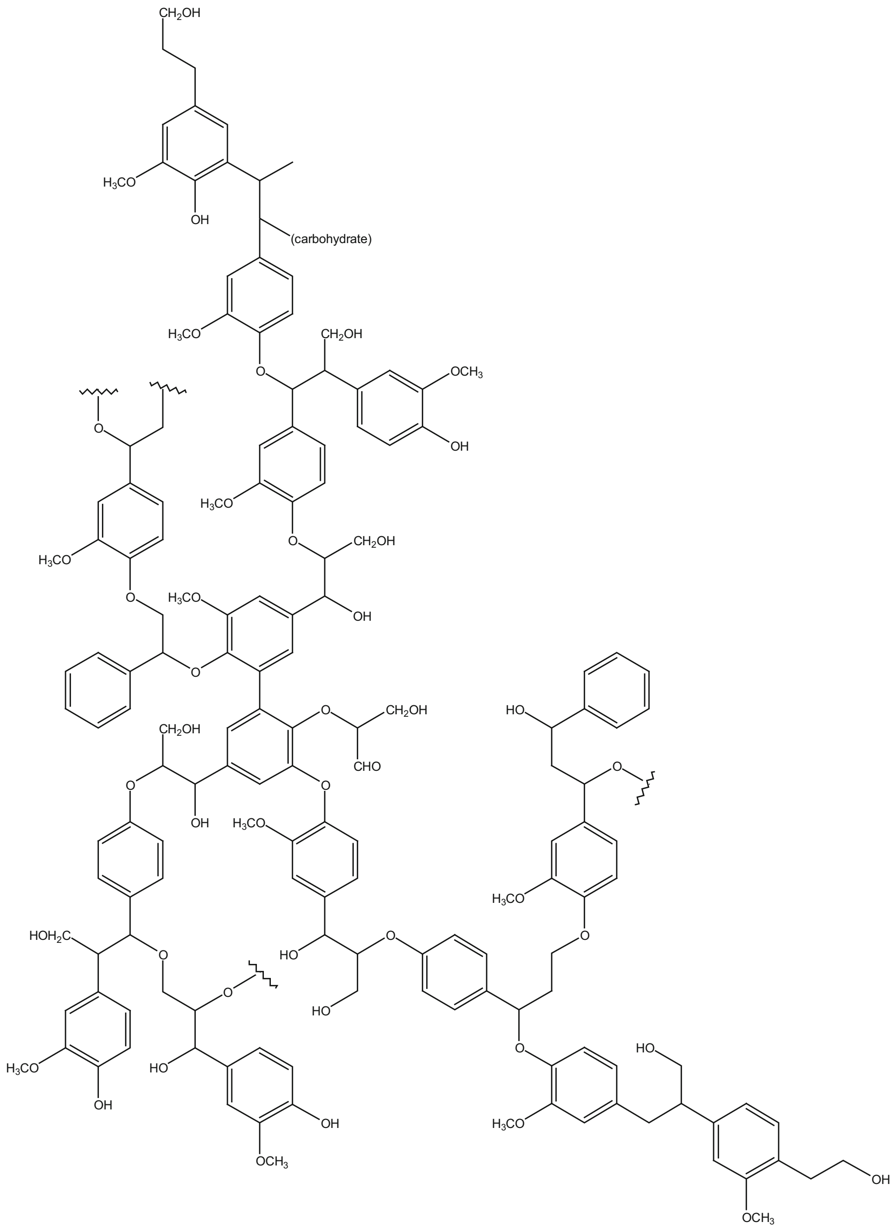

Fig. 1 Partial chemical structure of lignin

producing CO_2, hydrogen, and ammonia (NH_3). The activity of the fermenters provides important precursors to the other functional groups of anaerobic bacteria. As such, fermenters are essential anaerobic nutrient cyclers but may be limited in some environments.

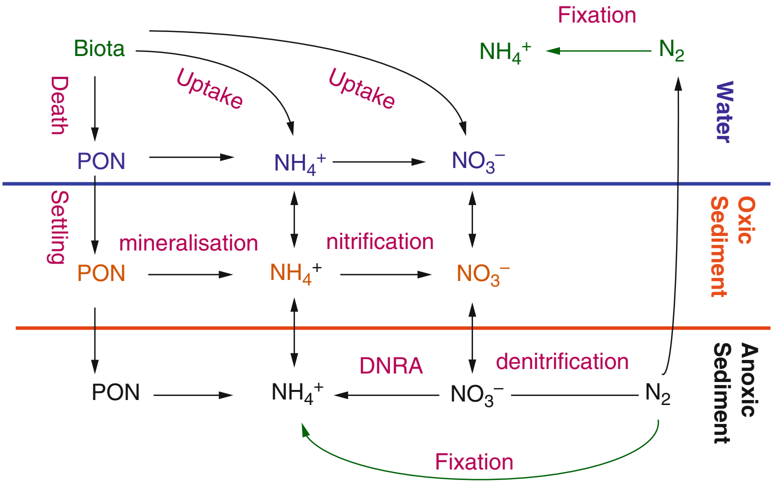

Fig. 2 Stylized nitrogen cycle (Modified after Rysgaard et al. 1993)

The terminal step in organic matter degradation in wetlands is the production of methane (methanogenesis). Methane is a potent greenhouse gas, which is produced solely under anaerobic conditions by bacteria called methanogens. Methanogens can only use a small suite of carbon molecules to produce methane (Oremland 1988), which include carbon dioxide in conjunction with hydrogen gas, formate, acetate, methanol, methylamine, or dimethyl sulfide. Most of these molecules are produced through fermentation. Not all the methane produced in a wetland is lost to the atmosphere. Under aerobic conditions, one group of methane-oxidizing bacteria, called methanotrophs, use methane as their sole source of both energy and carbon by converting methane and oxygen to form formaldehyde. Methane oxidation can also occur anaerobically, at least in marine systems (Valentine and Reeburgh 2008). Under anaerobic conditions a large group of bacteria reduce sulfate to sulfide and in the process convert methane to bicarbonate ion. There is some evidence to suggest that anaerobic methane oxidation can also occur in freshwater wetlands under the right conditions (Baldwin and Mitchell 2012).

Nitrogen Cycling

A stylized version of the aquatic N cycle is presented in Fig. 2 (after Rysgaard et al. 1993). Nitrogen can enter an aquatic system through a number of pathways including as part of either living or dead biomass (“organic N”), through fixation of N_2 gas, from anthropogenic inputs including point-source and non-point-source pollution, from aerial deposition, and groundwater sources.

After N enters the aquatic system, the N can undergo a number of transformations. Like most aquatic biogeochemical cycles, the cycling of N is predominantly mediated by the microbiota. Furthermore, similar to other biochemical cycles, different reactions occur under aerobic and anaerobic conditions. In the aerobic zone, ammonia, which is produced through the mineralization of organic N, can be converted to nitrate (via nitrite) by bacterial action, i.e., nitrification. Ammonia can also either be taken up by bacteria, algae, and/or macrophytes or, if the pH is sufficiently high, lost to the atmosphere through volatilization. In the anoxic part of the sediment, nitrate can either be reduced to ammonia (dissimilatory nitrate reduction to ammonia; DNRA) or converted to nitrogen gas (denitrification). DNRA preserves N in the system. Because N_2 gas has a low solubility in water, denitrification is a pathway for the removal of nitrogen from aquatic systems. This low solubility is particularly important if nitrification and denitrification are coupled (i.e., the two processes occur spatially close to each other), as the nitrate produced in nitrification can be quickly converted to nitrogen gas rather than being assimilated into biomass. Coupled nitrification–denitrification has to occur at the boundary of oxic and anoxic environments, and indeed the rhizosphere is believed to be the most important sites for denitrification at a global scale.

Phosphorus Cycle

Phosphorus is often the limiting nutrient in aquatic ecosystems. Excessive levels of P can lead to the formation of nuisance algal blooms and is associated with poor water quality. Like N, both natural and constructed wetlands have been used to control P concentrations before it can enter receiving waters. However, unlike N, P does not have an important gaseous form which means that any P trapped within a wetland needs to be actively harvested before it can be remobilized.

Phosphorus can enter a wetland either in dissolved form or in association with organic or mineral particles. The fraction of phosphorus delivered in each of these forms depends on the geology land use, land cover, hydrology, and the nature of the drainage network flowing into the wetland. Within a wetland, the P can be partitioned between the water column, biota, and sediments, and P can rapidly cycle between these pools.

Phosphorus in the water column can either be dissolved or particulate. Dissolved P forms include the free orthophosphate ion (HPO_4^{2-}), as polymers of phosphate molecules (“polyphosphates” such as tripolyphosphate, a compound found in detergents – $H_5P_3O_{10}$), or as part of an organic phosphorus molecule such as DNA. Only the free orthophosphate ion can be directly taken up by most bacteria, algae, and plants. Therefore to be bioavailable, the form the P is in must be converted to orthophosphate. Both dissolved polymeric phosphates and organic phosphate molecules can be converted to orthophosphate by enzymes secreted by bacteria and algae. Particulate P in the water column can either be mineral based or biotic (e.g., in bacteria and algae). Orthophosphate has a strong affinity for a number of metals including iron, calcium, and aluminum and will preferentially bind to suspended particles containing these materials.

By far the largest pool of P in most wetlands is in the sediments. P can enter the sediment through settling of particulate matter, death, and deposition of both plants

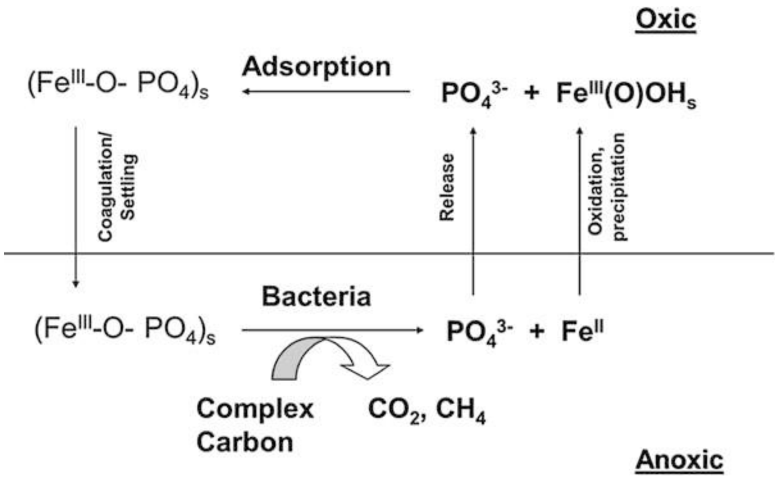

Fig. 3 Stylized phosphorus cycle

and animals and through adsorption of dissolved P onto sediment particles. While sediments may represent the largest sink in aquatic systems, these sediments may also represent a major source of P. Remobilization of P from sediments to the overlying water involves two processes, release of P from sediment particles to the interstitial waters within the sediment and transfer of P from the interstitial water to the overlying water column (Fig. 3). Although there are a number of ways that P can be remobilized from sediments into the surrounding pore water, the most important pathway seems to be the reductive dissolution of iron minerals under anaerobic conditions and subsequent release of any P adsorbed to the iron particle. Reductive dissolution can occur either directly or indirectly by bacteria. Iron-reducing bacteria directly convert oxidized (ferric) iron particles to dissolved reduced (ferrous) iron solubilizing any P associated with the particulate iron. Alternatively, there is another group of anaerobic bacteria – sulfate-reducing bacteria, which use sulfate as their electron acceptor. The sulfate is reduced to sulfide (S_2^-). Sulfide reacts with iron minerals releasing any P bound to the mineral. Under highly reducing conditions, anaerobic bacteria can reduce phosphate to phosphine, a colorless, odorless gas, which is spontaneously flammable in air. While phosphine has been detected in wetlands, the flux from the sediment is relatively small compared to the flux of phosphate.

For the P to remobilize from the pore water to the overlying water column, it must move from the sediment either by diffusion or by disturbance of the pore water in the sediment. Diffusion is a slow process and consequently probably is not important in the exchange of P from sediment pore water to the overlying water column. The disturbance may be caused by high-flow events which result in sediment bed scouring, anthropogenic causes (e.g., dredging), ebullition of gases (such as methane and CO_2) from the sediment-entraining pore water, disturbance of sediments by bottom-feeding organisms (e.g., carp), or by bio-irrigation caused by sediment-dwelling invertebrates. Alternatively, P in the sediment pore water can be taken up by aquatic plants through their root systems.

Sulfur Cycling

Traditionally, it has been assumed that the biogeochemical cycling of sulfur was only important for wetlands that were influenced by seawater. The levels of sulfur in most inland wetlands were believed to be too low to be important compared to other processes. However, mostly through the action of humans, the concentration of sulfur in many inland wetlands worldwide has increased substantially over the last few decades (e.g., Jansen and Roelofs 1996). There are a number of sources of sulfur to wetlands. Burning of sulfur-rich fossil fuels led to substantial aerial deposition of sulfur in the northern hemisphere (acid rain). While there are now actions in place to mitigate this pathway, the legacy of high concentrations of sulfur remains. Increased sulfur has also been associated with the salinization of wetlands. Salt of marine origin contains sulfate; hence where there are high levels of salt in the landscape, we also find significant concentrations of sulfate. Altered hydrology in landscapes (as a result of land clearing, irrigation, and drainage networks) has resulted in increased mobilization and discharge of saline groundwater in the landscape with associated increased mobility of sulfate and accumulation in wetlands (Baldwin and Fraser 2009). Sulfur compounds are also found in agricultural fertilizers and soil ameliorants. Elevated levels of sulfate are also associated with the discharge of municipal wastewater partially through the use of aluminum sulfate as a flocculent in water treatment plants.

The dominant form of dissolved sulfur under aerobic conditions is sulfate. Sulfate is the terminal electron acceptor for a group of anaerobic bacteria called sulfate reducers. Sulfate-reducing bacteria (SRB) can use a wide range of organic substrates including low-molecular-weight organic acids, aromatics, hydrocarbons, amino acids, and ethyl alcohol for the dissimilatory reduction of sulfate to produce sulfide. Metabolic by-products of SRB include carbon dioxide, hydrogen gas, and acetate, each of which is substrates for further anaerobic bacterial processes including sulfate reduction. The sulfate-reducing bacteria can outcompete methanogens for carbon substrate so that sulfate reduction will suppress methane production in wetlands. Sulfate reduction has also been implicated in anaerobic oxidation of methane to carbon dioxide in wetland sediments, further reducing methane emissions. Sulfate reduction has also been linked to the conversion of mercury to methylmercury in wetland sediments. Methylmercury is a potent neurotoxin and endocrine-disrupting chemical that can bioaccumulate through the food web.

The sulfide produced into during sulfate reduction can be oxidized to either elemental sulfur or sulfate by a range of anaerobic photosynthetic bacteria or aerobic lithotrophic bacteria, respectively. A lithotroph is a bacterium that can use a reduced inorganic substrate for biosynthesis. Alternatively sulfide can react with metal containing minerals to produce metal sulfides. The most common metal sulfides found in wetlands are iron sulfides including mackinawite (FeS), greigite (Fe_3S_4),

Fig. 4 Bottle Bend Lagoon, Murray River Watershed, Southeastern Australia (© Murray-Darling Freshwater Research Centre used with permission)

and pyrite (FeS_2). If the metal sulfides are exposed to oxygen (e.g., because of a drying out of the wetland), these compounds undergo a complex series of reactions that ultimately produces acid. If the amount of acid produced is greater than the system's ability to neutralize that acid, then the wetland can acidify. One of the most dramatic examples of this is Bottle Bend Lagoon, an oxbow lake near the Murray River in Southeastern Australia (Fig. 4). A partial drying event exposed metal sulfides to the atmosphere. The pH in the wetland fell from about eight to less than three in a matter of weeks and remained highly acidic for many years until the acid was flushed from the wetland during a large overbank flood (McCarthy et al. 2006). Acidification can also mobilize metals from the sediment.

Future Challenges

Wetlands play a critical role in the global biogeochemical cycling of a number of important elements including carbon, nitrogen, phosphorus, and sulfur. However, across the globe a large number of wetlands have been permanently altered because of human activity. Into the future, the combination of climate change and increasing human population will put further pressure on remaining wetlands. Loss of individual wetlands occurs at a local scale, but cumulatively the effects are global in nature.

Our challenge into the future is to balance the needs of local communities without further disrupting the important role wetlands play in the cycling of important constituents.

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Carbon Flux from Wetlands

32

Hojeong Kang and Inyoung Jang

Contents

Abstract

Human activities have accelerated decomposition in wetland ecosystems, destabilizing carbon stocks in them. In particular, global climate change, drainage and atmospheric deposition are key activities that affect wetland carbon cycle substantially. Global climate change can affect carbon decomposition in wetlands

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by warming effects as well as more frequent droughts. Elevated CO_2 itself can increase dissolved organic carbon leaching from wetlands through enhanced primary production. For coastal wetlands, sea level rise can also affect carbon mineralization by changes in water chemistry as well as oxygen availability. Wetlands have been subject to drainage for the development of agricultural fields and urban dwellings which can accelerate carbon decomposition by aeration. Finally, nitrogen and sulfur deposition can either increase or inhibit carbon decomposition depending on the nutrient status and locations. Overall, human activities can accelerate carbon decomposition in wetlands resulting in greater carbon releases into the atmosphere as CO2 or CH4, and into the aquatic ecosystems as DOC.

Keywords
Carbon storage · Carbon mineralization · Global warming · Greenhouse gas emission · Sea level rise · DOC

Introduction

Wetlands contribute to the global carbon cycle via two basic processes. One is primary production, which takes carbon dioxide from the atmosphere and converts it to organic carbon, comprising plant biomass and soil organic carbon. The second is the decomposition of organic matter, which releases carbon gases to the atmosphere. They also have several unique characteristics through which they have a significant effect on the global carbon cycle. For example, although northern peatlands have a low rate of primary production, their decomposition rate is even lower, resulting in the accumulation of undecomposed materials as peat. In contrast, some salt marshes exhibit very high primary production, higher than that of tropical forests. Each of these processes is discussed below.

Primary Production in Wetlands

In wetlands, primary producers, such as trees, shrubs, and floating plants, fix atmospheric carbon dioxide via photosynthesis. Because carbon dioxide is rarely a limiting factor, nutrients (e.g., nitrogen and phosphorus), water-logging, light intensity, and temperature often control the overall primary production of wetlands. The amount of carbon fixed by global wetlands is estimated to be 137 Tg year^{-1} (Mitra et al. 2005). As the types of wetlands are extremely diverse, their primary production is also highly variable. For example, many northern peatlands, such as bogs, are dominated by *Sphagnum* mosses, which usually grow in nutrient-poor and relatively cold regions. Therefore, their primary production is small. In contrast, marshes, another wetland type often found in temperate regions, have a much higher level of primary production (Mitsch and Gosselink 2007). Given this large variability, primary production in four widespread wetland types is discussed separately.

Peatlands

Peatlands are mainly found in high-latitude areas (northern peatlands) but also in tropical regions (tropical peatlands) or high-altitude areas of temperate regions (mountain peatlands). A typical peatland is a Sphagnum-dominated "bog," where the main input of water is from rain with low levels of cations and nutrients. These systems are referred to as "ombrotrophic." The primary production of such system is generally lower than that of other wetlands, ranging from 250 to 500 g C m^{-2} $year^{-1}$ (Mitsch and Gosselink 2007). In contrast, peatlands also include "fens," which are often influenced by the inflow of water from an adjacent watershed or ground water (called "lithotrophic"). This results in a higher primary production than that of bogs (Mitsch and Gosselink 2007).

Freshwater Marshes

Marshes can be found in temperate and tropical regions. The most distinctive difference between marshes and peatlands is that most marshes are dominated by mineral soils. Marshes are often characterized by emergent, submerged and floating-leaved plants. Depending on the annual mean temperature, hydrological patterns, and nutrient availability, the primary production of marshes ranges from 650 to 1600 g C m^{-2} $year^{-1}$ (Bridgham et al. 2006).

Salt Marshes

Salt marshes normally occur close to the oceans and are dominated by plants with special adaptations to high salt concentrations. For example, *Spartina* and *Salicornia* (glasswort) species are typically found in salt marshes in temperature regions. Although high salinity can stress most plants, specialized plants in salt marshes can also obtain energy from tidal flow and nutrients from adjacent seawater and freshwater. As a result, the primary production in salt marshes is often greater than that of other types of ecosystems, not only other wetlands. For example, the average primary production of salt marshes is estimated to be 900–2000 g C m^{-2} $year^{-1}$ (White et al. 1978).

Swamps

This type of wetland is dominated by large trees. A large amount of carbon occurs as standing biomass in this ecosystem, and primary production ranges from 500 to 1800 g C m^{-2} $year^{-1}$ (Titlyanova 2008).

Decomposition in Wetlands

Organic matter can be supplied to wetlands by primary production on site as litter and root exudates or by delivery from inflow water. Organic matter can accumulate in wetlands or is decomposed by invertebrates or microorganisms. In particular, various types of microorganisms, such as bacteria, fungi, and prokaryotic microbes (*Archaea*), play a central role in the complete decomposition of organic matter. The decomposition of organic matter in wetlands is of great importance because it regenerates inorganic nutrients for plant growth, produces various gases, provides dissolved organic carbon to wetland water, and determines whether wetlands accumulate carbon or not (Mitsch and Gosselink 2007).

Decomposition rates in wetlands are determined by several factors, including the chemical properties of organic matter (recalcitrant versus labile), oxygen supply (aerobic versus anaerobic), temperature, nutrient supply (eutrophic versus oligotrophic), and microbial properties (Bridgham et al. 2006). For example, easily decomposable carbon, such as monomeric sugar, is promptly decomposed under high temperature and low water table conditions (i.e., high oxygen supply) and with a sufficient supply of nitrogen. In contrast, litter with a high lignin content in cold and water-logged conditions, such as in northern peatlands, is very slowly decomposed.

From a carbon-cycle perspective, the fate of organic carbon decomposed in wetlands occurs in three forms: carbon dioxide (CO_2), methane (CH_4), and dissolved organic carbon (DOC). Gas emission is often measured by analyzing gas concentrations, which can be collected in a static chamber installed on wetland surface (Fig. 1). Key

Fig. 1 Static chamber method to collect gas samples from wetland

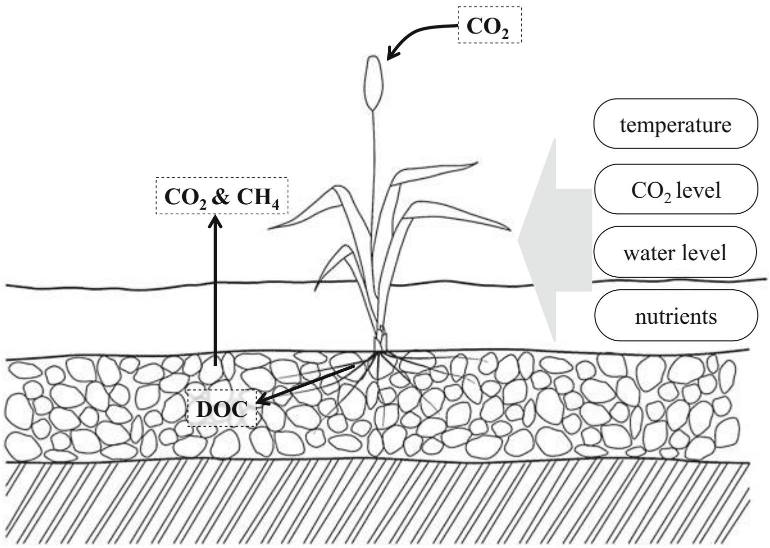

Fig. 2 Influences of environmental changes on carbon dynamics in wetlands

controlling variables for carbon cycle and decomposition are presented in Fig. 2. The pathways for the generation of these products are discussed in detail in the following sections.

Carbon Dioxide

The most common product of the microbial decomposition of organic matter is carbon dioxide. When oxygen is available, microorganisms decompose and metabolize various forms of organic carbon and release CO_2 as a final product. Wetland types differ substantially in the amount and characteristics of organic matter, and thus CO_2 emissions from wetlands widely vary. Furthermore, even within a single wetland, CO_2 release patterns often show a strong seasonality as well as a strong daily variation due to water level and temperature changes. In general, if sufficient carbon is supplied under high temperature and low water level conditions, CO_2 emissions peak. The total amount of CO_2 released from global wetland soils is estimated to be 76.5 Pg C year^{-1}, although substantial uncertainties are involved (Mitsch and Gosselink 2007).

Methane

Methane is mainly produced by a certain type of *Archaea*, collectively referred to as "methanogens" (Aselmann and Crutzen 1989). In wetlands, a lack of oxygen activates fermentation processes that generate products, including various organic acids (e.g., acetate) and hydrogen. Methanogens can metabolize such organic acids,

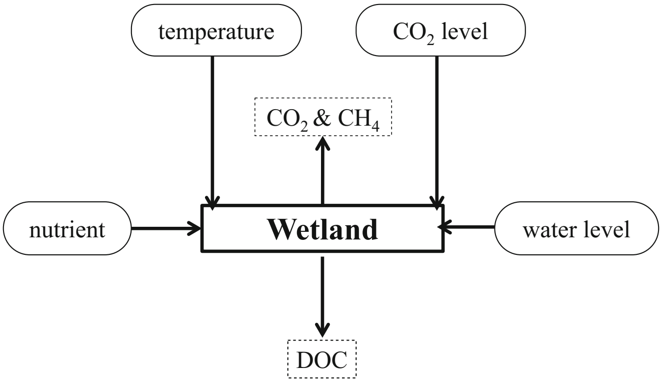

Fig. 3 Biochemical pathway of CH_4 oxidation to CO_2 by methanotrophic bacteria

hydrogen, and other compounds to produce CH_4 as the final product. This process can occur under extremely reduced conditions only with an ample supply of organic carbon, which is one of the typical conditions of many wetlands (Fig. 3). Accordingly wetlands represent the greatest source for CH_4 globally, contributing 50% of the total emission from natural sources, including rice paddies (Aselmann and Crutzen 1989). Another important microbial process is CH_4 oxidation (Mitsch and Gosselink 2007). A specific type of bacteria named "methanotrophic bacteria" can oxidize CH_4 to acquire energy under aerobic conditions. Methane produced in deeper parts of wetlands is released near the surface where it can be converted to CO_2 by methanotrophic bacteria. In most wetlands, methane can be emitted via bubbles (called 'ebullition'), diffusion through water, and flux through vegetation.

Wetland plants also play a key role in CH_4 production and emission from wetlands (Mitsch and Gosselink 2007). First, they provide organic carbon needed for CH_4 production as litter or root exudates. Second, plant roots release oxygen into wetland sediments, which can inhibit methanogenesis. Last, plants can function as a conduit for CH_4 release and thus facilitate the release of CH_4 produced from wetland sediment into the atmosphere.

Dissolved Organic Carbon

DOC is operationally defined as any organic carbon material in water that can pass through a 0.45 μm filter. Due to its small size, DOC often represents carbon that is easily degraded by microorganisms, although it can also be composed of highly recalcitrant carbon, such as phenolics found in peatlands. DOC is important for several reasons: (1) it is an important component of organic energy pathways in ecosystems; (2) it absorbs various materials, including organic pollutants and heavy metals to increase their mobility; (3) the flux of DOC is acknowledged as an important component of the global carbon cycle.

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Ecosystem Processes

33

Dennis Whigham

Contents

Abstract

Wetlands are important and essential components of landscapes because they provide important goods and services to societies. In some instances wetland values are difficult to quantify (e.g., aesthetic and recreational values) while other services that can be quantified (e.g., retention of water to mitigate flood events, recharge of groundwater, carbon storage, removal of pollutants and toxics); even though methods for quantification may not be fully developed. Three ecological processes (primary production, secondary production, decomposition) are key elements in enabling wetlands to support earth's life support systems that provide the global services that benefit humankind as well as directly generating food, fuel, and raw materials that support human needs. While the functional benefits that wetlands provide are widely known and accepted, there remain many challenges to integrating scientific knowledge about wetlands into societal decisions that restore or secure the ecological integrity of wetlands to assure that they continue to freely provide essential goods and services.

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Introduction

For decades it has been widely recognized that ecological processes that occur in wetlands (e.g., primary and secondary production, decomposition) result in important functions as well as goods and services from which societies benefit greatly. The benefits that wetlands provide to society have, in fact, been the primary driving force behind many efforts to conserve and restore wetlands (e.g., Maltby and Marker 2009), including international efforts such as the Ramsar Convention on Wetlands of International Importance (http://www.ramsar.org/cda/en/ramsar-home/main/ramsar/1_4000_0__) and the inclusion of wetlands such as the Sundarbans National Park in India in the list of UNESCO World Heritage Sites (http://whc.unesco.org/en/list). The Ramsar Convention now includes more than 2,160 sites covering more than 205.7 million hectares of wetlands. The fact that wetlands provide important functions at local, regional, and international scales (Larson and Mazzarese 1994) has resulted in a wide variety of efforts designed to assess the condition and restoration of wetlands (e.g., Maltby and Barker 2009) as well as provide for their protection, including geographically extensive wetlands that had been negatively impacted by human activities (e.g., Florida Everglades, Mesopotamian wetlands).

The number of functions that wetlands provide is large (e.g., de Groot 1992), but they can be organized into categories that can be used to direct attention to the ecological processes that are important determinants of the structure and function of wetlands. Keddy (2000), for example, organized the functions described by De Groot into four categories as follows:

- **Regulation functions** which include the ecological processes that enable wetlands to, in part, provide the Earth's life support systems (e.g., regulate carbon dioxide and oxygen in the atmosphere, regulate the flow and storage of water in catchments, store and recycle human wastes)
- **Carrier functions** which provide space and substrates for services such as nature protection, cultivation of wetland-dependent crops, and the location of settlements such as the long history of human occupation of wetlands associated with the Tigris and Euphrates rivers (Heyvaert and Baeteman 2008)
- **Production functions** such as the generation of food, fuel, and raw materials for human uses
- **Information functions** such as spirituality, heritage, and scientific research

The ecological processes described here are responsible for functions in all four categories, but they are key elements of the **regulation** and **production** categories identified by Keddy. They are primary production, secondary production, and decomposition.

Primary and Secondary Production

The capture of solar energy through the process of photosynthesis is the basis for almost all ecological functions on Earth, including many of the goods and services that are provided by wetlands. Primary production has two components: gross primary production (GPP) and net primary production (NPP). NPP is the most commonly cited measure of primary production because the simplest way to measure production of wetland plants is to harvest and weigh the vegetation in wetlands dominated by herbaceous species or by measuring trees in wetlands dominated by woody species to estimate biomass (Whittaker and Marks 1975). In wetlands dominated by herbaceous or woody species, direct measurements or estimates of biomass is the metric that is used as a measure of NPP.

Primary productivity of wetlands, both GPP and NPP, varies widely depending on a variety of factors, but it has been documented on numerous occasions that wetlands are among the most productive ecosystems on Earth, usually benefitting from an adequate supply of nutrients and water. A compilation of productivity data in which wetlands are compared with other ecosystems has been provided by Cook et al. (2009). The table provides a broad summer of patterns of wetland productivity across climate zones. For each type of wetland listed in the table (bogs, fens, swamps, marshes, floodplains), productivity on a yearly basis decreases from the tropics toward to poles, ranging from a high NPP of 1,500–4,000 g m^2 $year^{-1}$ for marshes in the tropics to 100–300 g m^2 $year^{-1}$ for bogs and fens in the polar region. This pattern is consistent with the notion that GPP and NPP are higher in warmer climates but the rates are highly variable from one geographic location to another and they depend on the availability of resources that are required to support photosynthesis. Annual rates of primary production, while informative for comparative purposes, can also be somewhat misleading because wetlands that have low annual production can be highly productive on a daily basis. Wetlands in arctic and alpine habitats can have, for example, daily rates of production that are very high due to warmer temperatures and almost continuous light in the summer growing season.

Examples of publications that have focused on wetland primary production can be found in review articles (e.g., Whigham et al. 1978; Hopkinson et al. 1980; Nixon 1980) and books (e.g., Lieth and Whittaker 1975; Lieth 1978). Typically estimates of NPP are lower than the actual NPP because few investigators have measured how much of the NPP was allocated to belowground biomass. It follows that because most of the biomass in the majority of wetland plant species occurs belowground (Whigham and Simpson 1978; Darby and Turner 2008), that actual NPP should be higher than the values reported in the examples cited. Belowground production is, however, very hard to measure because of the difficulty in determining how much of the biomass accumulated during a single growing season compared to the total amount of biomass that is present. As a result, there have been very few estimates of belowground biomass production in perennial wetland plants compared to many more studies that describe the distribution of belowground biomass. Two examples of wetland annual plant species (Whigham and Simpson 1977; Sickels and Simpsons 1985) suffice to demonstrate that the amount of NPP allocated to belowground biomass is significant. Wild rice (*Zizania aquatica* var. *aquatic*) and giant ragweed (*Ambrosia trifida*) are both annual species that occur in tidal freshwater wetlands, one of the most productive types of wetlands in the world (Whigham 2009). Using annual species to demonstrate the relationship between above- and belowground biomass production is useful because all of the belowground biomass in annual species is formed in the same growing season as the aboveground biomass. In the two examples of annual species, wild rice NPP was estimated to be 1,453 g m^2 $year^{-1}$, and about 1/3 was root biomass. Giant ragweed NPP, in the same wetland in which wild rice was studied, was even higher (6,574 g m^2 $year^{-1}$) with root biomass equal to almost half (1,500 g m^2 $year^{-1}$ for roots vs. 3,265 g m^2 $year^{-1}$ for shoots) of the annual production.

Secondary production is a measure of how much NPP is consumed directly or indirectly by animals and turned into biomass of animals, microbes, etc. In the vast majority of wetlands, only a relatively small percentage, a universal average estimate is 10%, of the NPP is consumed directly by animals, and the remainder enters the food webs associated with decomposition, the process described in the next section. There are examples, however, of the importance of NPP produced by wetland plants to animal species and communities, both negative and positive.

The migration of vast numbers of waterfowl and within-region migration of ungulate herds to arctic and tropical wetlands is associated with wetland production. Primary production of arctic wetland is low on an annual basis, but during the summers with 24-h daylight, the rates of NPP are equal to those of the most productive wetlands. As a result, large amounts of NPP in arctic wetlands are consumed by migratory animals such as greater snow geese (Masse et al. 2001; Jefferies et al. 2009). In tropical environments, the seasonal movement of animals into wetlands to consume NPP during the dry season is critically important (Keddy 2000; Keddy et al. 2009).

Examples of negative impacts on primary production associated with animals are almost always related to changes in the behavior of animals or the consequences of human activities (Jefferies et al. 2009). In tidal freshwater wetlands, wild rice (see above example of wild rice NPP) is an important species as the seeds provide an important resource for migratory birds and mammals. In recent decades, however, populations of Canada geese stopped migrating and established large populations along the Patuxent River (Maryland, USA). The nonmigrating geese had a negative impact on wild rice populations by consuming seedlings in the spring during the period when the adult birds are molting and juvenile birds are too young to fly. Eventually the decline in wild rice reached a point where there was concern that the species would no longer be viable. As a result, changes in goose-management practices resulted in a reduction of the population of nonmigrating geese and subsequent recovery of wild rice populations (Haramis and Kearns 2007).

A similar situation has happened in Europe where changes in land-use practices have resulted in dramatic increases in waterfowl populations, especially geese and swans, resulting in increased damage to vegetation on wintering habitats as well as breeding habitats in the summer arctic breeding ground (Jefferies et al. 2009).

Decomposition

Following senescence, most wetland NPP enters the detritus pool where decomposition results in its conversion to carbon dioxide and other carbon-containing compounds that may be metabolized or become part of the long-term storage of recalcitrant carbon compounds (i.e., hemicelluloses and lignin) that are associated with wetlands. Importantly, decomposing wetland plant biomass contains the nutrients and energy that support a wide range of complex aerobic and anaerobic decomposition-related processes (Megonigal et al. 2004). The decomposition process also is a key component, directly or indirectly, of the primary production process as well as the biological, chemical, and physical processes responsible for the goods and services provided by wetlands.

The fate of the senesced NPP in wetlands depends on many factors such as the presence, absence, or concentration of oxygen in the substrate, the nutrient and oxygen content of water that enters the wetland, the temperature, and the nutrient status of the plant material (Brinson et al. 1981; Vepraskas and Faulkner 2000; Verhoeven 2009). The nutrient status of the plant material varies widely based on the types of plants that are present, but generally, decomposition proceeds faster in wetland dominated by herbaceous species compared to wetland dominated by trees and shrubs (Webster and Benfield 1986). Verhoeven (2009) described four categories of plant materials that enter the decomposer food web: (1) easily degraded materials which disappear in weeks to months, (2) plant cell wall material (cellulose and hemicellulose) that decomposes of over a period of months to a year, (3) lignins and waxes which persist for several years, and (4) polyphenolics with antibiotic properties which influence the rate of decomposition of materials in the first three categories.

A general understanding of the decomposition process is that inorganic- and organic-soluble materials (category 1 above) are released by physical processes or decomposer activity soon after senescence and become available for uptake by plants and microbes, retention in the wetland substrate, or export to other ecosystems. Following the loss of the soluble labile materials, the more recalcitrant materials (categories 2 and 3 above) decompose slowly with compounds that are most resistant to decomposition (e.g., plant parts containing lignin), becoming part of the long-term storage function through microbially mediated processes (Fogel and Tuross 1999). The long-term storage of recalcitrant carbon-based compounds in wetlands is one reason behind the prevailing view that wetlands are long-term sinks for carbon and thus are an important component of efforts to reduce the impacts of climate change (Dise 2009).

The dynamics of nutrients during decomposition are closely tied to the fate of carbon as microbes use the organic material as a source of energy. The fate of nitrogen (N) and phosphorus (P) provides examples of the relationships between the characteristics of the substrate in which decomposition is occurring and the fate of essential nutrients. Both nutrients are required for plant growth, and wetland productivity is typically limited by N, P, or both (e.g., Koerselman and Meuleman 1996; Sundareshwar et al. 2003; Elser et al. 2007).

Nitrogen

The fate of nitrogen is complex, and while all components of the processes are still not fully understood (Megonigal et al. 2004; Verhoeven 2009), the conversion of N from one form to another during the processing of organic matter decomposition by microbes is intimately tied to the physiochemical characteristics of wetland substrates and results in changes in nutrient status and subsequently the services that wetlands provide. The sequence of N-related events is tied to the degree to which wetland substrates are aerobic or anaerobic and numerous processes are involved. When oxygen is present, bacteria and fungi can mineralize small carbon-containing compounds completely to carbon dioxide resulting in the production of ammonium (NH_4). Ammonium can be assimilated by plants to support primary production, or it can follow several pathways that influence water quality.

Ammonium is converted directly to nitrate by nitrification or indirectly to nitrite (NO_2) and then (NO_3) by ammonium oxidation in the presence of oxygen. The pathway from ammonium to nitrate can be reversed, but nitrate can also be converted to dinitrogen gas (N_2) by the process of denitrification. N_2 gas can also result from the oxidation of ammonium or nitrate. Denitrification is an important process in wetlands because N_2 losses to the atmosphere result in a quantitative loss of nitrogen and, potentially, an improvement in water quality. Denitrification, which is limited by the amount of nitrate and oxygen in the substrate, can occur in wetlands that are highly anaerobic because of the presence of oxygenated microzones around plant roots. In those instances, oxygen diffuses from plant roots resulting in mineralization and subsequently denitrification. The ability of wetland plants to assimilate ammonium and nitrate coupled to their ability to oxidize substrates has been successfully employed in the use of constructed wetlands to provide water quality improvement in a variety of landscape and social settings (Vymazal et al. 2009).

The conversion of nitrogen from one form to another may not always have a positive benefit. A product of denitrification is nitrous oxide (N_2O) which has been implicated in the climate change debate (Wuebbles 2009) because it has a long half-life in the atmosphere and contributes to the production of ozone in the trophosphere. Perhaps most important, it is approximately 300 times more effective than carbon dioxide as a greenhouse gas. The role of wetlands as a current and future source of atmospheric N_2O is, however, not clear because the presence of water, a common feature of wetlands, promotes the consumption of NO and N_2O by reducing their diffusion to the atmosphere and increasing residence time in the soil where they can be consumed by microorganisms.

While not a product of nitrogen cycling, another climate-related produce of decomposition is methane (CH_3). In the series of oxidation-reduction reactions that occur in wetlands, methanogenesis is the final process, occurring in substrates with a redox of less than −250 mV. In methanogenesis, bacteria use carbon dioxide as an electron acceptor, and the end product of organic matter decomposition is methane and carbon dioxide. Like N_2O, CH_3 is a strong greenhouse gas. As long as there is oxygen present in wetland soils or water, most of the methane produced in highly anaerobic microsites will be used by methane-oxidizing bacteria, and little will escape to the atmosphere. If there is no oxygen present, however, the wetland could be a source of CH_3 and thus contribute to global warming. The role that wetlands play in contributing to atmospheric CH_3 is currently an area of active research and discussion (Ortiz-Llorente and Alvarez-Cobelas 2012).

Phosphorus

Phosphorus (P) is an essential plant nutrient, and similar to N it also follows many pathways in wetlands during the decomposition process. Unlike N, however, very little P cycling involves an atmospheric component, and the one volatile compound that has been identified (phosphine) is apparently only emitted at very low levels from wetlands (Richardson and Vaithiyanathan 2009). Phosphorus is an important element in terms of the goods and services that wetlands provide because many wetlands are P limited and activities associated with human activities (e.g., discharge of sewage, erosion of soil particles that are rich in phosphorus, application of fertilizers containing P) often result in water quality problems because most wetlands have a limited ability to store phosphorus through plant uptake and physical and chemical storage in wetland substrates.

The primary sources of P for wetlands are atmospheric deposition in precipitation and input from water that enters wetlands, usually as surface water from streams or runoff from adjacent upland watersheds. Within wetlands some of the available P, typically in the form of phosphate (PO_4), is taken up by algae, microbes, and plants, while the remainder is adsorbed and exchanged on inorganic and organic particles. Usually the amount taken up actively by biota is only 10–15%, and the remainder is sequestered in soils or litter (e.g., Richardson 1985). The uptake of P in soil and litter leads to a number of studies in which wetlands were used to remove P, as well as N and other nutrients and pathogens, for water quality improvement (Kadlec 2009).

During the initial stage of decomposition, extracellular enzymes produced by microbes play an important role in assimilating as well as releasing P from labile plant material (Wetzel 1999), and microbes also play an important role in processing organic P in wetlands (Richardson and Vaithiyanathan 2009). The fate of P following its removal by biological and physical processes in wetland substrates is complex and depends on the same set of factors that are also important in the N cycle (the presence or absence of water, presence of absence of oxygen, pH), but unlike N, the transformations that occur in the P cycle are also highly dependent on the presence of elements (e.g., Al, Ca, Fe, Mg) to which P can bind tightly and subsequently enters long-term storage as peat/sediment accretion and rarely is ever again available for plant uptake. The ability of wetland substrates to provide short- or long-term storage of P is primarily dependent on the amount of Al and Fe that is present. When those two elements are in abundance, wetlands have a high absorptive capacity for P. Alkaline soils with a high Ca content (e.g., Florida everglades) also have a high absorptive capacity and provide long-term storage, but there are also limits to the amount that can be absorbed and stored when P-loading rates are high. Peat soils that are low in Al and Fe represent the other end of the absorption-storage spectrum and have low potential for both features.

Linking Wetland Primary Production and Decomposition to Ecological Services

In addition to the fate of N and P in wetlands as a result of the complex interactions that occur during decomposition, there are other elements required to support primary production. In addition to N and P, however, it is worth describing recent efforts that demonstrate that organic carbon, the primary constituent of wetland substrates (Dise 2009), also plays an important role in supporting primary production, especially in coastal ecosystems.

Carbon in wetland substrates occurs in many forms, and the export of dissolved organic and inorganic carbon from wetland results in a distinct carbon signature that affects the optical and chemical characteristics of estuarine water (Tzortziou et al. 2011). Recent efforts to characterize the role of high molecular weight carbon compounds, referred to as colored dissolved organic matter (CDOM), have indicated that it plays an important role in supporting algal production because it is highly photoreactive and more reactive than CDOM exported from upland ecosystems to estuarine waters (Tzortziou et al. 2007 and P. Neale 2014, personal communication).

Other wetland carbon-related issues are known to be important components of the goods and services provided by wetlands. The rate at which wetlands accumulate carbon and other nutrients, to a large degree is a function of the interacting effects of primary production and decomposition, has become an issue of concern related to the ability of wetland to persist when they are exposed to high levels of nutrient and sediment runoff from human activities (Maltby and Barker 2009). Another wetland-related factor that has been recognized to be important is whether or not coastal wetlands will persist as sea levels increase in response to climate change. It seems logical that some of the climatic factors responsible for sea level rise, increased temperatures in response to increased atmospheric CO_2, would benefit primary production of coastal wetland vegetation, but the ability of wetlands to keep pace with sea level rise is complicated by other factors such as sediment loads and increased eutrophication of coastal waters. All of these factors interact to determine how an individual tidal wetland will respond to sea level rise, and the primary question is whether or not the elevation of wetland substrates can keep pace with the rate of sea level rise (Langley et al. 2013). There is currently no clear answer to this question, but the answer ultimately is strongly influences by the rates of primary production of emergent plants and the rates of decomposition of the organic matter produced by the plants.

Two recent experiments shed light on the complexity of these interacting factors. The increase in the surface elevation of a Maryland brackish tidal wetland was almost twice the current rate of sea level rise when the plants were given increased levels of carbon dioxide, but when they were given nitrogen and carbon dioxide, the increase in surface elevation was equal to the rate of sea level rise. The latter result suggests that increased nitrogen resulted in higher decomposition rates, especially of belowground biomass, thus negating the positive effects of elevated carbon dioxide on net primary production. When tidal flooding (i.e., how long plants were flooded and the depth of flooding) was added as a third factor in another experiment, the results were more complicated as the responses of individual wetland species varied with one dominant species benefitting from additional nitrogen and carbon dioxide when the site flooded more frequently and to a deeper depth compared to another dominant wetland species that benefited from additional nitrogen but did not tolerate more frequent tidal inundation.

It is clear that the dominant wetland processes (primary and secondary production and decomposition) are all key components underlying the importance of wetlands from an anthropocentric viewpoint. While there is much more to be learned about each of the processes described here, our understanding of the linkage between wetland function and structure provides clear evidence of the past, current, and future importance of wetland ecosystems. Wetlands are clearly far more important to human cultures than would be indicated by the amount of land and water that they cover. Wetlands will continue to be used to improve water quality, and hopefully wetland restorations and declines in the rate of wetland losses will reduce the impacts of climate change while conserving and restoring biodiversity. Wetlands will continue to provide resources that support human societies (e.g., peat harvested for a variety of cultural purposes as well as providing a carbon-rich fuel in some parts of the world). Wetland substrates formed millennia ago will continue to be sources of fossil fuels upon which modern cultures depend. Still other plant remains that were stored in wetland substrates only later become items of cultural importance (e.g., fossilized animal bones that help understand evolution and past climate changes, human remains that are used to better understand previous cultures). For these and many other reasons not presented here, the processes that provide the goods and services that wetlands provide are key to a sustainable future.

Future Challenges

Knowledge about the structure and function of wetlands has increased enormously in the past several decades, including the public view of wetlands. Historically wetland had been viewed as wastelands that should be destroyed or converted to more appropriate uses (e.g., Stine 2008). Increasing awareness of the benefits that wetlands provide to societies has resulted in international, national, regional, and local laws directed toward the conservation and restoration of wetlands. In the USA and Canada, for example, there are national no net loss of wetland goals.

Wetlands, however, remain at risk around the world, and future challenges persist to assure that the processes that are described in this contribution continue to provide valuable wetland functions on local, regional, national, and international scales. While there are many challenges to more fully understand primary and secondary production and decomposition, it is more important to integrating knowledge about these processes into a more comprehensive assessment of the linkages between wetland processes and the goods and services that wetlands provide at all social levels. In the past, the goods and services that wetland processes are responsible for have been provided to societies at no cost, but the level of human impacts across the Earth has reached the point where the processes are threatened because of habitat loss, eutrophication, and discharge of ever-increasing numbers of contaminants into the environment. The challenges to the scientific community are to and more effectively integrate ecological understanding about natural wetland processes into societies so that economic and political decisions are made that assure the continuance of the natural processes described in this contribution.

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Photosynthesis in Wetlands

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S. Reza Pezeshki

Contents

Abstract

Several environmental factors influence plant photosynthetic process in wetlands, thus, affecting growth, productivity, and overall wetland functioning. Assessment of photosynthetic functioning is critical to the understanding of the range, threshold, and optimal conditions for plant growth in wetlands. In most wetlands, flooding is the predominant environmental factor although in certain wetlands other factors such as periodic drought, salinity, and temperature extremes may also affect plants. This paper focuses on photosynthetic responses to changes in wetland conditions following flooding events. Soil flooding leads to soil anoxia that is accompanied by production of many soil substances, known to be harmful to plants. As a result, wetland plants may suffer periodic stresses leading to physiological dysfunctions and reduced photosynthesis. The contribution of these soil compounds to the overall photosynthetic reductions, as well as plant response mechanisms involved are discussed. Also, additional key references on the topic are provided.

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Introduction

Assessment of plant carbon fixation in response to environmental factors in wetlands is fundamental to the understanding of wetland primary productivity, development, and function. Plant growth depends on photosynthetic functioning, while photosynthesis is affected by the plant's internal and external environment. In wetlands, many environmental factors influence plant functioning including periodic or permanent flooding, periodic drought, salinity, and extreme temperatures. However, soil flooding, the predominant feature of all wetlands, leads to the production of various substances known as "phytotoxins," which are by-products of soil anoxic conditions (see Reddy and DeLaune 2008 and the references cited therein). Therefore, assessment of plant responses to flooding and the associated by-products is critical to understanding of the range, threshold, and optimal conditions for plant growth and productivity in wetlands.

Wetland plants possess characteristics that enable them to survive soil saturation and the soil chemistry related to anaerobic conditions. The literature reveals a range of photosynthetic sensitivities to soil anoxia among wetland species. Initial reductions in net photosynthesis immediately after soil flooding are common among species representing various wetland ecosystems including marshes, forested wetlands, and riparian wetlands. At the whole-plant level, reductions of photosynthetic rate in flooded conditions are attributed to many factors including diffusional limitations due to stomatal closure and to metabolic (non-stomatal) inhibition of photosynthesis (Pezeshki 2001). Soil anoxic condition may lead to photosynthetic reduction due to decreased leaf water potential resulting from root dysfunction, reduced activity of major photosynthetic enzymes, disruption in photosynthate transport, alteration in source-sink relationship, and reduced sink demand (Pezeshki 1994, 2001). Furthermore, while root oxygen deficiency may partially account for the reduction in net photosynthesis, soil phytotoxins produced as by-products of soil flooding may play a major role in photosynthetic sensitivities. Clearly, the high oxygen demand in soil resulting from intense reduction exerts a profound influence on oxygen transport and release to the rhizosphere.

Overall, many wetland plants may suffer short-term to long-term stresses leading to physiological dysfunctions. The so-called "flood-tolerant" plants (hydrophytes) possess features that enable them to tolerate or avoid stresses including anatomical/morphological and physiological adaptations (Fig. 1). Anatomical/morphological adaptations include adventitious roots, stem hypertrophy, aerenchyma tissue formation in root and shoot, lenticels, knees, and pneumatophores (Mitsch and Gosselink 2007). Some examples of these structures include bald cypress' knees and mangrove's pneumatophores that protrude from the soil. Physiological adaptation features include anaerobic respiration, reopening of stomata, and rapid recovery of photosynthetic activity.

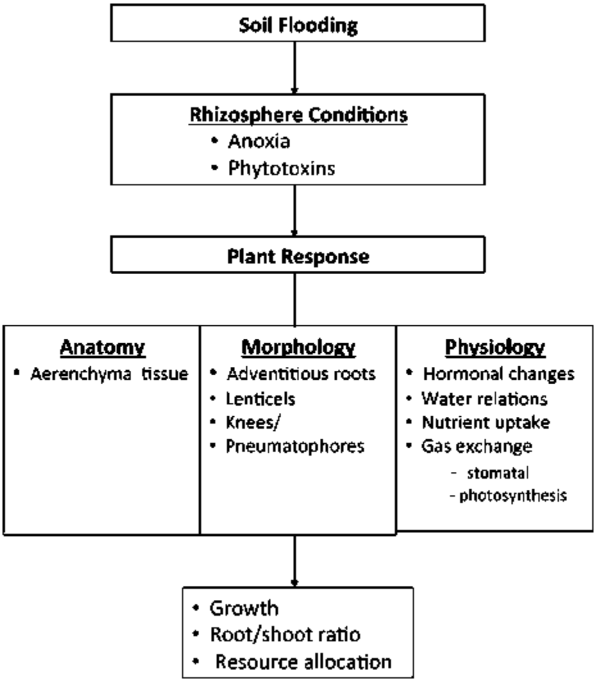

Fig. 1 The relationship between flood-induced changes in soil and rhizosphere conditions, wetland plant functioning, and the potential consequences for photosynthetic carbon fixation and growth

In the following sections, the effects of flooding on soils, the subsequent effects on plant photosynthetic functioning, plant response mechanisms, and the consequences for plant nutrition, photosynthesis and growth are reviewed.

Photosynthesis and Wetland Environment

Life on Earth is dependent on the photosynthetic process, a process by which light energy is converted to chemical energy. Carbon fixed in the photosynthetic process is a significant component of standing plant biomass. CO_2 uptake via the leaf occurs through openings in the leaves known as stomata (also see chapter "▶ Anatomy of Wetland Plants") allowing diffusion in the intercellular spaces of the leaf to the sites of fixation. CO_2 is then fixed via two main pathways known as C_3 or C_4 pathways although C_3 pathway is more common in wetland plants. The designation refers to the first stable compound that is formed in the process, a three-carbon compound in C_3 plants known as phosphoglycerate. In C_4 plants, the first compounds are four-carbon compounds oxaloacetate and malate. Many wetland plants utilize C_3 pathways; however, some wetland plants, many of which found in brackish and salt marshes, are C_4 plants including *Spartina alterniflora*, *S. foliosa*, and *Phragmites australis* (see Mitsch and Gosselink 2007 for details).

Optimal conditions for photosynthesis require, in addition to light and optimal temperatures, adequate supply of nutrients and water. However, in most wetlands, the conditions are less than optimal due to the harsh environment that imposes many unfavorable conditions. Such conditions include flooded soils and the associated anoxic conditions found in many wetlands. Wetland plant roots are the first organ that is exposed to the stress; thus, in the following section, the root environment (rhizosphere) following flooding will be discussed.

Effect of Flooding on Soils

Soil flooding initiates a chain of reactions leading to reduced soil conditions. The reactions encompass various physical, chemical, and biological processes that have significant implications for wetland plants. After flooding is initiated, the limited supply of oxygen is depleted by roots, microorganisms, and soil reductants. Oxygen depletion results in a series of soil chemical changes that include accumulation of CO_2, N_2, H_2, and methane (Mitsch and Gosselink 2007; Reddy and DeLaune 2008). The processes that follow include denitrification; reduction of iron, manganese, and sulfate; and changing soil pH and redox potential (Eh). In a typical series of reductions, NO_3^- is reduced to NO_2^-, Mn^{+4} to Mn^{+2}, Fe^{+3} to Fe^{+2}, and SO_4^{2-} to H_2S, S^{2-} or HS^- (depending upon the soil pH) and accumulations of acetic and butyric acids produced by microbial metabolism (also see ► Chap. 31, "Microbially Mediated Chemical Transformations in Wetlands"). As a result of these transformations, soil redox potential (Eh) becomes progressively more negative. For example, drained soils typically have Eh > +400 millivolts (mV), while waterlogged soils may exhibit Eh levels as low as −300 mV (for details see Mitsch and Gosselink 2007; Reddy and DeLaune 2008).

Soil Flooding and Wetland Plant Functioning

Typically, plants respond to soil physicochemical changes in flooded soil. These responses may lead to a wide range of stress symptoms including root dysfunctions, stomatal closure, and reduced photosynthetic rates (for reviews see Pezeshki 1994, 2001). Flooded soils lead to high competitive oxygen demand, which may affect internal plant tissue oxygen concentration. Other consequences of soil flooding include changes in availability and/or concentrations of nutrients essential for plant functioning and the production of a host of by-product compounds, many are known to be phytotoxic. Examples of such compounds include reduced forms of Fe and Mn, ethanol, lactic acid, acetaldehyde, and aliphatic acids such as formic, acetic, butyric acids, and cyanogenic compounds (for a comprehensive review see Reddy and DeLaune 2008). For instance, excess soil soluble sulfide species are toxic to the roots and are known to inhibit growth of various marsh macrophytes (Pezeshki 2001), while the organic acids have a variety of adverse effects on plants (Reddy and DeLaune 2008).

Anatomical and Morphological Responses of Plants

In many wetland species, an extensive air-conducting tissue (aerenchyma tissue) develop in roots, stems, and leaves in response to flooding. This system allows a plant to transport oxygen to the roots for maintaining aerobic respiration and to oxidize reducing compounds in the rhizosphere. In addition, the internal system of large gas spaces also reduces internal volume of respiring tissues and oxygen consumption, thus enhancing the potential for oxygen reaching the distant underground portions of the plant (Armstrong et al. 1996) (also see chapter "▶ Anatomy of Wetland Plants"). Due to such advantages, the oxygen transport system is an important mechanism to deal with soil anoxia (see Pezeshki 2001 for details and additional references). When flooded, root and rhizomes of wetland species obtain oxygen via gas-phase transport from the shoot system from internal photosynthetic oxygen or atmospheric oxygen. Diffusion, while the major pathway of root aeration in wetland plants, is not the only means of oxygen transport; rhizomes of some species can ventilate via pressurized through flow of gasses (Mitsch and Gosselink 2007).

Despite the reported increase in aerenchyma tissue (and hence porosity) in wetland species in response to flooding, the increase may not be sufficient to satisfy the root respiratory needs for oxygen due to the greater radial oxygen loss rates in reduced soils. For example, continued oxygen stress in the roots of several wetland species was indicated by a rise in alcohol dehydrogenase activity despite the development of aerenchyma tissue (see Pezeshki and DeLaune 2012 and the references cited therein).

Morphological responses include adventitious roots formation on the flooded portions of the stem reported for many wetland plants (also see ▶ Chap. 44, "Wetland Plant Morphology"). These roots play a critical role in supplying oxygen and nutrients to flooded plants as the original roots die in response to sudden flooding (Mitsch and Gosselink 2007). These responses have been reported for many woody wetland species (e.g., *Salix* spp.) as well as herbaceous species (such as *Ludwigia* spp.). Removal of adventitious roots leads to reductions in growth indicating the functional importance of these roots. Such roots replace the losses that occur due to the damaged functioning of original roots in response to flooding. Other morphological responses to flooding include lenticels, small pores (openings) that usually form above the water level on the stem and help with internal plant aeration. Lenticels apparently serve as a conduit for air diffusing into aerenchyma tissues and have been reported for numerous wetland woody species.

Stem hypertrophy, knees, and pneumatophores are among the other morphological responses to flooding. Hypertrophy or stem-base swelling caused by collapse or enlargement of cells in the cortex has been reported in many species (Pezeshki 1994). Knees of bald cypress (*Taxodium distichum*) and pneumatophores are formed on some mangroves (*Avicennia* spp.) are examples of morphological responses to flooding. These structures are thought to be important in aeration of the flooded roots, thus help a wetland plant maintain important root functions. For example, bald cypress knee development may be related to the nature of flooding (Pezeshki 1994; Mitsch and Gosselink 2007).

Plant Nutrition in Flooded Soils

Proper nutrition is critical for optimal photosynthesis; however, nutrient availability is adversely influenced by soil flooding and the associated anoxic soil conditions. Despite wetland plant adaptations, various nutritional deficiencies and toxicities may occur. Flooded soil conditions may lead to inhibition of nutrient uptake and transport due to root dysfunction and/or death, while blockages in the vascular and aerenchyma systems may result from tissue damage due to phytotoxin damage (Armstrong et al. 1996; Pezeshki 2001; Pezeshki and DeLaune 2012).

In some wetland species, ion uptake may continue partly because of the internal O_2 supply system, but partial anoxia in roots can reduce solute intake. Nutrient concentrations at toxic levels may be accumulated in tissues due to higher availability of certain nutrients and root dysfunction. For example, during prolonged flooding, soil pH decreases and zinc availability increases leading to increased tissue zinc concentration. Under such conditions, ferric and manganic forms are reduced to soluble ferrous and manganous forms. Thus, tissue Mn and Fe concentrations in flooded plants are greater than those in aerated conditions. Leaf discoloration (bronzing) due to high soluble ferrous iron has been reported in some species (for reviews see Pezeshki 2001; Pezeshki and DeLaune 2012).

Sulfide toxicity may also occur due to soil chemical changes that include reduction of sulfate to sulfide by anaerobic microorganism. Sulfide is considered to be phytotoxic, although, in most cases, wetland plants can oxidize sulfide in the rhizosphere, thus avoiding or minimizing injury to tissues. Nevertheless, there are numerous reports confirming that excess soil sulfide inhibits growth of various marsh species. The soluble sulfide species including H_2S are toxic to the roots (see Armstrong et al. 1996; Pezeshki 2001). The inhibitory effect of H_2S on cytochrome oxidase is disruptive to aerobic respiration, and excess cytosolic Fe and Mn is harmful to enzymatic structures (Drew 1990). The inhibitory effects of elevated sulfide concentrations on leaf photosynthetic capacity have been demonstrated in several wetland species (see Pezeshki 2001 for additional references). The decrease in photosynthetic capacity has been attributed to disruption of light reactions and/or photophosphorylation and alterations in activity of photosynthetic enzymes. Sulfide has been implicated as a factor responsible for decreased plant growth and productivity in several wetland species (Pezeshki and DeLaune 2012).

Plant Water Relations and Photosynthesis

Whole-plant water relation is influenced by soil flooding as root water uptake in flooded soils is slower than under aerated conditions. Under flooded conditions, internal water stress and leaf dehydration may develop in some species due to a decrease in root permeability leading to stomatal closure and decreased transpiration (also see ► Chap. 45, "Physiological Adaptations to Wetland Habitats"). Many wetland species initially close stomata in response to flooding; however, rapid stomatal reopening may occur. The degree of resumption of stomatal functioning

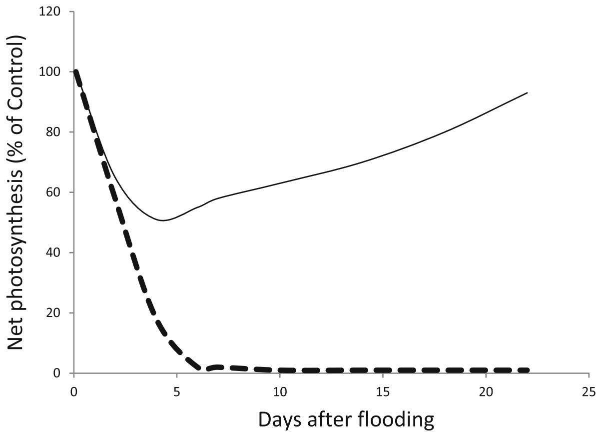

Fig. 2 Effect of initiation and continuation of flooding on photosynthetic activity in two groups of species with different levels of flood sensitivities. Group I represents flood-insensitive species (*continuous line*), while Group II representing flood-sensitive species (*dashed line*). Note the initial reduction in photosynthetic activity for both groups upon initiation of flooding followed by speedy recovery of photosynthetic activity in Group I (despite the continuation of flooding), while little or no recovery is noted in Group II (Data from Pezeshki 1994, 2001)

appears to be dependent on species, duration of flooding, and the intensity of soil reduction (see Pezeshki 2001 and the references cited therein).

Decreases in transpiration and photosynthetic activity have been reported for a number of wetland species in response to flooding. However, substantial differences are found in subsequent photosynthetic responses under continuous flooding among wetland species (Fig. 2). The initial photosynthetic decline has been attributed to diffusional limitations on gas exchange due to stomatal closure and metabolic (non-stomatal) inhibition. However, the contribution of each component to such a response varies across species. In response to flooding, a shift in photosynthetic light-response curves was noted in some species suggesting direct adverse effects on the photosynthetic capacity of the leaves. Photosynthetic recovery is initiated in some wetland species even during prolonged flooding (Fig. 2). Nevertheless, the impact of the apparent delayed and/or slow recovery on plant growth may be substantial due to the initial disruption of carbon fixation and decrease in photosynthate production (Pezeshki 2001).

The adverse initial effects of flooding on stomatal functioning and photosynthetic capacity of many wetland species are well documented (Fig. 3); however, numerous mechanisms may be involved. For example, ethylene has been implicated in the decline of leaf photosynthetic capacity. The effects may be due to loss of photosynthetic capacity of mesophyll. Furthermore, root zone anoxic conditions may lead to reductions of photosynthesis due to decreased leaf water potential, reduced rubisco activity (Pezeshki 1994), disruption in photosynthate transport, alteration in source-sink relationship, or reduced sink demand (Drew 1990; Pezeshki 2001). Other factors may contribute to the reduction in photosynthetic capacity including low

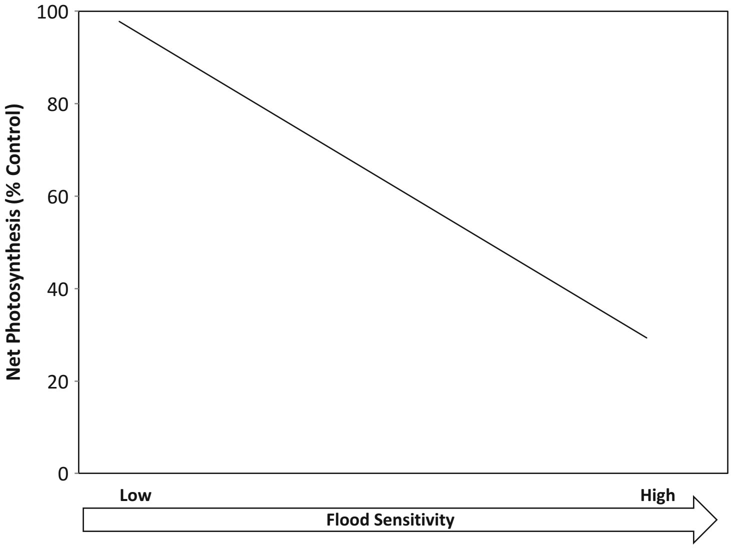

Fig. 3 A generalized relationship between photosynthetic activity and flood sensitivity for wetland plants (Redrawn from Pezeshki 1994)

leaf chlorophyll content and/or leaf chlorophyll degeneration. Leaf chlorophyll content decreased only in some wetland species subjected to flooding (for review see Pezeshki 2001).

As mentioned earlier, reduction in activity of photosynthetic enzymes has been reported in response to flooding including changes in activity of rubisco (see Pezeshki 1994, 2001 for detailed reviews). Stress factors can reduce rubisco activity and/or enhance rubisco degradation in some species. Because activity of this enzyme is critical for photosynthesis, flood-induced reductions of net photosynthesis may be due, at least partially, to decreased rubisco activity. In addition, the recovery of rubisco activity may contribute to the resumption of photosynthesis in some wetland species (Pezeshki 1994, 2001).

Flooding also influences translocation of various photosynthetic products. For example, reduced rates of assimilate translocation to roots has been reported for some species. The effects include low ATP production resulting from disruption of the oxidative phosphorylation (Kennedy et al. 1991), carbohydrate synthesis, transport, allocation, and utilization. Carbohydrate allocation patterns and translocation rates may be a critical aspect of flood tolerance, but this issue requires further research in wetland species.

Growth and Biomass Production

Decreased biomass accumulation often occurs in wetland species in response to flooding. Also, root/shoot ratios often decrease with flooding, because the effect of

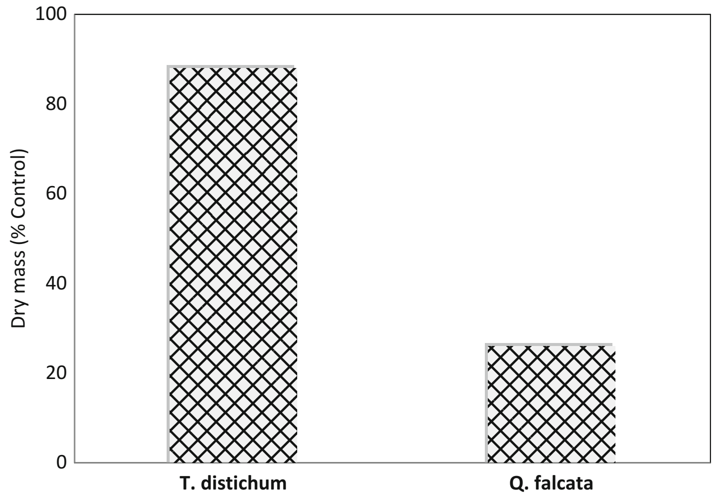

Fig. 4 Effect of flooding on growth (dry mass accumulation) of seedlings of two woody species with different levels of flood sensitivities. *Taxodium distichum*, a flood-tolerant species, and *Quercus falcata* var. *pagodifolia*, a flood-sensitive species. Seedlings were grown in a greenhouse and subjected to non-flooded drained (control) or continuously flooded treatments (Redrawn from Fig. 5, Pezeshki and Anderson 1997)

soil reduction on root growth is often very negative. While a range of responses to soil flooding may occur in a given species, these changes, nevertheless, occur within the context of an overall reduction in biomass accumulation (Fig. 4) as reported for flooded individuals of many species (see Pezeshki 2001 and the references cited therein). For example, root dry weight decreased by 40%, while shoot dry weight reduced by 25% in *Spartina patens* as soil reduction intensified following flood initiation. In general, soil reduction has pronounced effects on root elongation. Similar conclusions were drawn from a study on a flood-tolerant woody species *Taxodium distichum*. Additional data showed cessation of root growth in *Spartina patens* and noted smaller root system under anoxic soil conditions (see Pezeshki and DeLaune 2012 for details). Since roots are major sink for photosynthetic products, such reduction in sink size may in part be responsible for a negative feedback inhibition of photosynthesis, thus reduction in productivity of this species.

The increase in root oxygen loss rates reported under flooded soil conditions may explain the reductions in root growth of several wetland species including *Cladium jamaicense* and *Typha domingensis*. These wetland species produced significantly less biomass under sustained flooding as compared to plants grown in oxidized sediments (see Reddy and DeLaune 2008 and the references cited therein). Root growth is an energy-dependent process requiring oxygen; thus, under flooded conditions, root functioning is affected rapidly because molecular oxygen is required as an electron acceptor for oxidative phosphorylation (Drew 1990). Root elongation was inhibited in some species when soil became anoxic. Root penetration depth was also adversely affected under anoxic soils leading to development of shallow root system different in architecture than those of plants growing in aerated sediments (see Pezeshki 2001 for review).

Future Challenges

Assessment of wetland plant carbon fixation in response to environmental factors is critical to the understanding of wetland functioning. Plant growth depends on photosynthetic process, while photosynthesis in turn is affected by plant's internal mechanisms and external factors. In wetlands, many environmental factors influence plant functioning. These factors alone or in combination include flooding, periodic drought in certain wetlands, salinity in coastal wetlands, and extreme temperatures in some wetlands. However, soil flooding, the predominant feature of all wetlands, leads to the production of various substances known as "phytotoxins" that are by-products of soil anoxic conditions. The contribution of "phytotoxins" to the overall observed photosynthetic reductions in plants in flooded soils, as well as plant response mechanisms, remains as major areas of challenge for the future. In addition, wetland plants are likely to be subjected to additional complex sets of environmental stressors if the predicted future climate changes are realized (also see ► Chap. 2, "Succession in Wetlands"). Such changes include higher temperatures, increased atmospheric CO_2, excess salinity due to sea level rise in coastal wetlands, and prolonged drought in some interior wetlands. These factors alone and in combination pose new challenges for research that should encompass evaluation of plant responses to these predicted scenarios.

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Photosynthetic Measurements in Wetlands 35

S. Reza Pezeshki

Contents

Abstract

Assessment of wetland plant photosynthetic carbon fixation in response to environmental factors is critical to the understanding of wetland functioning. Various approaches including utilization of the methods described in the present paper is essential to the evaluation process. Advances in development of equipment, particularly field portable systems, in recent decades have led to excellent opportunities to conduct research in the field. The present paper focuses on several methods currently available for direct and indirect photosynthetic measurements. The advantages and disadvantages of each technique along with examples of currently available equipment are briefly mentioned while additional key references are provided for the readers to follow the details elsewhere.

Keywords

Wetland plants · Plant productivity · Plant gas exchange · Chlorophyll fluorescence · Wetland functioning

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Introduction

Plant photosynthesis and productivity in wetlands may be quantified by various techniques. For example, periodic harvests of biomass is one way to assess measurements of changes in productivity, which is an important indicator of photosynthetic carbon fixation. Destructive sampling is used for productivity assessment over a few weeks to an entire growing season. This method is a popular approach for assessment of plant productivity and for comparing across a range of environmental conditions in wetlands. In forested systems, nondestructive litterfall traps and wood production are used more often than harvesting to assess photosynthesis. Numerous studies have reported productivity estimates for marshes based on the standing crops obtained by various harvest methods (see Cronk and Fennessy 2001 for a detailed review).

There are, however, shortcomings associated with harvest methods. For example, the technique usually incorporates changes in live and dead materials during different intervals; however, harvest methods do not account for the physical removal of organic matter between harvests or loss through decomposition leading to an underestimate of productivity. Other harvest approaches to measurement of net primary productivity of emergent herbaceous wetland plants may overestimate or underestimate productivity. Furthermore, the technique is not appropriate when evaluation of short-term plant responses, i.e., hours or a few days, to changes in the environment is required. In such cases, the measurement of photosynthetic carbon fixation allows nondestructive, direct assessment of plant functioning and extrapolation of the data to show the potential for dry weight accumulation.

Quantifying photosynthetic carbon fixation is a useful approach for assessment of plant productivity. In the past, this technique was not a preferred method in the field because of technical difficulty in operating complex equipment, which required a power source and stable temperature for operation of the equipment. Using this equipment in wetlands necessitated the use of portable laboratories as was the case for other ecosystems, such as terrestrial ecosystems (Field et al. 1992). Additional advances made in building battery-operated, light-weight portable photosynthetic systems facilitated the use of photosynthetic measurements in productivity assessment in wetlands.

There are a number of methods available for quantifying plant photosynthetic responses at leaf, branch, whole-plant, and canopy levels. These techniques include measurement of oxygen (O_2) exchange, stable- isotope method, micrometeorological technique (eddy correlation), chlorophyll fluorescence, and the efficiency of Photosystem II (PSII) method, simple chamber with remote carbon dioxide (CO_2) analyzer, and portable chambers using onboard infrared gas analyzers (IRGA). This section focuses on these techniques and their utilization for measuring photosynthesis in wetlands. The advantages and disadvantages of each technique along with examples of currently available equipment are briefly discussed (see Pezeshki 2013 for additional details).

Measurement Techniques

Oxygen Exchange Measurements

Photosynthesis involves conversion of light energy to chemical energy by chlorophyll. During a series of reactions, CO_2 is taken up while O_2 is evolved from the plant. Thus, photosynthesis can be measured as either CO_2 uptake or O_2 evolved. The O_2-exchange measurement technique is based on the principle that a leaf enclosed in a chamber and supplied with carbon dioxide produces O_2 upon illumination. The accumulated O_2 in the chamber is measured using a Clark-type O_2-electrode. The measurements are then converted to CO_2 uptake per unit leaf area per unit time (such as μmol CO_2 m^{-2} leaf s^{-1}) through a series of calculations. The measurement is relatively simple using commercially available instruments.

The advantages of using this technique include the availability of relatively inexpensive portable units that can be used in the laboratory as well as in the field. The major limitations include the need for destructive sampling as well as many of the limitations due to the use of small leaf cuvettes including extrapolation of data to whole-plant and canopy level carbon fixation assessment.

Carbon Isotope Technique

Use of CO_2-labeled with a carbon isotope (^{14}C) for photosynthetic research is well established in the literature. A leaf that is exposed to air containing $^{14}CO_2$ incorporates ^{14}C into photosynthetic products. Thus, the technique allows assessment of photosynthetic carbon fixation. In a typical system, a cylinder that contains a known $^{14}CO_2/^{12}CO_2$ gas mixture is connected to a chamber. A leaf or plant part is exposed to the gas in the chamber for a brief period of time and measured precisely. The tissue is then removed, frozen in liquid N_2, and processed for determination of $^{14}CO_2$ content using a liquid scintillation counter. There are different systems that are specifically designed for laboratory or field application (see Field et al. 1992).

This technique is a useful approach; however, its limitations include the need for destructive sampling that is followed by significant laboratory tissue analyses adding to the complexity of the methodology as well as limitations associated with extrapolation of data from small leaf samples to whole-plant and canopy levels.

Micrometeorological Techniques

Carbon dioxide fixation by a stand of plants can be measured over a large area using micrometeorological techniques that do not require destructive sampling or any enclosure. Accordingly, the CO_2 exchange rate is quantified using air movements and CO_2 concentrations above a stand of plants. The technique known as eddy correlation (also referred to as eddy covariance) is an indirect measure of ecosystem productivity. Eddies are defined as layers of air that move as units carrying various gases such as CO_2 into and out of an ecosystem; thus within these units, gas concentrations are uniform. Once an eddy passes over a canopy surface, CO_2 diffuses from the eddy into the photosynthesizing canopy. Because photosynthesis removes CO_2 from the air immediately surrounding the plant tissue, the adjacent air layer is replaced with ambient CO_2 level. Therefore, the technique allows measurement of net CO_2 transfer into or out of an ecosystem.

The technique requires extensive equipment such as infrared gas analyzers (IRGA), computers, and data loggers to allow continuous data collection on the CO_2 gradient and airflow for determination of the photosynthetic rates. Thus, the rate of CO_2 uptake by plants is assumed to be equal to the rate of CO_2 supplied from the ground below the canopy and from the atmospheric air above the canopy (Field et al. 1992). Clearly, utilizing eddy correlation technique to assess wetland productivity can provide useful information, and it has already produced such information in other ecosystems (see Goulden et al. 2004).

The disadvantages of using the micrometeorological technique include lack of control over environmental factors and the expense of an elaborate field set up to allow continuous and concurrent measurements of air flow, CO_2, and other parameters. Also, the productivity measurement obtained is not a direct estimate of net primary productivity because it includes contributions from all CO_2 sources and sinks within an ecosystem. Nevertheless, the technique is very useful in the assessment of community level carbon fixation.

Chlorophyll Fluorescence Method

The usage of chlorophyll fluorescence to measure Photosystem II (PSII) efficiency is based on the principle that light energy absorbed by chlorophyll molecules are used to: (1) drive photosynthesis (referred to as photochemistry), (2) form excess energy that is dissipated as heat, or (3) reflected as light (chlorophyll fluorescence). The three processes are competing processes, i.e., an increase in the efficiency of one would lead to decrease in other processes. Thus, by measuring the yield of chlorophyll fluorescence, information about changes in photochemistry and heat dissipation is obtained (see Maxwell and Johnson 2000; Lichtenthaler et al. 2005, and Henriques 2009 for detailed reviews and additional references).

In recent years, advances have led to the design of “modulated fluorometers.” These systems allow assessment of PSII in the presence of other light sources including sunlight. Modern modulated fluorometer units from various manufacturers are portable and can be used in the field, with subsequent chlorophyll fluorescence data downloaded and analyzed.

The chlorophyll fluorescence measurements yield useful information on photosynthetic performance providing insights into plant stress and tolerance mechanisms. The technique is most useful for early detection of stress before visible effects become apparent. Although measurements of chlorophyll fluorescence can provide critical information on light energy absorption and conversion that occur in photosynthesis, direct extrapolation from these data to the carbon fixation of an individual plant requires caution. For example, the technique does not allow valid comparison between control and stressed plants when visible leaf color changes occur in response to a given stress (see Henriques 2009 for details and additional references).

Photosynthetic Measurements Utilizing Chambers

Plant carbon fixation studies utilizing chambers encompass a range of enclosures such as miniature cuvettes for leaf-level measurements to large tree enclosures. The rate of CO_2 fixation by the leaves in a chamber is calculated from the CO_2 concentration difference between the incoming air prior to exposure to plant material and the outgoing air after exposure to the plant.

Simple Chamber Technique

Use of a simple chamber for measurements of photosynthetic activity using a remote gas chromatograph (GC) or infrared gas analyzer (IRGA) includes a system for photosynthetic measurement in a wetland (see Reddy and DeLaune 2008 for details). This approach is based on the principle that the rate of CO_2 fixation by the leaves in a chamber is calculated from the CO_2 concentration changes in the chamber over time. In the dark chamber, only respiration is occurring resulting in an increase in CO_2, whereas in the light chamber both photosynthesis and respiration occur at the same time. Thus, utilizing both chambers provides an estimate of photosynthetic rate after adjusting for respiration. This approach is a useful method to study plant photosynthesis in wetlands. The system is relatively simple, easy to build, and economical. Thus, if replicated over a reasonable area of marsh and utilized carefully, it can produce useful comparative estimates of wetland productivity.

Portable Gas Exchange Systems Using Infrared Gas Analyzers

This approach is based on the principle that the rate of CO_2 fixation by the leaves in a chamber can be calculated from the CO_2 concentration changes in the chamber over time. This photosynthetic system includes one or more Infrared gas analyzers (IRGA). Because CO_2 is a strong absorbent of infrared radiation within a certain range, IRGA can measure CO_2 concentration of the air passing through the system. The leaf chambers are designed to control the internal chamber's environment including airflow, temperature, relative humidity, CO_2 concentration, and light level.

Modern photosynthetic systems have built-in automated data acquisition and recording. These devices allow the researcher immediate access to the raw and processed data of measured and/or calculated parameters, which can be downloaded to a computer. In addition, the system also measures plant transpiration to calculate stomatal conductance (g_{wv}) for determining the internal leaf CO_2 concentration (C_i). Calculation of C_i is important because it represents the availability of CO_2, the primary substrate for photosynthesis. Thus, C_i data is critical for determining the limitations to photosynthetic activity due to reductions in diffusion caused by stomatal closure (stomatal limitation) versus the biochemical limitations (nonstomatal limitation) that result from changes in photosynthetic efficiency (see Woodrow et al. 1987 for details).

In wetlands, portable gas exchange systems may be utilized to study plant responses to changes in environmental conditions (Pezeshki 2001). In addition, this method can also be used to assess wetland plant health and functioning and to estimate wetland productivity. Nonetheless, these systems provide leaf-level data. As such, one must make many assumptions to extrapolate the data to the whole-plant, population, and community levels. However, leaf-level measurements are useful fir understanding of the whole-plant level, particularly if coupled with measurements that represent carbon fixation above the whole-plant level. Photosynthetic measurement at higher levels such as the whole-plant or canopy is possible using whole-plant chambers or other techniques mentioned previously. Canopy chambers and micrometeorological techniques can provide information related to the effect of species composition on productivity although each of these techniques has limitations as mentioned in previous sections.

Future Challenges

Assessment of wetland plant carbon fixation in response to environmental factors is critical to the understanding of wetland productivity and functioning. Such assessment requires the utilization of various approaches including the use of one or more of the methods described above. Continuous advances have been made in developing instruments that are portable and useful in the field. However, many challenges remain since the majority of the techniques provide leaf-level data, which are difficult to extrapolate to the whole-plant, population, and community levels. Canopy chambers and micrometeorological techniques can provide information related to the effect of species composition on productivity although each of these techniques has limitations as well. Additional advances in techniques are needed to improve the accuracy and the overall assessment at the community and ecosystem levels.

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36 Primary Production and Respiration: Ecological Processes in Wetlands

M. Siobhan Fennessy and Julie K. Cronk

Contents

Abstract

Wetland primary production is a measure of ecosystem metabolism. It quantifies the amount of carbon fixed in the process of photosynthesis and released in the process of respiration by the plants and algae (photosynthetic organisms) in the ecosystem. At the ecosystem level, the total amount of carbon captured and stored via photosynthesis is known as gross primary production, or GPP. Respiration is the cellular process in which the chemical energy contained in organic compounds is converted to useful energy (in the form of adenosine triphosphate, ATP) to maintain cellular activity. The balance of these two processes represents the net carbon fixed into organic compounds, or net primary production (NPP), which can be used to build biomass.

Keywords

Autotrophs · Cellular metabolism · Gross primary production (GPP) · Gross primary productivity · Hydrology · Net primary production (NPP) · Net primary

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productivity · Photosynthesis and respiration · Primary productivity · Submerged photosynthetic organisms · Wetland primary production · Algal primary production · Autotrophs · Factors · Gross primary production · Measurements · Photosynthesis and respiration · Studies · Submerged photosynthetic organisms · Subsidy-stress model · Wetland ecosystem types

Introduction

Wetland primary production is a measure of ecosystem metabolism. It quantifies the amount of carbon fixed in the process of photosynthesis and released in the process of respiration by the plants and algae (photosynthetic organisms) in the ecosystem. At the ecosystem level, the total amount of carbon captured and stored via photosynthesis is known as gross primary production, or GPP. Respiration is the cellular process in which the chemical energy contained in organic compounds is converted to useful energy (in the form of adenosine triphosphate, ATP) to maintain cellular activity. The balance of these two processes represents the net carbon fixed into organic compounds, or net primary production (NPP), which can be used to build biomass. NPP can be calculated as:

$$\mathrm{NPP} = \mathrm{GPP} - \mathrm{R}$$

Information on primary production is essential to understanding the functions and ecosystem services that wetlands provide (Cronk and Fennessy 2001), and data on primary production are often the basis for quantitative studies of other ecosystem processes. For example, knowledge of primary production can provide insight into the trophic dynamics of an ecosystem as well as the cycling and retention of plant nutrients. Primary production leads to the provision of many ecosystem goods, such as fish and shellfish, timber, biomass fuels, fiber, and pharmaceuticals (Daily 1997). In addition, primary production affects soil oxygen levels and can be an important factor in determining oxidation-reduction conditions in wetland sediments.

The organisms that carry out photosynthesis are known collectively as primary producers, or autotrophs. Photosynthetic organisms of all types play a number of vital roles in wetland ecosystems. The following is a summary of their contributions in terms of primary production (Cronk and Fennessy 2001):

- Autotrophs form the base of the food chain and, as such, are the major conduit for energy flow into the wetland biota. Primary production of wetland plant communities varies, but some herbaceous wetlands have extremely high levels of production, rivaling those of tropical rain forests.
- Submerged photosynthetic organisms release dissolved oxygen into the water column, making it available for uptake by other aquatic organisms. They also release carbon compounds, and thereby influence the wetland plant, algal, microbial, and animal communities.
- The photosynthetic organisms of wetlands strongly influence water chemistry, acting as both nutrient sinks through uptake, and as nutrient pumps, moving

compounds from the sediments into the biomass. They improve water quality by taking up nutrients, metals, and other contaminants, and by providing organic substrates for microbial processes such as nitrification and denitrification.

In addition to their trophic and chemical roles, the vascular plants of wetlands provide critical habitat for other taxonomic groups, such as bacteria, epiphyton (algae that grows on the surface of plants), macroinvertebrates, fish, and birds. The composition and diversity of the plant and algal communities have a strong influence on the overall diversity of the wetland faunal community. In addition, plants influence wetland hydrology and sediments by stabilizing shorelines, and modifying currents. They also influence peat accumulation, shade the water column thus stabilizing water temperatures, and affect water cycling via evapotranspiration.

Cellular Metabolism: Photosynthesis and Respiration

The conversion of solar energy into stored, chemical energy in plant cells is known as photosynthesis. While the pathway that converts carbon dioxide and water into sugars and oxygen is a complex process, the overall reaction can be expressed in this chemical equation:

$$6CO_2 + 12H_2O \xrightarrow[\text{chlorophyll}]{\text{light energy}} C_6H_{12}O_6 + 6O_2 + 6H_2O$$

The sugars produced in this process provide both energy and the building blocks for larger biological molecules that form plants' structures. The oxygen is released into the air or water and can be taken up by other organisms in aerobic cellular respiration. Photosynthesis can be measured by carbon dioxide assimilation or oxygen production. Algal and submerged plant production can be estimated by measuring the dissolved oxygen levels in the water column. Production changes throughout the day as well as seasonally due to light, carbon dioxide, and nutrient availability.

Respiration is the cellular process in which chemical energy contained in biomolecules (sugars, lipids, and proteins) is oxidized. Cells convert the released energy into a chemical form (ATP) that is directly usable by cells. The chemical equation for the aerobic respiration of glucose is essentially the reverse of the equation for photosynthesis. While photosynthesis is limited to periods of light, respiration takes place at all times.

Aerobic cellular respiration can be measured by carbon dioxide evolution or oxygen consumption by a plant or portion of a community. For example, researchers can enclose the sample in a chamber and measure the changes in gas concentrations in the air or water that surrounds it. Respiration rates can be quite high and can "undo" a large portion of the carbon fixed in photosynthesis. In a review of primary production in wetlands, the average rate of respiration in emergent wetlands was estimated to amount to 72% of GPP (Brinson et al. 1981). As with photosynthesis, respiration rates are highly variable and change with temperature and the availability of water, oxygen, and energy compounds.

Measuring Primary Production

Primary production is typically measured as the amount of biomass produced per unit area, where biomass is the mass of plant material. The term, *primary productivity*, refers to primary production over time, or the rate of primary production. If gaseous exchange methods are used to measure primary productivity, the time period is a day or an hour and the units are grams of oxygen evolved or carbon assimilated. In wetland plant studies, results are usually given in units of dry plant matter produced per unit area per year (g dry weight m^{-2} $year^{-1}$). In the temperate zone, growth per year is actually growth during the growing season.

Net primary productivity is a measure of the observed changes in plant biomass over a time period. *Gross primary productivity* includes both net primary productivity and all of the predatory and non-predatory losses (respiration) from the plant divided by the time interval. Net primary productivity is the value most often reported in wetland plant production studies and it is usually reported in units of grams dry weight per square meter per year (g m^{-2} $year^{-1}$); this allows for easy comparison between systems.

Often, measurements of whole wetland primary production reflect a combination of measures because several plant forms coexist within a single wetland. These forms include algae, submerged and emergent plants as well as trees and shrubs. Each component of the community requires different methodology. For example, harvested, cleaned, and dried herbaceous plants can be weighed to gain a measure of biomass. Algal productivity is often measured by oxygen production while tree growth is often measured as a change in trunk diameter. All of these measures need to be converted to the same units and combined in order to reflect whole system productivity.

Measurements of productivity may be single point measures, giving a synoptic view of the ecosystem, or for a more meaningful analysis of productivity, repeated or sequential measures are required. Sequential measures of biomass enable more accurate estimates of net primary production. If sequential measurements are taken, the change in biomass (ΔB) can be calculated as:

$\Delta B = B_2 - B_1$ Change in biomass over the time interval, $t_1 - t_2$

where:

B_1 = Biomass of a wetland at time 1, t_1
B_2 = Biomass at time 2, t_2

If plant losses to mortality or leaf fall (L) and plant losses to grazing (G) can also be measured, then this information can be combined with the change in biomass to estimate net primary production for the given time period:

$$NPP = \Delta B + L + G$$

Table 1 Primary production rates for a range of wetland ecosystem types (From Cronk and Fennessy 2001)

Wetland type	Net primary production g dry weight m^{-2} $year^{-1}$
Salt marsh	130–3700
Tidal freshwater marsh	780–2300
Freshwater marsh	900–5500
Mangrove	1270–5400
Southeastern U.S. bottomland hardwood	830–1600
Cypress swamp	200–1540
Forested northern peatland[a]	260–2000
Non-forested northern peatland[a]	100–2000

[a]Includes above- and belowground production

In all wetland primary productivity studies, detailed descriptions of methods used are needed so that the results can be compared among wetlands.

Wetland Primary Production Studies

The primary production of many wetlands is relatively high, especially when compared to other types of ecosystems or even to highly managed agricultural lands. As Keddy (2000) points out, wetlands do this without the inputs of fossil fuels (in the form of fertilizers and machinery) or irrigation water. They are the workhorses of the landscape, producing biomass and oxygen that support adjacent ecosystems and other species. Many variables influence primary production, such as the type of wetland, species composition, hydrology, climate, and other environmental variables such as soil type and nutrient availability. Table 1 shows a general comparison of mean rates of primary production for different types of wetlands. Note that the range of values within each type is indicative of the wide variation of conditions and climates in which these general categories of wetlands can be found. It is also a reflection of different methodologies, with higher numbers usually indicating not only higher productivity but also combined and sequential measurements.

The length of the growing season and temperatures decrease with increasing latitude, and these factors have a substantial influence on productivity. In boreal wetlands and peat bogs, productivity rates typically range from 100 to 1000 g m^{-2} $year^{-1}$ (Cronk and Fennessy 2001), far less than in tropical or even temperate areas.

Two particularly productive wetland types are freshwater and salt marshes, which are dominated by emergent herbaceous vegetation. A typical high value for the aboveground primary productivity of marshes in temperate zones is 2500 g m^{-2} $year^{-1}$ (although higher rates have been reported). The primary productivity of mangrove forests, which are forested wetlands of the coastal tropics, also is quite high. In fact, the primary productivity of marshes and mangrove forests is at the same level as the primary productivity of highly productive tropical rain forests. Overall, primary production in wetlands globally varies by about a factor of 4, with mean rates ranging from about 500 to 2000 g m^{-2} $year^{-1}$ (Mitsch and Gosselink 2007).

Factors that Control Wetland Primary Production

In addition to climate and latitude, other major influences on wetland primary production are hydrology and nutrient availability. Hydrology is a key forcing function in wetlands and has been called the "master variable" that determines ecosystem structure and function. Wetland hydrology describes the timing, duration, and frequency of inundation, and the resulting hydroperiod (pattern of water levels over time) is an important predictor of both the type of wetland that will develop at a specific location and the productivity of that system (Brinson et al. 1981).

In part, hydrology is important because water carries nutrients and sediments that affect production rates (Keddy 2000). Sites that are open to hydrological fluxes with flowing water and a pulsing (variable) hydroperiod tend to have higher production rates, while stagnant water or nonvariable water levels tend to depress them (other factors being equal; Fig. 1). Studies of the relationship between net primary productivity and flooding regime in wetlands led Odum and others (1995) to propose the "subsidy-stress" model that essentially predicts that too little or too much flooding decreases productivity, while moderate levels (e.g., seasonal flooding) increases it (Fig. 2). In permanently flooded, or stagnant wetlands, anaerobiosis may become extreme and nutrient availability altered. In wetlands with highly abrasive flooding, erosion can carry away sediments, plants, and seeds. Those sites that are "open" to intermediate amounts of hydrological flux tend to be the most productive. Water flows also affect carbon storage by regulating primary productivity, the rate of litter decomposition, and the export organic matter. Tidal wetlands are influenced by twice per day tidal flushing as well as freshwater inputs from adjacent uplands. Primary production in tidal marshes tends to be higher where freshwater inputs are greater (Cronk and Fennessy 2001).

The availability of nutrients can also affect primary production in wetlands. Plant and algal growth tends to be limited by either nitrogen or phosphorus. If the concentrations of these limiting nutrients increase, plant growth, up to some limit, will also increase (Wetzel 2001). Nutrient subsidies occur naturally from water inflows to the site, or as a result of human activities such as farming that lead~~s~~ to nonpoint source runoff of applied fertilizer. Wetlands that receive such inputs tend to have higher primary productivity than those that do not. The least productive wetlands are those that receive nutrients only from rainwater, including some cypress swamps and ombrotrophic bogs.

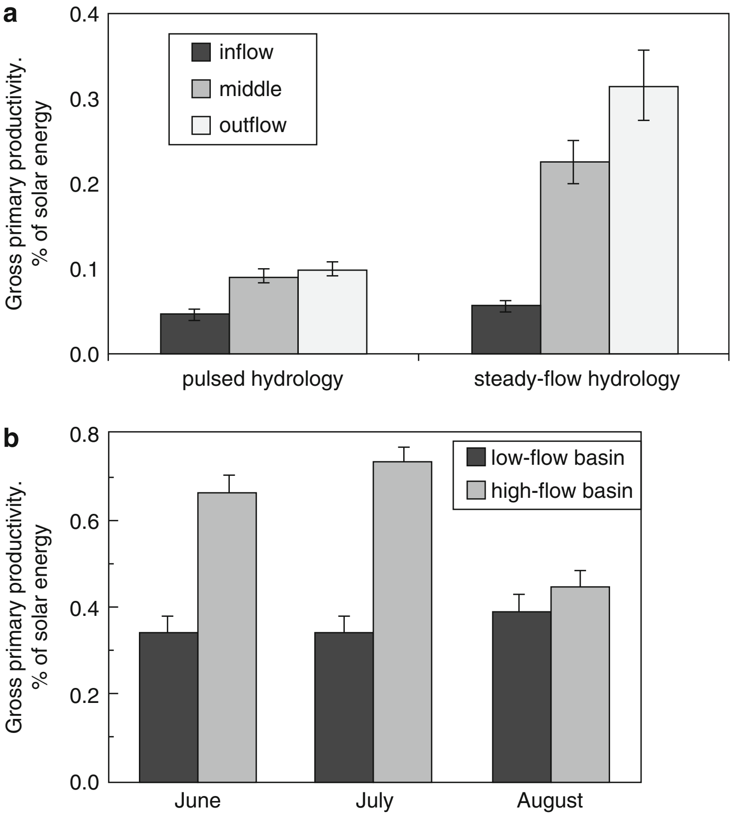

Fig. 1 Differences in algal primary production in the water column as a function of pulsing versus steady state hydrology in experimental wetlands (**a**) the Olentangy River Wetland Research Park in Ohio, and (**b**) the Des Plains River Wetland Demonstration Project (Mitsch and Gosselink 2007). At the Ohio site, pulsed hydrology led to sediment scour and plant damage (abrasive flooding)

Conclusion

Wetland primary productivity varies with climate, hydrology, and nutrient availability and is measured in terms of the amount of biomass produced over a defined time period. Measurements of primary production are vital to understanding the trophic dynamics of wetlands as well as their role in nutrient cycling. They also allow us to compare ecosystem functions among wetlands of all types and between wetlands and other ecosystem types. High rates of primary production support high rates of secondary production and the diversity of higher tropic levels. The provision of ecosystem goods, such as fish and shellfish, timber, biomass fuels, fiber, and pharmaceuticals also depends on the energy harnessed by primary producers

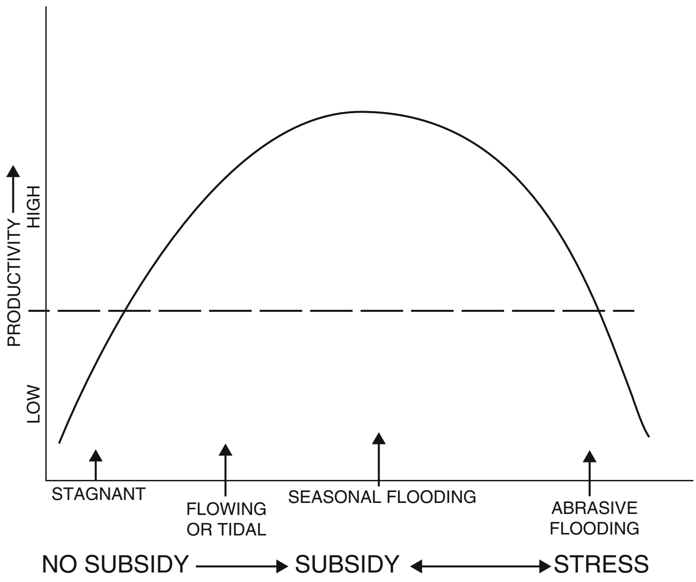

Fig. 2 The subsidy-stress model developed by Odum et al. (1995) showing the pattern of primary production along a hydrological gradient of flooding

(Daily 1997). Understanding primary production enhances our knowledge of the value of wetlands within the landscape.

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Wetland Ecosystem Services

37

Dolf de Groot, Luke Brander, and C. Max Finlayson

Contents

Abstract

Wetlands and the ecosystem services they provide are hugely valuable to people worldwide in many ways: for livelihood, for their biodiversity and existence values and for their economic benefits. Yet many of these services, such as the recharge of groundwater, water purification or cultural values are not immediately obvious when one looks at a wetland and most are public services that are not traded in conventional markets. This chapter gives a brief introduction into the concept of wetlands ecosystems services and their values.

Keywords

Coastal wetlands · Inland wetlands · Millennium Ecosystem Assessment · The Economics of Ecosystems and Biodiversity · Wetland ecosystem services · concept · description and examples · economic valuation

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Introduction

Wetlands and the ecosystem services they provide are hugely valuable to people worldwide in many ways: for livelihood, for their biodiversity and existence values, and for their economic benefits (MA 2005; TEEB 2010). Yet many of these services, such as the recharge of groundwater, water purification, or cultural values, are not immediately obvious when one looks at a wetland and most are public services that are not traded in conventional markets. Also planners and decision-makers at many levels are frequently not fully aware of the connections between wetland condition and the provision of wetland services and the consequent benefits and economic values. This lack of understanding and recognition leads to ill-informed decisions on management and development, which contributes to the continued rapid loss, conversion, and degradation of wetlands – despite the total economic value of natural wetlands often being greater than that of alternative land uses (e.g., Balmford et al. 2002; de Groot et al. 2010; 2012).

Development of the Concept of ecosystem services

The recognition and study of interactions between the environment and human welfare stretches back centuries and includes writings from Roman times on the increase in population and decline in what we now call ecosystem services (De Groot et al. 2010). In 1977, Westman published a paper in Science examining the link between ecological and economic systems entitled “How much are Nature’s Services Worth?” (Westman 1977). Ehrlich and Ehrlich (1981) later coined the term ”ecosystem services” and in the following decade ecologists further elaborated the notion of ecosystems as life-support systems, providers of ecosystem services and economic benefits. But it was not until the late 1990s that the concept received widespread attention with publications by Costanza et al. (1997) and Daily (1997). At the same time, the interdisciplinary field of ecological economics developed the concept of natural capital (Costanza and Daly 1992), which included non-renewable resources, renewable resources, and ecosystem services to demonstrate the significance of ecosystems as providing the biophysical foundation for societal development and all human economic activity (Common and Perrings 1992).

Based on these and other studies, the Millennium Ecosystem Assessment (MA 2005) recognized four categories of services: provisioning (e.g., food, fresh water, wood and fiber, and fuel); regulating (e.g., climate regulation, flood and disease regulation, and water purification); supporting (e.g., nutrient cycling, soil formation, and primary production); and cultural (aesthetic, spiritual, educational, and recreational). As a follow-up to the Millennium Ecosystem Assessment, a study on The Economics of Ecosystems and Biodiversity was carried out from 2007 to 2011 (TEEB 2010). Since then, many national Ecosystem Services assessments have been and are being carried out, and in 2012 the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established.

In spite of all the work done so far, there is still much debate about definitions and classifications and perhaps we should accept that no final classification can capture the myriad of ways in which ecosystems support human life and contribute to human well-being (De Groot et al. 2010).

Description and Examples of Wetland Ecosystem Services

Wetlands provide a broad range of goods and services that are of value to humans. These services are the product of, but should not be confused with, ecosystem processes and functions (Mace et al. 2012). Functions represent the *potential* that ecosystems have to deliver a service which in turn depends on ecological structure and processes (see Fig. 1). For example, primary production (= process) is needed to maintain a viable fish population (= function) which can be used (harvested) to provide food (= service); nutrient cycling (= process) is needed for water purification (= function) to provide clean water (=service). The benefits of these services are manifold, for example, food provides nutrition, but also pleasure and sometimes even social identity (as part of cultural traditions); clean water can be used for drinking, but also for swimming (pleasure) and other activities aimed at satisfying needs and wants. The linkage between ecosystem properties – functions – services – benefits is often presented as a "cascade" (see Fig. 1).

There is not necessarily a one-to-one correspondence between ecosystem functions and services. It might be the case that multiple functions combine to provide a

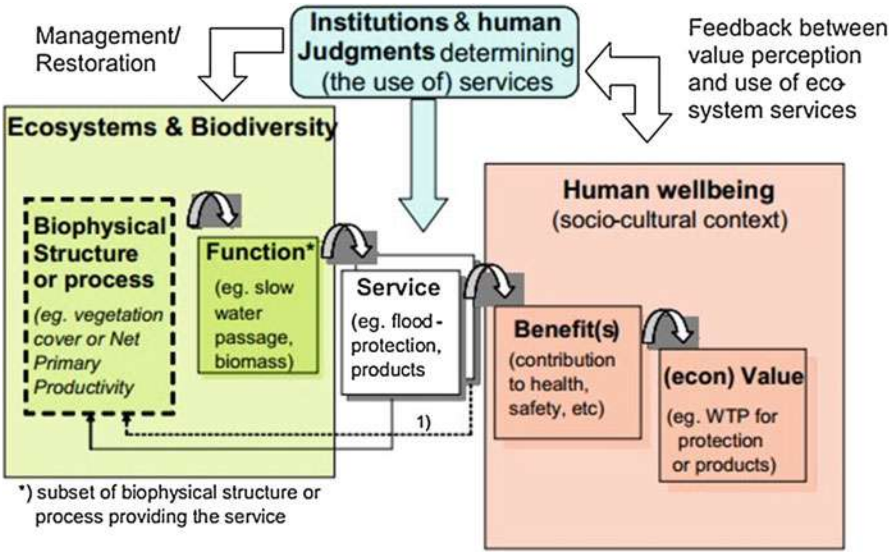

Fig. 1 Linking ecosystem properties to human well-being (source: de Groot et al. 2010; reproduced with permission from United Nations Environment Programme)

single service or that a single function provides multiple services. It is also the case that ecosystem services are combined, to varying extents, with other inputs (e.g., manufactured capital and labour) to produce the services that are consumed by humans. Table 1 lists the main services of inland and coastal wetlands.

Economic Valuation of Wetland Ecosystem Services

The study of wetland ecosystem services is a multidisciplinary undertaking, requiring the understanding of both the ecological provision of services and the socioeconomic benefits that are derived from them. The multidisciplinary study of wetland ecosystem services is, however, challenging due to the differences in paradigms and language across disciplines. In this section, we provide an introduction to the economic understanding of wetland ecosystem services and their values in order to facilitate transdisciplinary understanding.

Economic values associated with wetland ES can be categorized into distinct components of the "total economic value" (see Fig. 2) of wetland natural capital according to the type of use. Direct use values are derived from the uses made of a wetland's resources and services, for example wood for energy or building, water for irrigation, and the natural environment for recreation. Indirect use values are associated with the indirect services provided by a wetland's natural functions, such as storm protection or nutrient retention. Option value is related to the preference, or willingness to pay, to maintain the possibility of future use. The concept of option value includes preferences for preserving an environmental asset for possible future use by current (philanthropic value) or future generations (bequest value). Nonuse values of wetlands are unrelated to any direct, indirect, or future use, but rather reflect the economic value that can be attached to the mere existence of a wetland (Brander et al. 2006).

Some goods and services derived from wetlands may be traded directly in well-functioning markets and therefore have readily observable values. However, due to market failures resulting from undefined property rights or the (quasi-)public good characteristics of some wetland functions, many valuable wetland services may not be traded directly or even indirectly through markets. The term "public goods" describes services for which beneficiaries cannot be excluded from receiving the service and that the level of consumption by one beneficiary does not reduce the level of service received by another. For example, the reduction in downstream flood risk provided by wetlands in a water catchment is received by the entire population living downstream irrespective of their recognition of the service or the number of other beneficiaries. Due to these characteristics ('non-excludability' and 'non-rivalry' in consumption), markets for such services generally do not exist and the potential for private incentives to sustainably manage wetland services is limited. As a result, wetland ecosystem services are undervalued in decisions regarding the conversion of privately owned wetlands to other productive uses that generate marketable goods and services (e.g., agriculture). The lack of understanding of,

Table 1 Ecosystem services provided by (a) inland and (b) coastal wetlands (Source: Adapted from Millennium Ecosystem Assessment (Finlayson et al. 2005))

a. Inland wetlands

Services (Comments and Examples)	Permanent & Temporary Rivers & Streams	Perma-nent Lakes. Reser-voirs	Seasonal Lakes, Marshes & Swamps incl . Floodplains	Forested Wetlands, Marshes & Swamps incl. Floodplains	Alpine & Tundra Wet-ands
Provisioning					
Food: Production of fish, wild game, fruits, etc.	⬤	⬤	⬤	⬤	•
Fresh Water: Storage and retention of water; provision of water for irrigation and for drinking.	⬤	⬤	●	•	•
Fiber, Fuel & other raw materials: Production of timber, fuel wood, peat, fodder, aggregates	●	●	•	⬤	●
Biochemical products and medicinal resources	•	•	?	?	?
Genetic Materials: genes for resistance to plant pathogens	•	•	?	•	?
Ornamental species (eg. aquarium fish)	•	•	?	•	?
Regulating					
Air quality regulation (eg. capturing dust particles)			•	●	
Climate Regulation: Regulation of greenhouse gases, temperature, precipitation and other climatic processes	•	⬤	•	⬤	•
Hydrological regimes: Groundwater recharge/ discharge; storage of water for agriculture or industry	⬤	⬤	●	●	•
Pollution Control & Detoxification: Retention, and removal of excess nutrients and pollutants	⬤	●	•	●	•
Erosion protection: Retention of soils and prevention of structural change (e.g. coastal erosion, bank slumping etc.)	●	•	•	●	?
Natural Hazard mitigation: Flood control, storm protection.	●	⬤	⬤	●	●
Biological regulation: eg. control of pest species and pollination	•	•	•	●	•

(*continued*)

Table 1 (continued)

Cultural & Amenity					
Cultural heritage and identity (sense of place and belonging)	⬤	⬤	●	●	•
Spiritual & artistic Inspiration: Personal feelings and well-being, religious significance	⬤	⬤	●	●	•
Recreational: Opportunities for tourism and recreational activities.	⬤	⬤	●	•	•
Aesthetic: Appreciation of natural features.	⬤	●	•	●	•
Educational: Opportunities for formal & informal education & training.	⬤	⬤	●	●	•
Supporting & Habitat					
Biodiversity & nursery: Habitats for resident or transient species.	⬤	⬤	●	●	•
Soil Formation: Sediment retention and accumulation of organic matter.	⬤	•	●	⬤	•
Nutrient Cycling: Storage, recycling, processing and acquisition of nutrients.	⬤	⬤	⬤	⬤	•

b. coastal wetlands

Services (comments and examples)	Estuaries & marshes	Man-groves	Lagoons (incl. salt ponds)	Inter-tidal flats, beaches and dunes	Kelp	Rock and shell reefs	Sea-grass beds	Coral reefs
Provisioning								
Food: Production of fish, algae and invertebrates	⬤	⬤	•	●	•	•	•	●
Fresh Water: Storage and retention of water; provision of water for irrigation and for drinking	•		•					
Fiber & Fuel & other raw materials: Production of timber, fuel wood, peat, fodder, aggregates	⬤	⬤	●				•	
Biochemical products and medicinal resources	•	•			•			•
Genetic Materials: Medicine, genes for resistance to plant pathogens	•	•	•		●			•
Ornamental species(eg. aquarium fish)	•	•	•					⬤

(*continued*)

Table 1 (continued)

Regulating								
Air quality regulation (eg. capturing dust particles)	•	●	•					
Climate Regulation: Regulation of greenhouse gases, temperature, precipiptation and other climatic processes	●	●	●	•		•	•	●
Hydrological regimes: Ground-water recharge/ discharge; storage of water for agriculture or industry	•		•					
Pollution Control & Detoxification: Retention, recovery and removal of excess nutrients/ pollutants	●	●	•		?	•	•	•
Erosion protection: Retention of soils	●	⬤	•				•	•
Natural Hazard mitigation: Flood control, storm protection	⬤	⬤	•	•	•	●	●	⬤
Biological Regulation: eg. control of pest-species and pollination	●	⬤	●	•		•		•
Cultural & Amenity								
Cultural heritage and identity (sense of place and belonging)	⬤	•	●	⬤	•	•	•	⬤
Spiritual & artistic Inspiration: Personal feelings and well-being, religious significance	⬤	•	●	⬤	•	•	•	⬤
Recreational: Opportunities for tourism and recreational activities	⬤	•	•	⬤	•			⬤
Aesthetic: Appreciation of natural features	●	•	●	●				⬤
Educational: Opportunities for formal and informal education & training	•	•	•	•		•		•
Supporting& Habitat								
Biodiversity & nursery: Habitats for resident or transient species	●	●	•	⬤	•	⬤	•	⬤
Soil Formation: Sediment retention and accumulation of organic matter	●	●	•	•				
Nutrient Cycling: Storage, recycling, processing and acquisition of nutrients	●	●	●	•	•	•		●

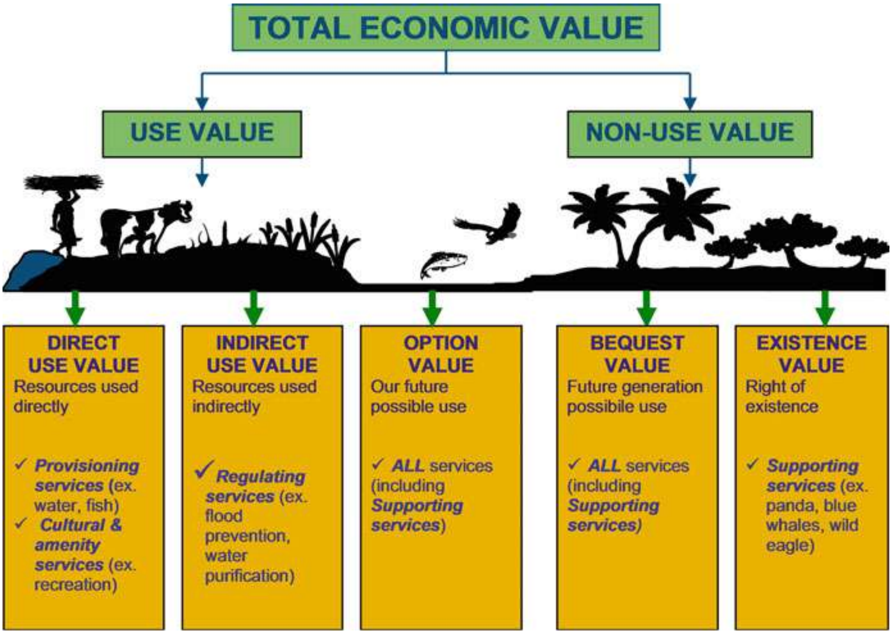

Fig. 2 The Total Economic Value Framework (Adapted from Millennium Ecosystem Assessment (2003), based on a.o. Dixon and Pagiola (1998))

and information on, the value of wetland ecosystem services has also led to their omission in public decisions regarding the conservation of wetlands. Without information on the economic value of wetland services that can be compared directly against the monetary value of alternative public investments, the economic importance of wetlands has tended to be ignored.

In response to this lack of information on the market value of wetland services, there is a large and expanding literature that employs the so-called non-market valuation approaches to estimate the economic value of wetland services. Reviews and meta-analyses of the literature (e.g., by Brouwer et al. 1999; Woodward and Wui 2000; Brander et al. 2006; Ghermandi et al. 2010) show that the estimated values of wetland ecosystem services vary enormously across wetlands with different biophysical and socioeconomic characteristics. Statistical meta-analyses of the literature have attempted to identify factors that help explain this variation to: inform conservation, management, and restoration decisions; enable the prediction of wetland ecosystem service values for policy appraisal or for future scenarios of wetland change; and understand potential trade-offs in the provision and value of wetland ecosystem services.

Based on a large number of case studies, Figure 3 gives an overview of the monetary value of the main services provided by wetlands.

A recent study of 458 value-estimates (De Groot et al. 2012) gives the following (rounded) averages for the Total Economic Value of six main wetland types (on 2007

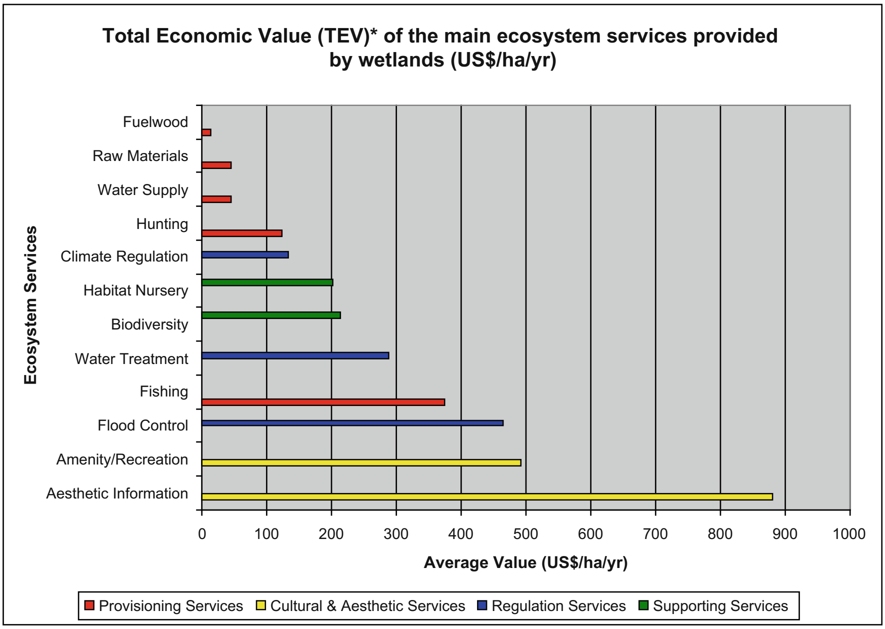

Fig. 3 The Total Economic Value (TEV) of the main ecosystem services provided by wetlands (US$/ha/year). All figures are average global values based on sustainable use levels and taken from two synthesis studies: Schuijt and Brander (2004) (calibrated for 2000), and Costanza et al. (1997) (calibrated for 1994); together covering over 200 case studies (De Groot et al. 2006)

Int.$/ha/year): open ocean 490; coral reefs 350,000; coastal systems (including beaches) 29,000; coastal wetlands (including mangroves) 190,000; inland wetlands 25,000; rivers and lakes 4,300.

Conclusions and Future Prospects

- Following the Millennium Ecosystem Assessment, the Ramsar Convention on Wetlands embraced the concept of ecosystem services and incorporated the benefits obtained by people from wetlands into the Convention's key concept of ecological character.
- The broad classification proposed of ecosystem services by the Millennium Ecosystem Assessment included four categories, namely provisioning (e.g., food, fresh water, wood, and fiber); regulating (e.g., climate, flood and disease regulation, and water purification); supporting (e.g., nutrient cycling, soil formation, and primary production); and cultural (aesthetic, spiritual, educational, and recreational).
- In response to the increasing recognition of ecosystem services, a science-policy platform known as the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) has been established.

- Wetlands provide a broad range of services that are of value to humans. These are the product of ecosystem processes and functions where functions represent the *potential* that ecosystems have to deliver a service which in turn depends on ecological structure and processes. There is not necessarily a one-to-one correspondence between ecosystem functions and services.
- Economic values associated with wetland ecosystem services can be categorized into distinct components of the "total economic value" of wetland natural capital according to the type of use. Direct use values are derived from the uses made of a wetland's resources. Indirect use values are associated with the indirect services provided by a wetland's natural functions, such as storm protection or nutrient retention. Option value is related to the preference, or willingness to pay, to maintain the possibility of future use. Nonuse values of wetlands are unrelated to any direct, indirect, or future use, but rather reflect the economic value that can be attached to the mere existence of a wetland.
- Some goods and services derived from wetlands may be traded directly in well-functioning markets and have readily observable values. However, due to market failures resulting from undefined property rights or the (quasi-)public good characteristics of some wetland functions, many valuable wetland services may not be traded through markets.
- In response to a lack of information on the market value of wetland services, there is a large and expanding literature that employs the so-called non-market valuation approaches to estimate the economic value of wetland services.

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The Economics of Ecosystems and Biodiversity (TEEB)

38

C. Max Finlayson

Contents

Abstract

TEEB is a recent initiative that draws attention to the economic benefits of biodiversity globally. Its objective is to highlight the cost of the loss and degradation of biodiversity, including species and ecosystems. The initiative provides information that can help decision-makers recognize and demonstrate the values of biodiversity and support procedures to incorporate these values into decision-making. A report on water and wetlands identified gaps and inconsistencies in knowledge about the economics of water and wetlands as a base for further investigations.

Keywords

Ecosystem services · Economics · Ecosystems

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History of TEEB

TEEB is a recent initiative that drew attention to the economic benefits of biodiversity globally. Its objective is to highlight the cost of the loss and degradation of biodiversity, including species and ecosystems. The initiative provides information that can help decision-makers recognize and demonstrate the values of biodiversity and support procedures to incorporate these values into decision-making. The information that follows is largely drawn from the TEEB webpage (http://www.teebweb.org accessed 27 August 2016).

TEEB was developed after a meeting of environment ministers from the G8 + 5 countries in Potsdam, Germany, in March 2007. They agreed to initiate an analysis of the global economic benefit of biological diversity, the costs of the loss of biodiversity, and the failure to take protective measures versus the costs of effective conservation. The German Federal Ministry for the Environment and the European Commission (EC) initiated a global study with the support of an Advisory Board and presented an interim report to the Ninth Conference of the Parties to the Convention on Biological Diversity (CBD COP-9) in Bonn, Germany, in May 2008.

The interim report entitled *The Economics of Ecosystems and Biodiversity* (European Commission 2008) collated evidence and examples of economic valuation, identified elements of a bio diversity valuation framework, and emphasized the importance of issues such as ethics in making choices regarding future values. This report stimulated further interest and the production of a series of reports for specific stakeholders (see below).

The momentum for TEEB has shifted to implementation at the country level in response to requests from governments to build national, regional, and local capacity to produce tailored economic assessments of ecosystems and biodiversity and support to mainstream this information into policy-making. The support provided includes:

- Developing guidance material on how to mainstream the value of ecosystems and biodiversity into decision-making at the country level
- Organizing workshops to build the capacity of national, regional, and local stakeholders to produce tailored economic assessments of ecosystems and biodiversity
- Providing technical expertise to five pilot countries to undertake assessments of their ecosystems and biodiversity and mainstream this information into policy

TEEB comprises a tiered approach for analyzing and structuring valuation through three core principles, namely, that:

- Recognizing value in ecosystems, landscapes, species, and other aspects of biodiversity is a feature of all human societies and communities and is sometimes sufficient to ensure conservation and sustainable use.
- Demonstrating value in economic terms is often useful for policy makers and others to reach decisions that consider the full costs and benefits of an ecosystem

rather than just those costs or values that enter the markets in the form of private goods.
- Capturing value involves the introduction of mechanisms that incorporate the values of ecosystems into decision-making through incentives and price signals.

Major Activities

The TEEB initiative has now presented an impressive number of reports. These include the following reports presented at the Convention on Biological Diversity conference of parties in Japan in October 2010, as listed below:

- TEEB Ecological and Economic Foundations (2010) – the fundamental concepts and state-of-the-art methods for economic valuation of biodiversity and ecosystem services
- TEEB in National and International Policy Making (2010) – an analysis and guidance on how to value and internalize biodiversity and ecosystem values in policy decisions
- TEEB in Local and Regional Policy (2010) – an analysis and guidance for mainstreaming biodiversity and ecosystem values at regional and local levels, with case study examples;
- TEEB in Business and Enterprise (2010) – an analysis and guidance on how business and enterprise can identify and manage their biodiversity and ecosystem risks and opportunities.
- TEEB Synthesis Report (2010) – an introduction to the approaches used and recommendations on how to mainstream the economics of nature into decision-making.

Building on the impetus generated by the initial report, a number of additional studies have been undertaken. These include the following:

- TEEB Manual for Cities (2012) – focuses on how ecosystem services and their valuation can create direct benefits for cities and provides a stepwise guidance on how to do this, with in-depth case studies.
- TEEB Climate Issues Update (2009) – shows how climate change and biodiversity are inextricably linked and how investments in the restoration and conservation of ecosystems can play a major role in combating climate change.
- TEEB Oceans discussion paper (2012) – was presented at the World Oceans Summit, February 2012, and provides guidance on improving ocean management and investing in the economic benefits of marine conservation.
- Nature and its Role in the Green Economy (2012) – a discussion paper prepared for the United Nations Conference on Sustainable Development (Rio + 20) looking at how nature and natural capital contribute to a green economy.

TEEB for Water and Wetlands

This report was initiated by the Ramsar Convention Secretariat with support from the Norwegian, Swiss, and Finnish Governments, and underlines the fundamental importance of wetlands in the water cycle, and presents insights on critical water-related ecosystem services in order to encourage additional policy momentum, business commitment, and investment in the conservation, restoration, and wise use of wetlands (Russi et al. 2013). It uses the TEEB approach to generate a better understanding of the ecosystem service values of water and wetlands and encourages improved decision-making and business commitment for their conservation, investment, and wise use. The main purpose of the report was to identify major gaps and inconsistencies in current knowledge of the economics of water and wetlands, so as to inform agenda-setting for further work on the economics of water and wetlands. The report is accessible from http://www.teebweb.org/wp-content/uploads/2013/04/TEEB_WaterWetlands_Report_2013.pdf (accessed 27 August 2016).

The key messages from the report by Russi et al. (2013) are reproduced below:

1. The "nexus" between water, food, and energy is one of the most fundamental relationships – and increasing challenges – for society.
2. Water security is a major and increasing concern in many parts of the world, including both the availability (including extreme events) and quality of water.
3. Global and local water cycles are strongly dependent on wetlands.
4. Without wetlands, the water cycle, carbon cycle and nutrient cycle would be significantly altered, mostly detrimentally. Yet policies and decisions do not sufficiently take into account these interconnections and interdependencies.
5. Wetlands are solutions to water security because they provide multiple ecosystem services supporting water security as well as offering many other benefits and values to society and the economy.
6. Values of both coastal and inland wetland ecosystem services are typically higher than for other ecosystem types.
7. Wetlands provide natural infrastructure that can help meet a range of policy objectives. Beyond water availability and quality, they are invaluable in supporting climate change mitigation and adaption and support health as well as livelihoods, local development, and poverty eradication.
8. Maintaining and restoring wetlands in many cases also lead to cost savings when compared to manmade infrastructure solutions.
9. Despite their values and despite the potential policy synergies, wetlands have been, and continue to be, lost or degraded. This leads to biodiversity loss – as wetlands are some of the most biodiverse areas in the world, providing essential habitats for many species – and a loss of ecosystem services.
10. Wetland loss can lead to significant losses of human wellbeing and have negative economic impacts on communities, countries, and business, for example, through exacerbating water security problems.

11. Wetlands and water-related ecosystem services need to become an integral part of water management in order to make the transition to a resource efficient, sustainable economy.
12. Action at all levels and by all stakeholders is needed if the opportunities and benefits of working with water and wetlands are to be fully realized and the consequences of continuing wetland loss appreciated and acted upon.

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Impact of Human Activities on the Carbon Cycle

39

Hojeong Kang and Inyoung Jang

Contents

Abstract

Human activities have accelerated decomposition in wetland ecosystems, destabilizing carbon stocks in them. In particular, global climate change, drainage and atmospheric deposition are key activities that affect wetland carbon cycle substantially. Global climate change can affect carbon decomposition in wetlands by warming effects as well as more frequent droughts. Elevated CO_2 itself can increase dissolved organic carbon leaching from wetlands through enhanced primary production. For coastal wetlands, sea level rise can also affect carbon mineralization by changes in water chemistry as well as oxygen availability. Wetlands have been subject to drainage for the development of agricultural fields and urban dwellings which can accelerate carbon decomposition by aeration.

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Finally, nitrogen and sulfur deposition can either increase or inhibit carbon decomposition depending on the nutrient status and locations. Overall, human activities can accelerate carbon decomposition in wetlands resulting in greater carbon releases into the atmosphere as CO_2 or CH_4, and into the aquatic ecosystems as DOC.

Keywords
Carbon storage · Carbon mineralization · Global warming · Greenhouse gas emission · Sea level rise · DOC

Introduction

The carbon cycle in most wetlands has reached an equilibrium or quasiequilibrium state; however, recent years have witnessed rapid changes in this balance (Yu 2012). In particular, decomposition in many wetlands has been accelerated by human activities, directly or indirectly, thereby destabilizing carbon stocks that had accumulated over geological time scales. This is of great concern as it may result in higher emissions of greenhouse gases (CO_2 or CH_4), thereby exacerbating global climate change. Additionally, the release of greater quantities of dissolved organic carbon (DOC) from wetlands may cause water quality problems or the long-term movement of carbon stocks from inland to marine ecosystems.

Many different activities can affect carbon cycles in wetlands. Three key issues are considered here, namely, global climate change, drainage, and atmospheric deposition.

Global Climate Change

Warming

In general, microbial proliferation and activity are closely and directly correlated with temperature. As such, increase in mean temperature will accelerate soil organic carbon decomposition, thereby releasing greater amounts of CO_2, CH_4, or DOC depending on the edaphic conditions. This effect may be particularly distinct in northern peatlands or wetlands in Artic regions because decomposition in these areas is often limited by low temperature, and the largest increases in temperature are expected to occur at high latitudes in the northern hemisphere. Both experimental studies and longer-term observations have shown that carbon sinks in peatlands or tundra ecosystems may become carbon sources due to global warming (Yu 2012).

Drought

Changes in precipitation due to global climate change are much more complicated than temperature increases, and thus it is difficult to predict how changes in precipitation would affect wetland carbon cycles (Yu 2012). However, more frequent and intensive summer droughts are expected in Europe and Southeastern Asia, which would lower the water levels of wetlands. Such lowering of water levels in wetlands can introduce oxygen into the wetland sediment and accelerate decomposition. Additionally, some wetlands, such as tropical peatlands in Asia, may be exposed to a higher incidence of forest fires. Overall, such changes will destroy previously stable carbon pools in these wetlands (Moore et al. 2013).

Elevated CO_2

Plants use CO_2 as a substrate for photosynthesis, and numerous studies have shown the effects of elevated CO_2 on ecosystems. In wetland ecosystems, the fertilizing effects of CO_2 often do not appear as increases in net primary production of aboveground biomass (i.e., larger plants under elevated CO_2) (Kim and Kang 2008). Instead, activated photosynthesis may result in a greater extent of DOC leaching into water or an increased biomass of plant roots in wetland sediments (Freeman et al. 2004).

Sea Level Rise

The average sea level has increased globally due both to the melting of polar ice and the thermal expansion of the oceans. As sea level rises and flooding in coastal areas becomes more frequent, wetlands located in these areas may experience substantial changes. First, the infiltration of salt water can cause the subsidence of soil layers in coastal wetlands. Second, more salts may interfere with decomposition processes in coastal wetlands. For example, methane-producing wetlands may turn into "sulfate-reducing" wetlands due to a higher supply of sulfate from sea water (Gauci et al. 2004). Further studies are needed to fully explain such issues as the multiple impacts of sea level rise may interact with each other. However, overall results indicate that global climate change would destabilize carbon stocks in wetlands, which may turn into carbon sources in the long term (Yu 2012).

Drainage of Wetlands

Globally, wetlands have historically been subjected to drainage for the development of agricultural fields and urban dwellings. Such drainage is ongoing and increasing in scale. This is particularly serious in developing countries in subtropical or tropical regions where the drainage of wetlands is often used as a means of enhancing crop or biomass production. Recent studies have shown that the construction of ditches to lower water levels has been reported to cause the release of DOC from ancient organic carbon stocks in tropical wetlands (Moore et al. 2013).

Atmospheric Deposition or Fertilization

The atmospheric deposition of sulfur or nitrogen compounds along with the inflow of water containing high concentration of nitrogen (N) and phosphorus (P) fertilizers can have a substantial effect on the carbon cycles in wetlands (Elser et al. 2007). For certain types of wetlands, such as ombrotrophic peatlands, atmospheric input is the only or main source of nutrients, and hence changes in deposition (e.g., decreases in sulfuric acid or increases in inorganic nitrogen) may accelerate the decomposition of organic carbon. This occurs as strong acids, such as sulfuric acid, impede microbial proliferation, whereas nitrogen is typically a limiting nutrient for microbial growth. Marshes located near rivers or estuaries often receive nutrients from nonpoint sources, such as leachates from agricultural fields, and may experience similar consequences. The specific addition of nutrients could increase primary production in wetlands, resulting in increased carbon accumulation in the short term (LeBauer and Treseder 2008).

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Ecosystem Services Partnership

40

C. Max Finlayson

Contents

Abstract

The Ecosystem Services Partnership was launched in 2008. It comprises both institutional and individual members. The Partnership has more than 50 organization and more than 400 individual members and comprise the main organizations involved in ecosystem services science, policy, and practice. The governance structure is designed to develop the Partnership and coordinate activities led by active members.

Keywords

Ecosystem services · Partnerships · Networks · Information exchange

History

Information on the Ecosystem Services Partnership (ES-Partnership) is available online at http://www.es-partnership.org/esp (accessed 18 September 2016). The partnership was launched by the Gund Institute for Ecological Economics at the University of Vermont, USA, in 2008, in collaboration with the Environmental

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Systems Analysis Group from Wageningen University, the Netherlands, the Institute for Ecological Economics from Portland State University, USA, the Ecosystem Management Department from the University of Kiel, Germany, the Centre of Environmental Management from the University of Nottingham, UK, and the Salento Landscape Ecology Laboratory from the University of Salento, Italy. The Partnership is now coordinated by the Environmental Systems Analysis Group from Wageningen University, the Netherlands, and supported by the Foundation for Sustainable Development in Wageningen, The Netherlands. It comprises both institutional and individual members.

The aims of the Partnership are to enhance communication, coordination, and cooperation, and to build a strong network of individuals and organizations interested in many aspects of ecosystem services. As such it intends to enhance and encourage a diversity of approaches, while reducing unnecessary duplication of effort in the conceptualization and application of ecosystem services. By raising the profile of ecosystem services and promoting better practice, the Partnership will also increase opportunities for financial support and help focus the funding of individual organizations to encourage more efficient use of available resources.

Membership

The Ecosystem Services Partnership is an institutional membership organization with the following governance structure:

- Organizations as well as individuals can become full (paying) members. Core partners are organizations that support or carry out a substantial task or service in line with the aims of the Partnership. A steering committee, elected by full members, has been formed.
- The role of the Steering Committee is to appoint and provide advice to an Executive Committee, as well as make decisions that affect the wider membership. The Steering Committee can comprise a maximum of 30 members.
- The Executive Committee is responsible for the efficient operations of the Partnership. It has 7–9 members including a Chair, Cochair, Secretary, Treasurer, and 3–5 Coordinators.
- A secretariat has been established to support the Steering Committee and Executive Committee and ensure effective communication with the Members and provide support to specifically formed working groups.
- Working Groups have been established to support the development of the science and practice on specific topics and stimulate information exchange and outreach. These are led by active members of the Partnership.

The Partnership currently has more than 50 organization and more than 400 individual members from around the world and comprise the main organizations involved in ecosystem services science, policy, and practice. The above governance structure is designed to develop the Partnership and coordinate activities led by active members.

Conferences

The Partnership encourages discussion and cooperation amongst its members through conferences and workshops. It has held an annual Ecosystem Services Partnership Conference for members to discuss the progress of the Partnership, and developments in ecosystem services science, policy, and practice. From 2016 it will switch to a bi-annual cycle for international conferences, with regional conferences in the in intervening years. It specifically encourages its working groups, national networks, and partnerships to take advantage of the conferences and workshops. It also encourages the involvement of other networks and organizations in the workshops and conferences.

The ninth annual ESP conference was held in Ansan-city, Republic of Korea from 30 May to 3 June 2016. Information on previous conferences available at http://es-partnership.org/esp-conferences/previous-conferences/ (accessed 18 September 2016). In 2016 five Regional Conferences will be organised and will provide a platform for policy makers, practitioners and scientists to connect, exchange ideas, and strengthen the relationship with other networks and partners in the regions.

ESP Working groups

The Ecosystem Services Partnership facilitates the creation of working groups to create a platform for Ecosystem Services science, policy, and practical application. A brief description of existing working groups is provided below.

Biome Expert Groups provide a platform for researchers and practitioners to exchange ideas on Ecosystem Services Assessment in specific biomes (e.g., forests, grasslands, wetlands, etc.) and make the information available to a wider community of users.

Thematic Working Groups enable the creation of a platform for researchers and practitioners to exchange information and ideas on Ecosystem Services Assessment on specific topics, such as indicators, mapping, modeling, valuation, etc.

Regional chapters and national networks focus on the creation of a network on Ecosystem Services Assessment on the national and regional level.

Journals

The journal Ecosystem Services (http://www.journals.elsevier.com/ecosystem-services/ accessed 18 September 2016), published by Elsevier, is officially associated with the ESP. It is an international and interdisciplinary journal that covers the science, policy, and practice of ecosystem services across a number of research disciplines and economic sectors. Papers address the links that occur between (i) ecosystem services and social and economic benefits and associated values; (ii) the levels of ecosystem services and economic, environmental, and land use policies and practices; and (iii) government and business strategies and the sustainability of ecosystem services. Three other journals closely linked to the Partnership include Ecological Indicators (http://www.journals.elsevier.com/ecological-indicators/ accessed 18 September 2016), Solutions (http://www.thesolutionsjournal.com/ accessed 18 September 2016), and The International Journal of Biodiversity Science, Ecosystem Services, and Management (http://www.tandfonline.com/toc/tbsm21/current accessed 18 September 2016).

Check for updates

41 Intergovernmental Panel for Biodiversity and Ecosystem Services (IPBES)

C. Max Finlayson

Contents

Abstract

The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) was established in 2012, as an independent intergovernmental body open to all member countries of the United Nations. It has been developed as the leading intergovernmental body for assessing the state of biodiversity, including its ecosystems and the ecosystem services that provide benefits to people. The structure and purpose of IPBES are described and the work program introduced, including the conceptual framework that has been developed.

Keywords

Ecosystem services · Biodiversity · Intergovernmental agreement

History

The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) was established in April 2012, as an independent intergovernmental body open to all member countries of the United Nations. It is intended to develop IPBES as the leading intergovernmental body for assessing the state of biodiversity, including its ecosystems and the ecosystem services that provide benefits to people (Diaz et al.

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2015a, b). Information on the platform and its development is available from http://www.ipbes.net/ accessed 30 August 2016.

Following the completion of the biodiversity and ecosystem services assessment, the Millennium Ecosystem Assessment (www.millenniumassessment.org), there were a number of discussions about developing an ongoing mechanism for assessing the condition of global biodiversity and ecosystem services (Carpenter et al. 2009). This developed further with a consultation process to assess the need and scope of a possible International Mechanism of Scientific Expertise on Biodiversity (IMoSEB) (Loreau et al. 2006; Hoareau and Arico 2010). Following the final meeting of the steering committee for IMoSEB in November 2007, the Executive Director of UNEP was invited, in collaboration with governments and other partners, to convene an intergovernmental and multistakeholder meeting to consider the establishment of an intergovernmental mechanism for biodiversity and ecosystem services.

This led to three intergovernmental and multistakeholders meetings (Malaysia 2008, Kenya 2009, Republic of Korea 2010) to discuss ways to strengthen the science-policy interface on biodiversity and ecosystem services. These meetings identified the gaps and needs for strengthening the science-policy interface and agreed that an IPBES should be established. The last was part of what became known as the Busan Outcome, and was welcomed by the 10th Conference of the Parties to the Convention on Biological Diversity (CBD) in October 2010 and subsequently considered at the 65th session of the United Nations General Assembly (UNGA). This UN Assembly passed a resolution requesting UNEP to convene a plenary meeting to fully operationalize IPBES. The plenary was held in two sessions, the first hosted by UNEP in Nairobi in October 2011 and the second hosted by UNEP in collaboration with UNESCO, FAO, and UNDP, in Panama City in April 2012. The final outcome was that 94 governments adopted a resolution establishing the platform as an independent intergovernmental body.

The Platform

It is intended that IPBES will provide a mechanism for conducting reviews, assessments, and critical evaluations of information and knowledge from multiple sources, including from academic, research organizations, nongovernmental organizations, or community or indigenous organizations. A key component of the processes will be recognition of the role of both the wider scientific and policy communities globally based on establishing credible and transparent processes. In particular, it will endeavor to respond to the needs of Multilateral Environmental Agreements that are related to biodiversity and ecosystem services, including those dealing with wetlands, such as the Ramsar Convention on Wetlands and the Convention on Biological Diversity.

IPBES-1, the first meeting of the Platform's Plenary, was held in Germany in January 2013. The outcomes included decisions on the following: rules of procedure for the Plenary of the Platform, next steps for the development of the initial IPBES work program, procedure for receiving and prioritizing requests put to the platform, IPBES administrative and institutional arrangements, and status of contribution and initial budget for the platform for 2013.

Arrangements were made for intersessional activities and the establishment of a Multidisciplinary Expert Panel to oversee the development of the initial work program. The Multidisciplinary Expert Panel is required to develop its own working methods with the support of the secretariat and to retain its independence and focus on scientific and technical issues. Vohland et al. (2011) emphasized the necessity for IPBES to be seen as credible and effective with challenges including setting the agenda for technical assessments, organizing the assessment process, and making the findings more policy relevant.

IPBES-2, the second meeting of the Platform's Plenary, was held in Turkey in December 2013. The outcomes included decisions on the following: the work program for 2014–2018, including the related scoping documents, institutional arrangements for implementing the work program, communication and stakeholder engagement, guidance on strategic partnerships, consideration of the budget for the implementation of the work program, and a conceptual framework. The meeting adopted a very ambitious initial work program for the platform for the next 5 years and demonstrated a strong commitment to its implementation by pledging more than half (US$ 25.4 million) of the total US$ 43.5 million required, in what will be remembered as the "Antalya consensus."

IPBES currently has 125 government members and is administered by UNEP under the auspices of UNEP, FAO, UNDP and UNESCO. The IPBES secretariat is hosted by the German government on the UN campus, in Bonn, Germany. The decision making body of IPBES is the IPBES Plenary that comprises government members and observers, with two subsidiary bodies, the Bureau which is responsible for administrative activities, and the Multidisciplinary Expert Panel that is responsible for the technical and scientific work program. The work program consists of objectives, deliverables, actions, and milestones for the four core functions of Capacity Building, Knowledge Generation Catalysis, Assessment, and policy support. Scientists from around the world contribute on a voluntary basis to the work of IPBES. Peer review is used to ensure high scientific standards are achieved in the work program and that a range of views is incorporated. IPBES has also made a concerted effort to attract social researchers, including from sociology, economics, geography, political science and others, to join the IPBES program to complement the work undertaken by experts from the natural sciences (Larigauderie et al. 2015).

Diaz et al. (2015a, b) report on the development of a conceptual framework that designed to bring the IPBES effort together to ensure integration of the knowledge and biodiversity interface for biodiversity, and to accommodate diverse views on biodiversity while stimulating new thinking and approaches. The framework built on the Millennium Ecosystem Assessment and further emphasized the role of institutions as sources of environmental problems and solutions. It also extended the valuation of the contribution of nature to the quality of human life to include monetary to spiritual values. It also explicitly incorporated multiple knowledge systems. The conceptual framework is shown in Fig. 1.

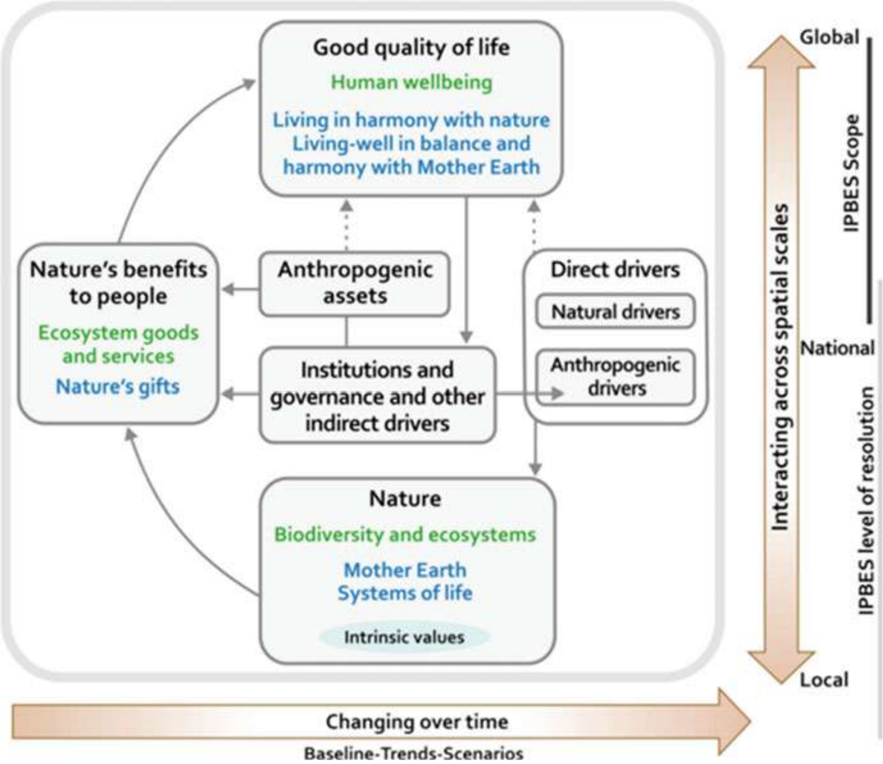

Fig. 1 IPBES conceptual framework (from IPBES http://www.ipbes.net/conceptual-framework)

The IPBES workplan was implemented from 2014 with a set of assessments and capacity building activities (Gilbert 2014). The initial assessments considered pollination and pollinators associated with food production (IPBES 2016a), and scenarios analysis and modeling of biodiversity and ecosystem services (IPBES 2016b). Others are planned.

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Millennium Ecosystem Assessment

42

C. Max Finlayson

Contents

Abstract

The Millennium Ecosystem Assessment (MA) was initiated in 2001 with the objective being to assess the consequences of ecosystem change for human wellbeing and the scientific basis for action needed to enhance the conservation and sustainable use of those systems. The outcomes were presented in five technical volumes and six synthesis reports. The combined outputs provided a state-of-the-art appraisal of the condition and trends of the world's ecosystems and the services they provide for people as well as an analysis of the options to restore, conserve, or enhance the sustainable use of ecosystems.

Keywords

Ecosystem services · Ecosystems · Assessment · Biodiversity

History

The Millennium Ecosystem Assessment (MA) was initiated in 2001 after a call from the United Nations Secretary-General Kofi Annan in 2000. The objective of the MA was to assess the consequences of ecosystem change for human wellbeing and the scientific

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basis for action needed to enhance the conservation and sustainable use of those systems. It involved more than 1,360 biophysical and social scientists worldwide with the outcomes presented in five technical volumes and six synthesis reports. The combined outputs provided a state-of-the-art appraisal of the condition and trends of the world's ecosystems and the services they provide for people as well as an analysis of the options to restore, conserve, or enhance the sustainable use of ecosystems. The information that is presented below is taken from the overview of the MA and its outcomes provided at http://www.millenniumassessment.org/en/Index-2.html (accessed 27 August 2016).

The main activities of the MA occurred between 2001 and 2005 having officially started in April 2001 and being formally launched by UN Secretary–General Kofi Annan, on World Environment Day, 5 June 2001. The first year was primarily concerned with designing the approaches that would be used for the global and subglobal assessments; this information was presented as a framework for undertaking the assessment (MA 2003). The core assessment and drafting of the technical reports was carried out in the second and third years. The draft reports went through two rounds of governmental and peer review with the findings being formally approved by the Board on 23 March 2005.

The key "users" of the MA were represented through a Board and included representatives of the Convention on Biological Diversity, Convention to Combat Desertification, the Ramsar Convention on Wetlands, and the UN Convention on Migratory Species, as well as national governments, United Nations agencies, civil society, including indigenous peoples, and the private sector. Other members were also added to ensure appropriate geographical and sectoral distribution among Board members. UNEP provided the overall coordination and employed the MA Director who was based at the WorldFish Centre in Malaysia. The overall budget was approximately US$24 million including around $7 million through in-kind contributions for subglobal assessments.

The assessment was undertaken by an international network of scientists and other experts, with a process based on that used for the Intergovernmental Panel on Climate Change. More than 1,300 authors from 95 countries were involved and organized into four working groups. Three of these groups – Condition and Trends, Scenarios, and Responses – carried out the global assessment component with the fourth undertaking a set of subglobal assessments.

Main Findings

The main findings were presented in four volumes covering current status and trends, scenarios, policy responses, and multiscale assessments. The general synthesis report presented the main findings of the MA (MA 2005a). These are presented below and accessible at http://www.millenniumassessment.org/documents/document.356.aspx.pdf (accessed 27 August 2016).

- Over the past 50 years, humans have changed ecosystems more rapidly and extensively than in any comparable period of time in human history, largely to

meet rapidly growing demands for food, fresh water, timber, fiber, and fuel. This has resulted in a substantial and largely irreversible loss in the diversity of life on Earth.
- The changes that have been made to ecosystems have contributed to substantial net gains in human wellbeing and economic development, but these gains have been achieved at growing costs in the form of the degradation of many ecosystem services, increased risks of nonlinear changes, and the exacerbation of poverty for some groups of people. These problems, unless addressed, will substantially diminish the benefits that future generations obtain from ecosystems.
- The degradation of ecosystem services could grow significantly worse during the first half of this century and is a barrier to achieving the Millennium Development Goals.
- The challenge of reversing the degradation of ecosystems while meeting increasing demands for services can be partially met under some scenarios considered by the MA, but will involve significant changes in policies, institutions, and practices that are not currently under way. Many options exist to conserve or enhance specific ecosystem services in ways that reduce negative trade-offs or that provide positive synergies with other ecosystem services.

The main conclusion from the MA findings is that "human actions are depleting the Earth's natural capital and putting such strain on the environment that the ability of the planet's ecosystems to sustain future generations can no longer be taken for granted."

Wetland Findings

The water and wetland synthesis was developed, in part, to inform the Ramsar Convention on Wetlands about the findings of the MA. This was published in 2005 (MA 2005b) and presented to the Conference of Parties of the Convention later that year.

The main findings from the wetland synthesis were summarized as a number of key points, as shown below.

- Wetland ecosystems (including lakes, rivers, marshes, and coastal regions to a depth of 6 meters at low tide) are estimated to cover more than 1,280 million hectares, an area 33% larger than the United States and 50% larger than Brazil. However, this estimate is known to underrepresent many wetland types, and further data are required for some geographic regions. More than 50% of specific types of wetlands in parts of North America, Europe, Australia, and New Zealand were destroyed during the twentieth century and many others in many parts of the world degraded.
- Wetlands deliver a wide range of ecosystem services that contribute to human wellbeing, such as fish and fiber, water supply, water purification, climate

regulation, flood regulation, coastal protection, recreational opportunities, and, increasingly, tourism.

- When both the marketed and nonmarketed economic benefits of wetlands are included, the total economic value of unconverted wetlands is often greater than that of converted wetlands.
- A priority when making decisions that directly or indirectly influence wetlands is to ensure that information about the full range of benefits and values provided by different wetland ecosystem services is considered.
- The degradation and loss of wetlands is more rapid than that of other ecosystems. Similarly, the status of both freshwater and coastal wetland species is deteriorating faster than those of other ecosystems.
- The primary indirect drivers of degradation and loss of inland and coastal wetlands have been population growth and increasing economic development. The primary direct drivers of degradation and loss include infrastructure development, land conversion, water withdrawal, eutrophication and pollution, overharvesting and overexploitation, and the introduction of invasive alien species.
- Global climate change is expected to exacerbate the loss and degradation of many wetlands and the loss or decline of their species and to increase the incidence of vector-borne and waterborne diseases in many regions. Excessive nutrient loading is expected to become a growing threat to rivers, lakes, marshes, coastal zones, and coral reefs.
- Growing pressures from multiple direct drivers increase the likelihood of potentially abrupt changes in wetland ecosystems, which can be large in magnitude and difficult, expensive, or impossible to reverse.
- The projected continued loss and degradation of wetlands will reduce the capacity of wetlands to mitigate impacts and result in further reduction in human wellbeing (including an increase in the prevalence of disease), especially for poorer people in lower-income countries, where technological solutions are not as readily available. At the same time, demand for many of these services (such as denitrification and flood and storm protection) will increase.
- Physical and economic water scarcity and limited or reduced access to water will be major challenges for society and are likely to be key factors limiting economic development in many countries. However, many water resource developments undertaken to increase access to water have not given adequate consideration to harmful trade-offs with other services provided by wetlands.
- Cross-sectoral and ecosystem-based approaches to wetland management – such as river (or lake or aquifer) basin-scale management, and integrated coastal zone management – that consider the trade-offs between different wetland ecosystem services are more likely to ensure sustainable development than many existing sectoral approaches and are critical in designing actions in support of the Millennium Development Goals.
- Many of the responses designed with a primary focus on wetlands and water resources will not be sustainable or sufficient unless other indirect and direct drivers of change are addressed. These include actions to eliminate production subsidies, sustainably intensify agriculture, slow climate change, slow nutrient loading,

correct market failures, encourage stakeholder participation, and increase transparency and accountability of government and private-sector decision-making.

- Major policy decisions in the next decades will have to address trade-offs among current uses of wetland resources and between current and future uses. Particularly important trade-offs involve those between agricultural production and water quality, land use and biodiversity, water use and aquatic biodiversity, and current water use for irrigation and future agricultural production.
- The adverse effects of climate change, such as sea level rise, coral bleaching, and changes in hydrology and in the temperature of water bodies, will lead to a reduction in the services provided by wetlands. Removing the existing pressures on wetlands and improving their resiliency is the most effective method of coping with the adverse effects of climate change. Conserving, maintaining, or rehabilitating wetland ecosystems can be a viable element to an overall climate change mitigation strategy.
- The MA conceptual framework for ecosystems and human wellbeing provides a framework that supports the promotion and delivery of the Ramsar Convention's "wise use" concept. This enables the existing guidance provided by the Convention for the wise use of all wetlands to be expressed within the context of human wellbeing and poverty alleviation.

The outcomes of the MA were welcomed by the Convention through Resolution IX.1 Annex A (http://www.ramsar.org/pdf/res/key_res_ix_01_annexa_e.pdf, accessed 16 September 2013), and the guidelines for wise use of wetlands mapped onto the MA conceptual framework to illustrate the breadth of existing guidelines and gaps (Finlayson et al. 2011).

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Section VI

Biological Adaptations

Kevin J. Stevens

Anatomy of Wetland Plants

43

James L. Seago

Contents

Abstract

Plants that inhabit wetlands range from very tiny plants to large trees and display various structural adaptations to wet areas. Most plants retain an epidermis that in roots may produce hairs and are one to several cell layers thick. Large air spaces (aerenchyma) are often found in the cortex and/or pith of stems and roots and the bark of some plants. Aerenchyma begins as small spaces between cells and then progresses through one of the three developmental paths: schizogeny, lysigeny, and expansigeny. In roots, the aerenchyma is delimited by an inner barrier (endodermis) and an outer barrier (hypodermis). Modifications to these layers may include Casparian bands, suberin lamellae, or secondary cell wall development. In roots, vascular tissues may be comprised of a single xylem element or multiple strands of alternating xylem and phloem. In stems, vascular tissues may be found in ring(s), as scattered bundles or with root-like configurations.

Keywords

Vascular tissue · Structural adaptations · Anatomy · Epidermis · Arenchyma · Lamellae

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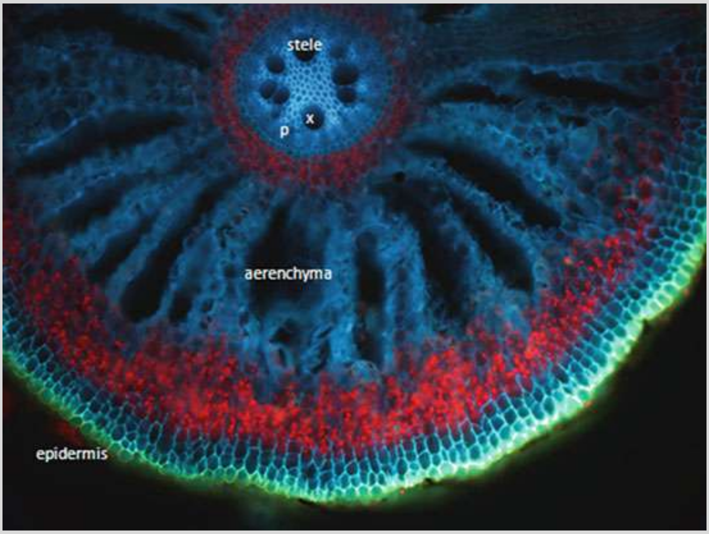
Definitions

Anatomy of a wetland plant root:

In a root cross section, three tissue systems are evident. The outermost layer of cells, the epidermis, makes up the dermal tissue system. Internal to the dermal tissue system is the cortex. Cells in the cortex comprise the ground tissue system. The ground tissue largely consists of parenchyma cells. The innermost layer of cortical cells, the endodermis, have modifications called Casparian bands, which help limit the movement of solutes and water through the roots. The outermost layer of cortical cells may also possess Casparian bands, which in this case is called exodermis. If these cortical cells lack Casparian bands, the region is called the hypodermis. While most roots and stems have gaps or intercellular spaces between cortical cells that allow gas movement, in many wetland plants these spaces or lacunae are large and parenchyma cells containing these spaces are referred to as aerenchyma. Internal to the endodermis is a central cylinder or stele where vascular tissues (xylem and phloem) are found. Xylem is involved in the transport of water, nutrients, and plant-derived compounds up the plant while the phloem is involved in the transport of plant-derived compounds down the plant. In dicotyledonous plants, the xylem radiates out from the center of the stele forming arms or poles. Phloem is found between the poles. Species with two poles have a diarch xylem arrangement. Those species with three poles have a triarch arrangement and more than three poles are called polyarch. Monocot roots are similar; however, the xylem does not extend to the center of the stele. Monocot roots have a central pith of parenchyma cells. Below is a cross section of a cat-tail (*Typha latifolia*) root, viewed using fluorescence microscopy.



Introduction

It might seem surprising, but flowering plants that inhabit wetland habitats, including aquatic habitats of ponds and streams, are extremely varied in their structural adaptations to these watery places. These species range in size from extremely tiny plants such as *Trithuria filamentosa* (Hydatellaceae), a member of a small group once classified as a grass but now included in the water lilies as a Basal Angiosperm, to Eudicot tree species of southern US swamps, e.g., *Nyssa aquatica* (Nyssaceae) and the very widespread Monocot, cat-tail, *Typha glauca* (Typhaceae).

Among the Basal Angiosperms, the water lilies (e.g., *Nymphaea*, Nymphaeaceae; *Cabomba caroliniana*, Cabombaceae) are all wetland/aquatic species. Many Monocot families of flowering plants have wetland or aquatic species – from the Acoraceae (*Acorus calamus*) through the Amaryllidaceae (*Leucojum aestivum*) to the Poaceae (e.g., *Oryza sativa*). We tend to find more Monocot species than Eudicot species characteristic of wetlands, and these are herbaceous annuals to perennials. The wetland Eudicots range from some Ranunculaceae (*Ranunculus repens*), through the Gunneraceae (*Gunnera magellanica*) of the Core Eudicots, to the Menyanthaceae (*Nymphoides cristata*) and are usually herbaceous. Anatomical features like epidermis, cortex, and stele originate from apical meristems and are characterized as follows.

Epidermis

Most wetland species retain their epidermis throughout their lives (see Fig. 3, courtesy of Aleš Soukup). The roots of aquatic species often have particular adapations for the aquatic environment such as hairs. However, some species do not have hairs, and these have a thick hypodermis. These species lose their epidermis, and the exodermis, the specialized layer(s) of the hypodermis, becomes the outermost zone of the roots; thus, there are no root hairs. The occurrence of cutins in the epidermis of the stems and roots of aquatic species has been recorded, but in recent years diffuse suberin has been found in the root epidermis, thus making it even more of a barrier layer, especially to fungal infection (Meyer and Peterson 2013).

Cortex: Aerenchyma

While are no typical wetland plant anatomies, some anatomical adaptations are common to many wetland plants. One of the most common adaptations of many plants which inhabit wetland/aquatic environments is the occurrence of sizeable air spaces which are known as aerenchyma and almost always occur in the cortex or ground tissue of organs. In some species, expanded spaces or cavities can be found in the pith of roots or pith and ground tissue of stems.

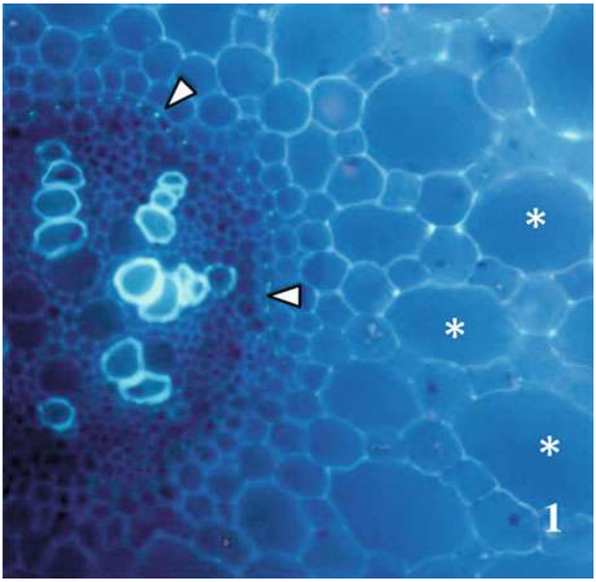


Fig. 1 Cross section of a *Victoria amazonica* root. Expansigenous aerenchyma (*). Endodermis with Casparian bands (*arrowheads*)

The term, aerenchyma (Schenck 1890), is used to represent the organized air space tissue of roots, stems, and leaves. All types of aerenchyma arise from very small intercellular spaces which themselves develop by schizogeny or separation of cell walls at the intersections of cells, usually where the cells arise from meristems. Further expansion of these tiny spaces into lacunae (the enlarged air spaces of aerenchyma) occurs in three main ways, developmental paths which have only recently been well elucidated by Seago et al. (2005), who applied the terminology to roots, and confirmed by Jung et al. (2008), who expanded the concepts to include stems. The ways in which the living cells of aerenchyma can be packaged have been documented for many species by Justin and Armstrong (1987).

Expansigeny is the development of aerenchyma from these tiny spaces by means of cell divisions of the surrounding cells and by concomitant cell enlargement or elongation of the cell walls along the developing space in such a way that neither cell separations nor cell deaths contribute to the enlarged air spaces. Examples are especially common in the water lilies (Fig. 1, e.g., *Victoria amazonica* in the Nymphaeales).

Schizogenous aerenchyma develops after the initial intercellular space formation and only by the separation (or schizogeny) of adjoining cell walls, and the resulting schizogenous space forms a lacuna (Fig. 2, *Justicia americana*, Acanthaceae). Examples can be found in many Monocots and Eudicots.

In lysigeny, aerenchyma arises primarily by the death of specific cells to create an enlarged space; these cell deaths are usually accompanied by some schizogenous separations early in their formation. The developmental patterns which produce this lysigeny are clearly constitutive in many wetland plant species (Jung et al. 2008).

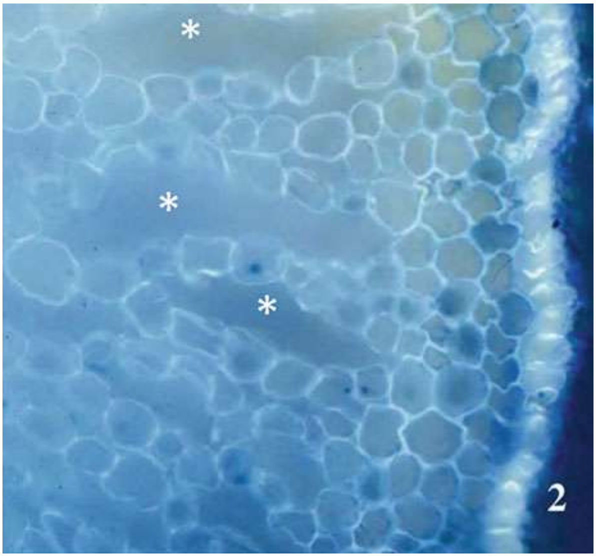


Fig. 2 Cross section of a *Justicia americana* root. Schizogenous aerenchyma (*)

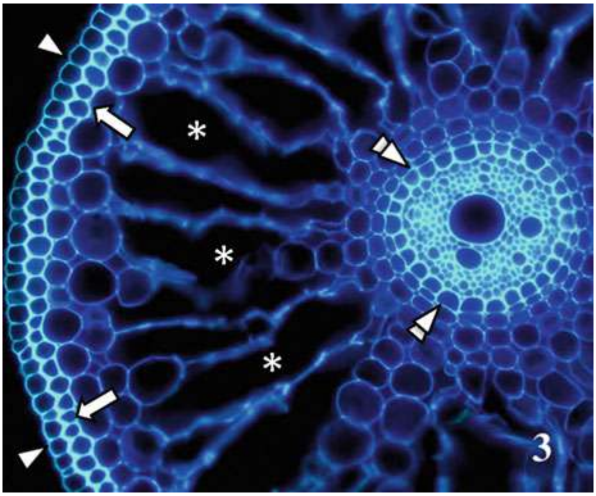


Fig. 3 Cross section of a *Glyceria maxima* root. Lysigenous aerenchyma (*). Epidermis (*arrowhead)*, exodermis (*arrow*), endodermis (*double arrowhead*). Note the central polyarch stele

Oryza sativa is one of the best known examples of lysigeny (Fig. 3). If some schizogeny also occurs in lacunar formation, we term the aerenchyma schizo-lysigenous (Fig. 4, *Phragmites australis*, Poaceae).

In stems and petioles of many species, e.g., *Cabomba caroliniana*, a Basal Angiosperm in the Nymphaeales, aerenchyma organization is often much like that in roots, but enlarged cavities, even in the pith, are also common, especially in the grasses.

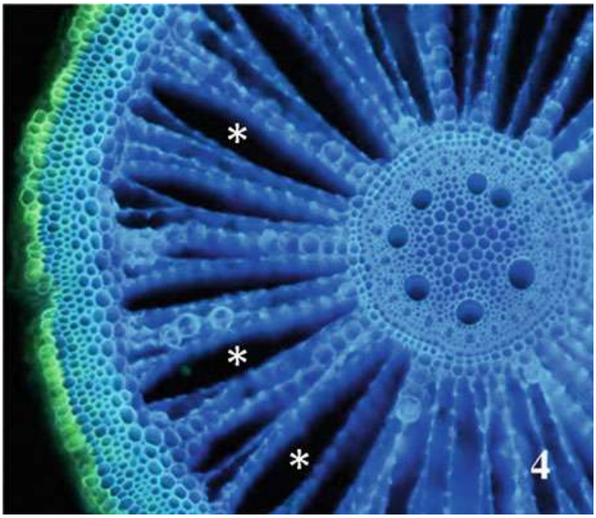


Fig. 4 Cross section of a *Phragmites australis* root. Schizo-lysigenous aerenchyma (*). Note the central polyarch stele

Cortex: Barrier Layers

In roots, aerenchyma is delimited by two barrier layers (Enstone et al. 2003). The endodermis with Casparian bands represents the innermost barrier layer (Fig. 1), while the hypodermis forms the outermost barrier layer. The hypodermis is a distinct, concentric layer or layers of cells interior to the epidermis that is derived from the meristem and that is often modified into an exodermis, a layer like the endodermis with Casparian bands, plus suberin lamellae. An endodermis is a characteristic of the innermost layer of the root cortex in all flowering plants (Fig. 5, *Nymphoides cristata*), but it can also be found in stems of some wetland plants – from emergents like *Typha* to floating plants like *Najas* (Hydrocharitaceae; Fig. 6).

In the wetland Basal Angiosperms, there is always an endodermis with Casparian bands, but a few species also have suberin lamellae. Most species, especially Eudicots, have a simple endodermis with Casparian bands only (Fig. 5, *Nymphoides cristata*); however, many species, especially Monocots, have suberin lamellae and secondary lignified walls added to Casparian bands (Fig. 7, *Iris pseudacorus*, Iridaceae). An exodermis, the hypodermis with Casparian bands and suberin lamellae in the cell walls, can be found in many wetland species, especially Monocots, all wetland Basal Angiosperms, e.g., *Nymphaea odorata*, and a few Eudicots, even including stems, e.g., *Nymphoides peltata* (Fig. 8).

An exodermis can be found in rhizomatous plants such as cattails and can be several cell layers thick (Fig. 9, *Typha glauca*), each cell having Casparian bands, suberin lamellae, and secondarily lignified cell walls. The nature of these cell wall modifications in the barrier layers and epidermis has been summarized by Enstone et al. (2003) and Meyer and Peterson (2013).

Fig. 5 Cross section of a *Nymphoides cristata* root. Endodermis with Casparian bands (*arrowhead*). Note the central pentarch stele (5 poles of xylem and phloem)

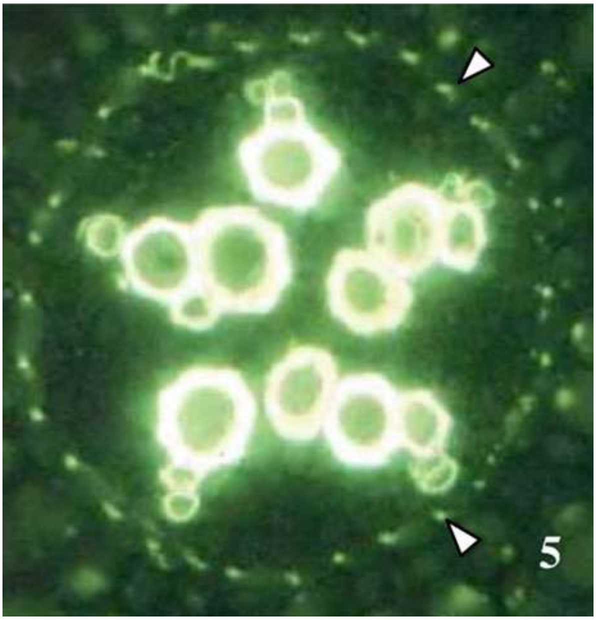
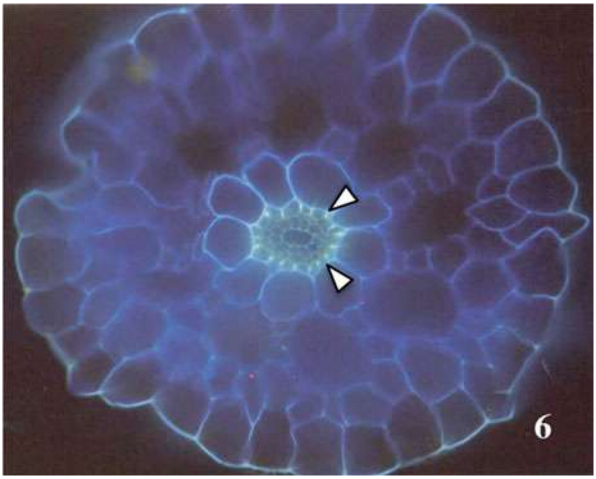


Fig. 6 Cross section of a *Najas minor* stem with endodermis (*arrowhead*) and a simple stele



Vascular Tissues

The primary vascular tissue, or stele, in roots is a product of the root apical meristem. Basal Angiosperms have a very wide variety of arrangements of their root primary xylem and phloem, varying from one xylem element to a polyarch condition,

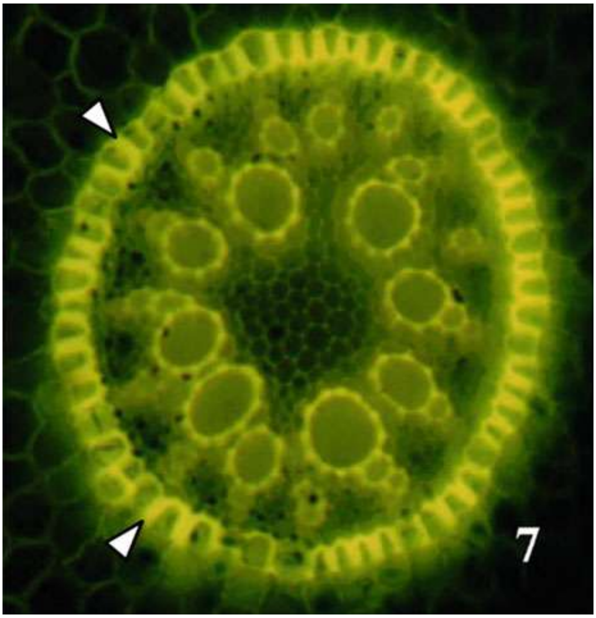


Fig. 7 Cross section of an *Iris pseudacorus* root. In addition to Casparian bands the endodermis has been further modified by the deposition of suberin lamellae, and lignified secondary walls (*arrowhead*). Note the central polyarch stele

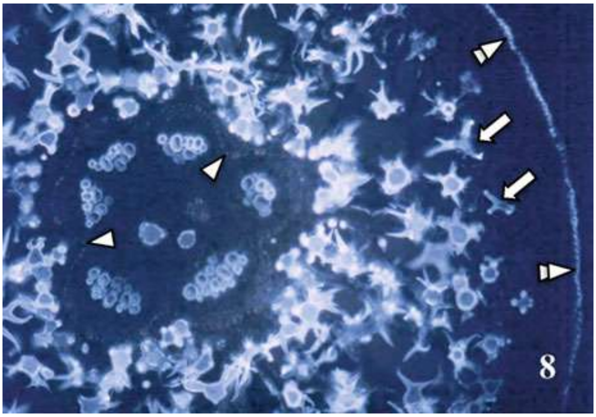


Fig. 8 Cross section of a *Nymphoides peltata* stem with endodermis (*arrowheads*) around the stele and an exodermis with suberin lamellae (*double arrowheads*). Note the presence of astrosclereids (*arrows*) throughout the cortex

whereas Monocots and Eudicots generally differ from each other in conspicuous ways (see Seago and Fernando 2013). Eudicots are usually depicted as having 2 –6 poles or strands of alternating xylem and phloem (termed diarch to hexarch, respectively; e.g., Fig. 5, pentarch in *Nymphoides*), while Monocots typically have 6 or more such poles (hexarch to polyarch, often 10–20 poles; see Figs. 3, 4, 7, 10 – *Typha glauca*). However, there are some notable exceptions, even within a genus;

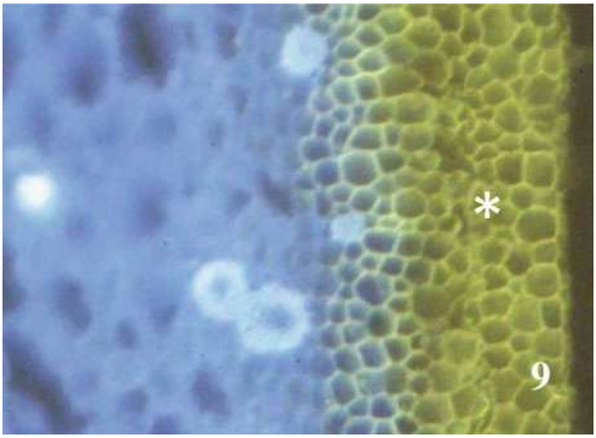


Fig. 9 Cross section of a *Typha glauca* rhizome showing a multiseriate exodermis (*)

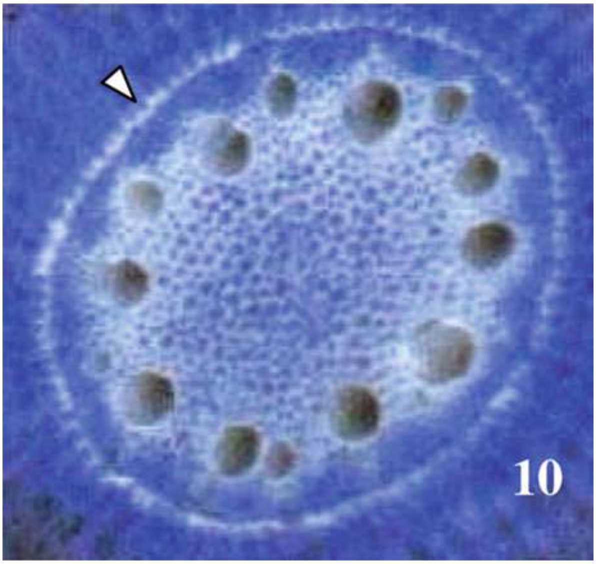


Fig. 10 Cross section of a *Typha glauca* root. Note the endodermis (*arrowhead*), and polyarch stele

some species of the basal core Eudicot, *Gunnera*, have polyarch arrangements of primary xylem and phloem without secondary growth (Fig. 11), but other species are tetrarch with secondary growth. Species with more than 6 strands of xylem and phloem usually do not undergo secondary growth in their roots.

In stems, Eudicots generally have one ring of vascular bundles (xylem and phloem, an arrangement which can allow for secondary growth), and Monocots have two, more or less, concentric rings of vascular bundles or vascular bundles

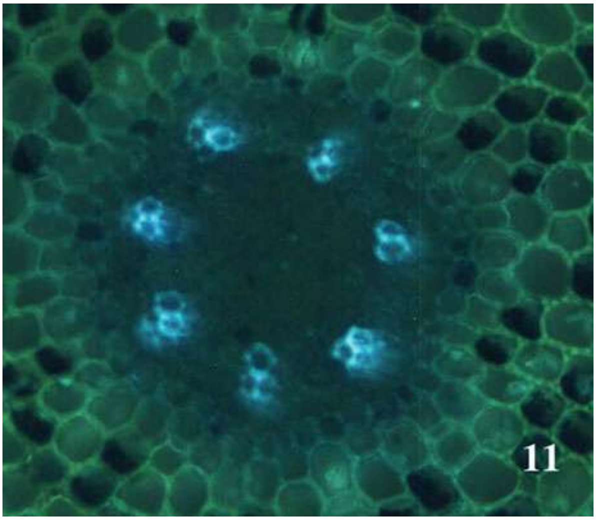


Fig. 11 Cross section of *Gunnera magellanica* root. *G. magellanica* is a eudicot with polyarch stele (7 xylem and phloem poles)

scattered throughout the ground tissue of the stem. However, some species with very small stems have structural configurations and cell types resembling those of roots. For example, the water lilies (Nymphaeaceae, Cabombaceae, Hydatellaceae) have stems with a stele surrounded by an endodermis. Some species of Monocots, like the river-dwelling *Najas*, have very small steles with little vascular tissue, plus an endodermis with Casparian bands and aerenchyma (Fig. 6). Far larger emergent plants, like *Typha*, have large rhizomes with an endodermis separating a central cylinder of scattered vascular bundles and a cortex with vascular bundles. Among Eudicots, some of the wetland plants, e.g., *Gunnera magellanica*, *Trapa natans* – Lythraceae, and *Nymphoides peltata*, have stems with an endodermis surrounding the entire stele (Fig. 8) or all vascular bundles (e.g., *Justicia americana*, not shown).

Petioles of wetland plants ordinarily have some aerenchyma, but they do not usually have an endodermis delimiting the ground tissue from the vascular bundles. One exception to this is the genus *Gunnera* (Fig. 12) in which most species have a constitutive endodermis with Casparian bands in lower portions of petioles, even when they are above water levels.

Secondary Tissues: Aerenchymatous Phellem

Arenchymatous tissue is not found in secondary xylem or phloem but can be found in periderm (outer bark) in which the phellem, or outer bark, is modified into aerenchyma. This has been well documented in North America in the well-known invasive species, *Lythrum salicaria* (Lythraceae; Stevens, et al. 2002). Here, cell division and cell expansion lead to the enormous lacunae of the phellem.

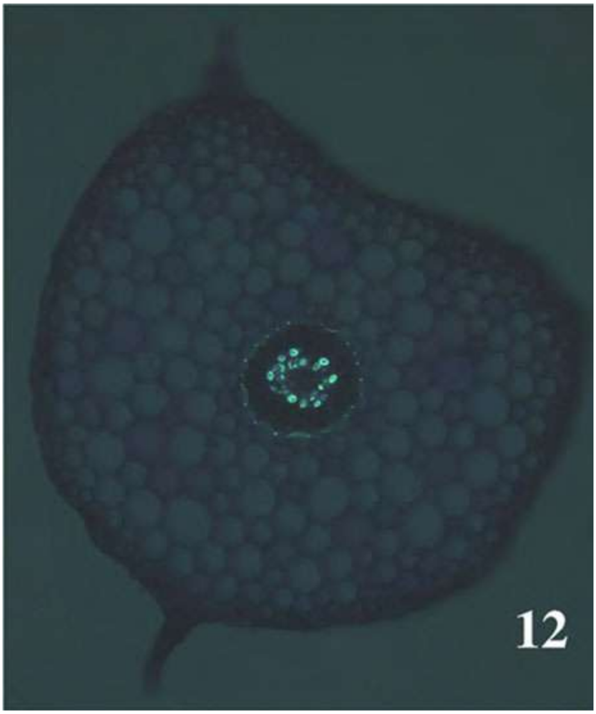


Fig. 12 Cross section obtained from midway along a *Gunnera flavida* petiole. Note the endodermis with Casparian bands around the vascular bundle

Future Challenges

Relatively few plant species that inhabit wetland or aquatic environments have actually been examined well enough for scientists to know their structural features; thus, many more species, especially from bog habitats which are woefully underexamined, should be investigated. Much progress has been made on the ways in which the plant parts under the surface of the water get aerated, but the structural connections among the different plant organs to facilitate this gaseous exchange are not well known.

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Wetland Plant Morphology

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Gary P. Shaffer and Demetra Kandalepas

Contents

Abstract

Anaerobic conditions cause death in plants grown in saturated soils. Where flooding is predictable, species may time germination to coincide with low water levels. In other areas, specialized adaptations, such as rapid shoot elongation, adventitious root production, or shoot buttressing and fluting, are strategies to counteract oxygen depletion. Aerenchyma in shoots and roots allow oxygen to move from emergent to submerged organs and may diffuse out of roots oxidizing

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surrounding soils. Temperature gradients drive pressurized air movement within some plants. Shallow, extensive, intertwining roots can improve oxygen penetration and provide a belowground network, limiting hurricane damage. Prop roots and pneumatophores enhance oxygen movement in some mangrove plants. Floating mats are produced in areas experiencing subsidence. Solute imbalances can create conditions where water leaves the cytoplasm. Water loss can be prevented by production of salt barriers and nontoxic solutes in saline conditions, but also by confining CO_2 production to night time.

Keywords
Anaerobic avoidance strategies · Aerenchyma · Root adaptations · Adaptation to salt · Photosynthesis

Definitions

Roots out of soil in wetland plants:

While roots are generally thought of as growing down, underneath the shoot, this is not always the case. Many flooded plants will produce roots along submerged parts of the stem. These roots are referred to as adventitious roots (a). Adventitious root development can occur rapidly following submergence. Adventitious roots may assume the role of the root system if the original root system perishes and their location in the water column allows them access to more oxygen rich waters. The capacity to develop roots on stems permits the ready propagation of wetland plants in commercial nurseries. Some plants produce upward growing roots called pneumatophores (b). The roots are believed to function like snorkels providing a means of transferring oxygen to submerged plant parts. Several types of pneumatophores, differing in their developmental patterns, are recognized. In addition to pneumatophores, some mangrove plants produce stilt roots. These are downward growing roots forming on the trunk or lower branches that anchor in the substrate and provide additional support to emergent shoots (Photos by Kevin Stevens).

(*continued*)


(a)
69
2
(b)

Introduction

The vast majority of plant species on Earth cannot survive long in water-saturated soils, not because of the water itself but because the soils are anaerobic. Oxygen diffuses into water 10,000 times slower than air. Therefore, saturated, or *hydric*, soils become anaerobic quickly, and without oxygen as the terminal electron acceptor in the electron transport chain, aerobic metabolism ceases in plant roots. Under anoxic conditions, terrestrial plants, and some wetland plants, will begin metabolizing through fermentation, whose end product (ethanol) is toxic and generally cause death within 24 h. Many wetland plants actually accelerate rates of fermentation to maintain adequate levels of ATP but release the ethanol to the environment and thus do not accumulate this toxic byproduct. An additional stress triggered by anaerobic hydric soils involves production of other toxic byproducts, as specialized microbes are able to substitute other electron acceptors in the electron transport chain, which become toxic when reduced, such as sulfate reduction to lethal hydrogen sulfide. Many species of wetland plants have evolved shoot and root adaptations that enable delivery of oxygen to the roots, avoiding the problems caused by anaerobic soils. In areas where the timing of flood events is predictable, many species have evolved timing mechanisms enabling seeds to germinate during draw down conditions.

Coastal wetlands experience the additional stress of salt water and have evolved adaptations to either exclude salt from entering the roots, sequester, or secrete it. These mechanisms involve preventing entry of salt at the root soil interface, producing nontoxic organic solutes to compensate for osmotic imbalance, packing the salt into vacuoles, xylem tissue, or old leaf tissue, or some species may excrete salts via salt glands.

Anaerobic Avoidance Strategies

General strategies used to avoid anaerobic metabolism include (1) shoot elongation to render a portion of the plant out of the water, (2) production of adventitious roots along the aerobic portions of the stem, (3) slowing or shutting down metabolism until floodwaters retreat, (4) buttressing or fluting of the basal shoot to increase surface area exchange, and (5) specialized root adaptations (see below).

Aerenchyma

The most ubiquitous adaptation enabling shoots to deliver oxygen to the roots is the creation of air spaces called *aerenchyma*. When the surrounding environment becomes hypoxic, the hormone ethylene is produced, which triggers production of the enzyme cellulase. This enzyme breaks down cellulose within both shoots and roots, creating honeycomb or snorkel-like air spaces. These airspaces allow oxygen to diffuse into the roots and from the roots to the pore space lining of surrounding soils to create an *oxidized rhizosphere*. In the oxidized rhizosphere, reduced toxic substances, such as ammonia and hydrogen sulfide, become oxidized into nontoxic nitrate and sulfate, respectively. Aerenchymous tissue may occupy up to 60% of a wetland plant's root tissue volume (Mitsch and Gosselink 2007).

Root Adaptations

Root adaptations that enable aerobic metabolism include shallow but extensive rooting, *prop root* formation, and *pneumatophore* or air root production. In general, forested wetland tree species kill off their taproots when young and send shallow lateral roots far beyond their drip line. In addition to enhancing oxygen penetration, these extensive lateral roots form a tapestry with one another, rendering such species as baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*) highly resistant to hurricane damage; when basal areas reach greater than 30 m^2 ha^{-1}, these two canopy species also keep readily wind-thrown midstory and herbaceous vegetation intact (Shaffer et al. 2009).

To enhance oxygen penetration to the roots, red mangroves (*Rhizophora* spp.) develop aboveground roots called prop roots. These roots are densely topped with spongy air-filled lenticels that allow oxygen diffusion into the roots.

Black mangrove (*Avicennia* spp.) similarly contain thousands of pneumatophores, which are spongy extensions of the main roots (about 30 cm long), which climb vertically through the mud and are exposed during low tides. Both prop roots and pneumatophores greatly enhance root oxygenation, as demonstrated by experiments that blocked airflow using substances such as Vaseline®. This has not been demonstrated in the vertical root extensions, or *knees,* of baldcypress; it does appear, however, that the knees offer structural support during tropical storm events (Conner et al. 2012).

Perhaps the most intriguing root adaptation to chronic flooding is the development of floating marsh or *floatant*. The world's large delta complexes undergo cycles of progradation and degradation as their rivers periodically switch paths. As a newly forming delta progrades, the abandoned delta subsides as its sediments compact and dewater. This subsidence leads to increased flood durations and at some point the vegetative community must "sink or swim." Vegetative species that form floatant create gaseous roots and rhizomes such that the entire root mat detaches from the subsoil and essentially becomes a hydroponic marsh (Sasser et al. 1991). While these floating marshes can be very healthy ecosystems, they do not stand up to the strong winds of tropical storms. In high winds and strong storm surge, the floatant marshes roll up in carpet-like fashion and get thrown landward where they strand and die.

Thermo-Pressurized Gas Flow

Thermo-pressurized gas flow is a mechanism used by plants to actively move oxygen from shoots to roots. To achieve this, plants use the temperature and water pressure differentials between the exterior ambient air and the internal cortical tissue to create pressure gradients, which push oxygen through aerenchymous shoots into aerenchymous roots. Pressures generated during daytime peak temperatures are up to 30 times higher than those generated at night (Brix et al. 1992). Originally, humidity was thought to be the driving force in pressurized gas flow (Brix et al. 1992). However, this process can occur in trees even when the plant is dormant, so humidity is not likely to be the driving force. First discovered by Dacey (1980, 1981) in water lilies (*Nuphar luteum*), thermo-pressurized flow was once thought to be unique; however, it has since been demonstrated in several other hydrophyte (Brix et al. 1992) and wetland tree species (Grosse and Schroder 1984; Grosse et al. 1992). The route for gaseous transport may differ among species. *Alnus glutinosa*, for example, uses meristematic tissue in lenticels for pressurized gas flow, rather than leaf tissue. This pathway, however, is 8× less efficient than gas flow through leaf aerenchyma. Nevertheless, pressurized gas flow in mature trees occurs only when the tree is dormant, but in seedlings it occurs in leaves, before the seedling establishes aerenchyma (Mitsch and Gosselink 2007).

Adaptations to Salt

In general, wetland vascular plants are faced with the same challenges as single celled organisms when the outside aqueous environment has more solutes than inside the cell. The water potential of the outside solution is lower and water escapes the cytoplasm. A suite of wetland species have solved the problem by (1) eliminating salt (NaCl) at the root surface to maintain nearly fresh internal solutions, (2) increasing nontoxic organic compounds inside the cytoplasm to reverse osmotic imbalance, (3) isolating or eliminating salt that enters through the roots, and (4) using a photosynthetic pathway that minimizes fresh water requirements.

Salt Barriers

Mangroves of the genera *Sonneratia*, *Rhizophora*, and *Laguncularia* have tight membranes on the root surface that reduce the salinity of internal water to 1–1.5 ppt compared to 35 ppt of the outside sea water (Mitsch and Gosselink 2007). The mangrove *Avicennia* spp. is also able to greatly reduce the salinity internally but only to about 10% of sea water.

Organic Solutes

To prevent salt from becoming toxic internally, some wetland plant species produce nontoxic organic compounds to more than match the osmotic potential of the outside aqueous solution. These compounds are called *osmotica* and include prolene and beta glycine.

Salt Elimination

At the cellular level, vacuoles can serve as trash cans for sodium chloride. Salt also can be packed into old leaf tissue and the leaves excised. More elaborate mechanisms include the salt glands on the leaves of *Spartina* spp., which actively secrete sodium ions relative to potassium. Remarkably, baldcypress packs salt into last year's dead xylem tissue. Baldcypress, which live over 1,000 years, can therefore be used to reconstruct historic tropical storm events (Hupp and Morris 1990).

C_4 Photosynthetic Pathway

Wetland plants inhabiting saline environs face much the same "physiological drought" (Schimper 1903) problems as plants in arid environments; in both cases plants are rooted in a substrate with very low water potential, because of either dryness or high salt content. The dilemna from a water loss perspective is that vascular plants lose water by opening their stomata to capture carbon dioxide for carbon fixation in the photosynthetic process. Water is a much smaller molecule than carbon dioxide, so it readily diffuses out of open stomata. Water loss through evapotranspiration must be compensated by water uptake through the roots. An additional problem for species that rely exclusively on the C_3 pathway involves the wasteful fixing of oxygen rather than carbon dioxide, a process known as *photorespiration*. C_3 plants obtain the CO_2 needed for the initial steps in photosynthesis during daylight hours. Photorespiration occurs because the initial enzyme in the C_3 pathway, known as Rubisco (ribulose biphosphate carboxylase), has a relatively low affinity for carbon dioxide and also fixes oxygen to the 3-carbon sugar glycophosphate. Vascular plants with the additional C_4 pathway solve the problem by initially fixing carbon dioxide at night. Fixation occurs via the enzyme PEP (phosphoenol pyruvate) carboxylase, which has a very high affinity for CO_2. Wetland plant species that use the C_4 pathway are far more efficient with water use per amount of carbon fixed.

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Physiological Adaptations to Wetland Habitats

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William Armstrong and Timothy D. Colmer

Contents

Abstract

Low gas-diffusivity and oxygen-solubility in water are primary constraints in wetland and aquatic environments. Consequentially, microbes may remove oxygen from all but surface layers of waterlogged soils and generate phytotoxins, while shoot-submergence may substantially reduce CO_2-availability for photosynthesis, compromising carbohydrate production. Oxygen limitations may cause energy crises within the plant and self-generation of damaging reactive oxygen species; enhanced generation and entrapment of the gaseous hormone ethylene also accompany waterlogging and submergence. Globally, wetlands differ in altitude, timing, duration and depth of flooding, light, temperature, and biogeochemistry. Consequently, suites of adaptive traits have evolved in plants to accommodate these varied conditions. Adaptations include anoxia avoidance by facilitating gas

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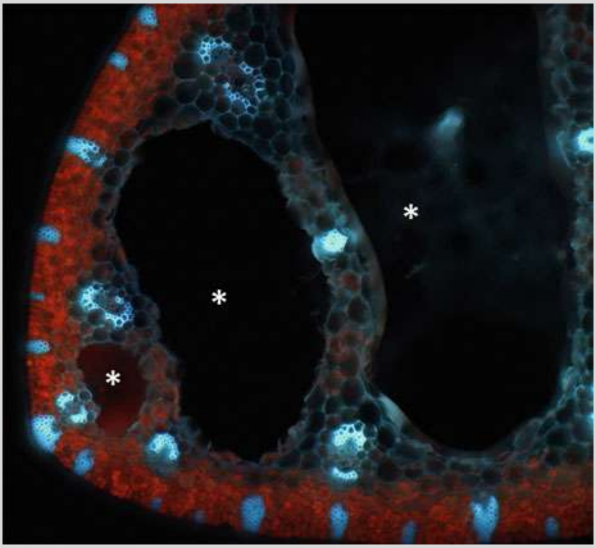
exchange with the atmosphere to support aerobic metabolism, an emergence escape strategy or a medium-term submergence tolerance strategy, or production of leaves capable of photosynthesis when submerged. Other adaptations include short- or long-term anoxia tolerance or protection from reactive oxygen species and phytotoxins.

Keywords
Anoxia avoidance · Reactive oxygen species · Tolerance of anoxia · Post-anoxic injury · Phytotoxins

Definitions

Breathing underwater:

Plants, like animals, require oxygen for respiration. Usually, this is obtained from the air. Plant organs below the ground, roots and sometimes stems, obtain oxygen from air in spaces between soil particles. As water levels rise, the air spaces become filled with water and oxygen becomes limited. Many plants, however, possess an internal pathway for the exchange of gases between submerged and nonsubmerged tissues. This pathway consists of spaces that are interconnected forming a continuous pathway for the flow of gases between leaves, stems, and roots. The most pronounced spaces, clearly evident in cross sections, are referred to as aerenchyma. Oxygen moves down this pathway to submerged tissues, while carbon dioxide and other gases move from the submerged tissues out through the stems and leaves. The following image shows aerenchyma in the leaf of cat-tail (*Typha latifolia*). A cross section of a leaf was obtained by hand and viewed using a fluorescence microscope (Photomicrograph by Kevin Stevens).



Introduction

Impeded gas exchange is a primary constraint for plants in wetland and aquatic environments. Excess water can interrupt O_2 supply for respiration causing "energy crises" and when shoots are submerged restrict CO_2 supply for photosynthesis with adverse effects on the carbohydrate economy. Other major constraints include soil-generated phytotoxins and self-generated reactive oxygen species (ROS).

Waterlogging robs the soil of much of its gas-filled space. Together with low O_2 solubility in water, very low liquid-phase diffusivities, and microbial activities, this causes *flooded soils* to become *very O_2 deficient, high in CO_2, and reservoirs of potential phytotoxins* (see also Vol. 1, ▶ Chap. 31, "Microbially Mediated Chemical Transformations in Wetlands"); free O_2 is limited to surface layers (Armstrong 1979). Unless roots and other buried organs can exploit these layers, obtain O_2 from/via an emergent shoot/root (the snorkel effect) (Armstrong 1979; Vartapetian et al. 2008; Colmer and Voesenek 2009), or become quiescent and/or utilize anaerobic metabolism until the soil drains, death is inevitable (Vartapetian et al. 2008; Bailey-Serres and Voesenek 2008; Colmer and Voesenek 2009). Even re-exposure to O_2 may be damaging unless cells are able to remove ROS that are often generated (Blokhina et al. 2003; Vartapetian et al. 2008).

Whole or partial submergence of shoots further impedes gas-exchange; although CO_2 is highly soluble in water, its availability for underwater photosynthesis is limited by an even lower diffusivity than that of O_2 (Armstrong 1979) and (for most plants which can only utilize CO_2) by its progressive transformation to bicarbonate and carbonate with increasing pH (Maberly and Madsen 2002). Submergence also hinders photosynthetic O_2 escape, raising internal $[O_2]$:$[CO_2]$ ratios, which can enhance photorespiration and reduce any net photosynthesis (Colmer et al. 2011). However at night, CO_2 can accumulate, particularly in roots, to levels that might impede metabolism (Greenway et al. 2006). With submergence, production of the gaseous phytohormone ethylene (C_2H_4) may be stimulated by lower internal O_2 concentrations and its escape hindered by its very low solubility in water (Colmer and Voesenek 2009). Depending upon the species, C_2H_4 accumulation can have a variety of physiological consequences, some favorable, e.g., induction of lysigenous aerenchyma and new adventitious roots (Colmer and Voesenek 2009), and some unfavorable, e.g., premature senescence in submerged shoots.

Since wetlands occur worldwide and at most altitudes, the types of flooding regime vary enormously in seasonality, duration, water depth, light, temperature, and biogeochemistry. The selection pressures imposed by such variety of regime have led to the evolution of suites of adaptive traits in plants that fit various flooding scenarios. Adaptations include: (i) those directed to minimizing development of anoxia in tissues viz. "anoxia avoidance" by (a) facilitating gas-exchange with the atmosphere to support aerobic metabolism (Armstrong 1979; Colmer and Voesenek 2009), (b) a total submergence escape strategy (Bailey-Serres and Voesenek 2008; Colmer and Voesenek 2009), (c) a medium term submergence tolerance strategy involving a nonelongation response (termed "quiescence" (Bailey-Serres and

Voesenek 2008; Colmer and Voesenek 2009) to reflect restricted elongation and reduced carbohydrate consumption, although the tissues do remain metabolically active), and (d) leaves capable of photosynthesis when submerged (Colmer et al. 2011; Maberly and Madsen 2002); (ii) anoxia tolerance: short or long term (Gibbs and Greenway 2003; Greenway and Gibbs 2003); and (iii) protection from ROS (Blokhina et al. 2003; Vartapetian et al. 2008) and phytotoxins (Armstrong 1979; Colmer and Voesenek 2009; Watanabe et al. 2013). Each broad category of adaptation involves several contributing traits. Key traits relating to the principal flooding regimes are set out in Table 1 and commented upon below.

Anoxia Avoidance

Soil flooding only. Here, the major source of O_2 for most rhizomes and roots is internal gas-phase diffusion from/via the shoot, and despite daytime photosynthesis, the source concentration usually differs little from day to night at *circa* 21% (Armstrong 1979). Root tips are particularly sensitive to anoxia, but since their growth is O_2 dependent and in waterlogged soils supported by internal transport from shoots, roots of wetland species may rarely experience total tissue anoxia. The length of root (and rhizome) growth in anoxic soils depends upon: (i) their internal (chiefly cortical) gas-space provision and (ii) O_2 "consumption" along the root by respiration in the cells and radial O_2 loss (ROL) from the root – ROL is influenced by microbial and chemical O_2 demands in the soil and root anatomy and morphology (Armstrong 1979).

Root aerenchyma (see also Vol. 1, ► Chap. 43, "Anatomy of Wetland Plants") is pivotal to the success of most wetland species; it lowers respiratory demand per unit volume and, even more importantly, reduces substantially the resistance to long distance diffusion (Armstrong 1979). Where aerenchyma formation involves lysis, the cell death is thought to be caused by reactive oxygen species (ROS), with production triggered by endogenously produced ethylene. Cells that remain alive may owe their survival to ROS scavengers such as metallothionins and/or phytoglobins (Yamauchi et al. 2011; Mira et al. 2016). Roots of many wetland species also have a "tight" suberized sub-apical O_2-conserving exodermal barrier to ROL (Armstrong 1979; Armstrong et al. 2000; Colmer and Voesenek 2009; Watanabe et al. 2013) that also impedes phytotoxin entry, although some nutrient and water uptake can still proceed. Most nonwetland species form relatively small amounts of cortical aerenchyma, and some cannot form it (Justin and Armstrong 1987). In species without aerenchyma, the fractional porosity (FP; gas volume/organ volume) remains low ($\leq$5%) and respiratory demand relatively high; the interaction of this porosity and respiratory demand is strongly synergistic and limiting to adequate root aeration. Figure 1 demonstrates the benefits of aerenchyma formation, which conveys a high FP (15–60%), lower diffusive resistance, and lower O_2 demand to the plant. The superior aeration of wetland root types is dependent upon, in order of importance, reduced diffusive resistance > reduced respiratory

Table 1 Overview of the hypothesized importance of various traits associated with plant tolerance of soil waterlogging and/or submergence, for five contrasting types of wet environments inhabited by some terrestrial plant species (Reproduced, with a few modifications, from Colmer and Voesenek (2009), Functional Plant Biology, with permission of CSIRO Publishing. After Colmer and Voesenek (2009))

	Environments				
Traits	Waterlogged, short duration	Waterlogged, long duration	Submerged, short duration	Submerged, shallow prolonged (*emergence possible)*	Submerged, deep prolonged (*emergence not possible*)
Adventitious roots (sediment)	*	***	*	***	*
Adventitious roots (water)	NA	NA	NA	**	***
Aerenchyma	**	***	**	***	***
ROL and phytotoxin barrier	*	***	*	***	***/*[(a)]
Anaerobic energy production	***	*	**	*	***
Energy conservation	**	*	***	*	***
Prevention of ROS formation/ ROS defense system	***	***	***	***	**
Tolerance of toxic soil constituents	NA	***	*	***	***
Nastic movements	*	**	*	***	**
Shoot elongation	NA	NA	−ve*	***	*
Aquatic leaf traits (biophysical)	NA	NA	NA	***	***
Leaf gas films	NA	NA	***	***	**
Convective gas movement	*	***	NA	***	NA

Short duration = < 1 week. Prolonged duration = > 1 week, typically > 4 weeks
Shallow = < 1 m (i.e., water levels that plants are capable of "outgrowing")
Deep = > 1 m, typically a few meters or more
* = of little importance; ** = of moderate importance; *** = of high importance; −ve* = costs out-weigh benefits – such a response can decrease fitness in the specific environment; NA = not applicable; superscript (a) represents Isoetids that have weak barrier formation facilitating entry of sediment CO_2 for internal diffusion via lacunae to the leaves for photosynthesis
Note: True aquatic species often display biochemical leaf traits for enhanced underwater photosynthesis Only biophysical leaf traits are considered here, as our focus is on terrestrial species
ROL radial oxygen loss, *ROS* reactive oxygen species

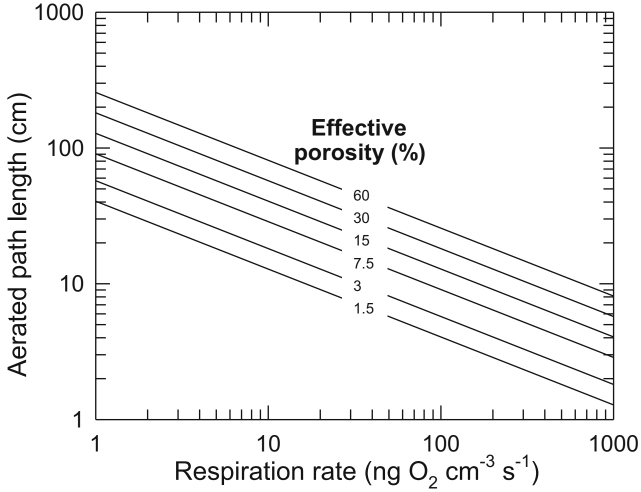


Fig. 1 Predicted lengths (aerated path length) to which roots might grow in waterlogged soil at 20 °C supported only by internal O_2 diffusion from the shoot. Based on the equation $C_o = ML^2/2D_e\varepsilon\tau$, (Armstrong 1979), where C_o is the source pO_2 (*ca* 270×10^{-6} g cm^{-3}); M, the respiration rate; L, the maximum aerated path length; D_o the O_2 diffusion coefficient (0.201 cm^2 s^{-1}); ε, the fractional porosity (FP) ; τ, a tortuosity factor. Respiratory demand and porosity were assumed to be uniform along the diffusion path, tortuosity (=1) and no ROL. Also a zero order reaction for O_2 consumption vs. internal pO_2 was assumed since O_2 consumption at the cellular level is unaffected by [O_2] down to very low levels (≤1 kPa)

demand > reduced ROL (Armstrong 1979). Species with highest root FP are usually the deepest rooting (Justin and Armstrong 1987).

Rice root growth and respiration decline when the O_2 partial pressure [pO_2] in the cortical gas-space near the apex approaches 2 kPa. Other species (not just wetland) behave similarly (Armstrong and Beckett 2011), but it is unclear at what internal pO_2 growth stops or where in the root the critical point might be. However, this pO_2 is likely to be close to zero and in nonporous meristematic tissue. The apex and stele are least likely to receive an optimal O_2 supply (Armstrong et al. 1994) and the first tissues where anaerobic metabolism will be required. However, wetland species generally have narrow steles that help minimize the O_2 deficit that can develop radially across them (Armstrong et al. 2000). Since *root growth* is *O_2-dependant* and stops at some very low value before anoxia can become pervasive, this is, in a sense, *an anoxia avoidance strategy*; localized anoxia/severe hypoxia may not be lethal. Anaerobic metabolism in stele or meristem may result in ethanol leakage into the cortex where it can be aerobically catabolized.

In some wetland species, air is driven through the rhizomes by daytime pressure flows (convections) from emergent leaves/shoots, while at night diffusion predominates (Armstrong et al. 1992; Brix et al. 1992). Aerobic conditions prevail internally during the day but localized hypoxia/anoxia may develop at night. Even in species without convective flows, e.g., *submerged* rice, diurnal changes in O_2 supply from

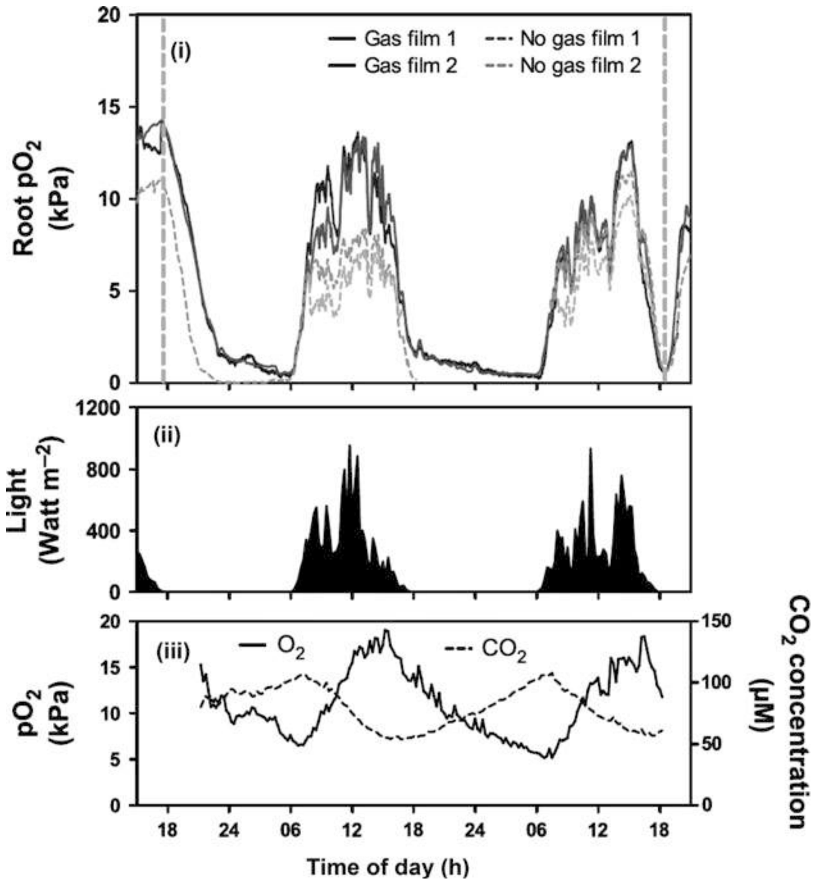


Fig. 2 Partial pressures of O_2 (pO_2) in roots of rice (*Oryza sativa*) and floodwater O_2 and CO_2 during a complete 2-d submergence event in a bunded paddy field. (i) Adventitious root pO_2 was measured using O_2 microelectrodes inserted ca. 200 μm into the root, 10 mm below the root–shoot junction, in two plants with intact leaf gas films (*solid lines*) and in two plants in which the gas films had been removed by brushing with 0.01% Triton X-100 just before submergence (*gray dashed lines*); *vertical dashed lines* represent submergence/de-submergence. Roots were buried in the anoxic soil with the root–shoot junction ca. 4 cm below the soil surface, so that O_2 entered roots from the shoot via aerenchyma. Only intermittent data are available during the night for plants without leaf gas films (i.e., no data visible in the recordings of O_2 shown in part (i) when it was below detection). Water depth varied from 48 to 50 cm above the soil during submergence. (ii) Surface light was measured ca. 440 m from the bunded field using a weather station. (iii) Floodwater pO_2 (iii; *solid line*) was measured using a Clark-type O_2 electrode; floodwater CO_2 (iii; *dashed line*) was calculated from the pH, temperature, and electrical conductivity (Reproduced from Winkel et al. (2013), New Phytologist, with permission from John Wiley and Sons)

photosynthesis (Fig. 2) (Winkel et al. 2013) can result in root growth stopping at night and resuming at first light (Waters et al. 1989). A short-term requirement for anaerobic metabolism then complements the over-riding strategy of anoxia avoidance via internal O_2 transport (Waters et al. 1989).

Shoot submergence: the low O_2 escape syndrome (LOES) and other traits. LOES-type plants have the ability to emerge above floodwaters re-establishing gas-phase continuity between buried/submerged organs and the atmosphere (Bailey-Serres and Voesenek 2008). In *Rumex palustris*, petioles re-orientate upwards and subsequently elongate, both processes being stimulated by entrapment of the endogenously produced plant hormone ethylene, and a signal cascade involving other hormones and processes modulating petiole growth (Voesenek et al. 2006). In deepwater rice, endogenous ethylene together with gibberellic acid (GA) is involved in rapid (e.g., 25 cm/day) internode and leaf sheath extension to raise foliage above the water surface during long-duration floods. Free-floating aerenchymatous adventitious roots emerge from the submerged nodes. These roots might replace the original basal system (malfunctioning because of O_2 deficiency/ phytotoxins) and can be photosynthetically competent, contributing at least to their own respiratory substrate needs (Colmer and Voesenek 2009). Free-floating roots are common to several wetland species.

Many rain-fed rice varieties of moderate stature respond to ethylene with some elongation but cannot outgrow their submergence. When waters recede, the elongated shoot collapses due to water deficits and/or ROS damage (Blokhina et al. 2003; Bailey-Serres and Voesenek 2008; Colmer and Voesenek 2009). However, a few rice land races that do *not* elongate are tolerant of short-term complete submergence endowed by: an insensitivity to ethylene with substrates conserved by a lack of shoot extension, a slow down of leaf senescence, down-regulation of other nonessential processes, induction of anaerobic energy production in tissues that become anoxic, and upregulation of ROS defenses to avoid damage upon O_2 re-entry (Bailey-Serres and Voesenek 2008). ROS defense might also be needed to cope with large diurnal O_2 fluctuations in submerged plants. Although roots might experience some anoxia/ severe hypoxia during nights, shoot tissues continue to have access to O_2 via leaf gas films at night and from photosynthesis when light is available.

Gas films on leaves and leaf sheaths of some submerged and partially submerged plants are an aid to anoxia-avoidance. If in contact with the atmosphere, gas films aid O_2 transport from the atmosphere (Beckett et al. 1988). Where submergence is total, and leaves cutinized, gas films reduce resistance to CO_2 entry from the water via the stomata and facilitate underwater photosynthesis; these films also reduce resistance for O_2 entry at night, e.g., submerged rice (Winkel et al. 2013). The beneficial role of gas films on leaves of terrestrial wetland plants is highlighted in Colmer et al. (2011) and Winkel et al. (2013).

Photosynthesis under water. Morphological and anatomical leaf traits that reduce resistance to gas-exchange when submerged, as well as biochemical traits collectively termed carbon-concentrating mechanisms, enhance underwater photosynthesis (Maberly and Madsen 2002). Specific examples include: (i) leaves that lack or have a reduced cuticle so facilitating inward diffusion of bicarbonate (HCO_3^-), CO_2, and O_2 to/from the water, e.g., underwater water-lily and *Rumex palustris* leaves (Colmer and Voesenek 2009; Voesenek et al. 2006) and (ii) crassulacean acid metabolism (CAM)-type metabolism for night time fixation of CO_2 (e.g., *Isoetes* sp.) (Maberly and Madsen 2002). In addition, plants may use sediment CO_2 for photosynthesis when completely submerged, the major CO_2 source in some species such as *Lobelia dortmanna* (Pedersen et al. 1995).

Tolerance of Anoxia

True quiescence is central to prolonged anoxia tolerance. Few plant organs (some seeds, over-wintering rhizomes and turions) survive prolonged total anoxia, living in a kind of anaerobic retreat. Bud extension can also occur anaerobically in only some species – a switch from quiescence to shoot growth and LOES (Colmer and Voesenek 2009).

Prerequisites for long-term survival of anoxia are: (i) the control of energy metabolism (i.e., production and consumption) under O_2 deprivation and (ii) the presence of considerable carbohydrate reserves (Brandle and Crawford 1987). The former involves changes in transcription and thus transcript abundances of certain genes and translational regulation also of synthesis of proteins; this altered state may be induced by a sensing of energy depletion (or resulting changes, e.g., lowered cytoplasmic pH) or as recently suggested directly by O_2 deficiency (Gibbs et al. 2011; Licausi et al. 2011), but it is uncertain at what internal O_2 concentration this occurs. The suggestion, however, that some plant species/tissues might downregulate respiration in response to declining O_2 supply even at relatively high concentrations has been challenged (Gibbs and Greenway 2003; Armstrong and Beckett 2011) as the interpretations had not taken into consideration the existence of O_2 gradients created by diffusive resistance and O_2 consumption in tissues of low porosity, leading to "anoxic cores." Gibbs and Greenway (2003) concluded that high anoxia tolerance must be to a large extent associated with directing the sparse energy produced under anoxia to processes crucial to survival and only in some strategic organs to growth. Maintenance of transmembrane gradients is crucial, otherwise compartmentation between cytoplasm and vacuole will fail and death will ensue. Metabolic differences between anoxia tolerant and intolerant tissues are set out in Fig. 3.

Wetland plant rhizomes are exposed to long periods of anoxia as the result of autumnal dying down of foliage and loss of gas-phase connection to the atmosphere. Prior to this, carbohydrates, mostly starch, constitute up to 50% of rhizome dry weight. Downregulation of metabolism conserves these carbohydrates for sustaining maintenance processes during anoxia, for supporting renewed growth in the spring, and for processing ROS scavenging molecules on re-exposure to O_2 (Colmer and Voesenek 2009).

Reactive Oxygen Species (ROS) and Post-anoxic Injury

Excessive formation of ROS (e.g., superoxide radicals, hydroxyl radicals, hydrogen peroxide, singlet oxygen) during low O_2 conditions and upon re-aeration is common amongst plants (Blokhina et al. 2003; Vartapetian et al. 2008; Colmer and Voesenek 2009). An important source of superoxides is the mitochondria, and superoxide can

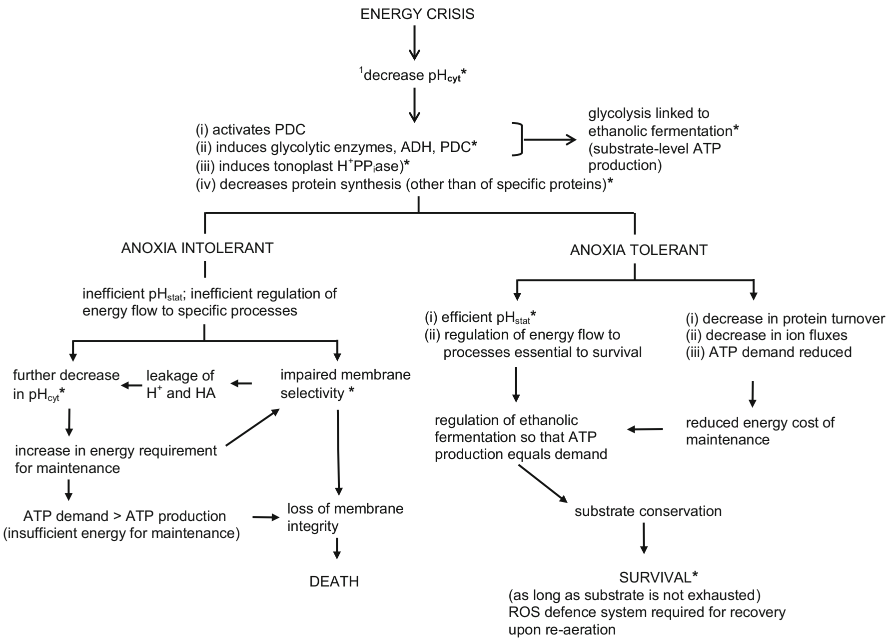


Fig. 3 A scheme for events that follow the onset of anoxia, and therefore an energy crisis, in plant tissues, indicating possible mechanisms of tolerance and causes of intolerance to anoxia (Reproduced, with a few modifications, from Greenway and Gibbs (2003), Functional Plant Biology, with permission of CSIRO Publishing). *Asterisks* are as used by Greenway and Gibbs (2003) to indicate processes for which substantial evidence exists, whereas other processes indicated are supported by limited evidence or in some cases are speculative. Note: [1]Decreased pH of the cytoplasm was regarded as a primary signal in anoxia (Greenway and Gibbs 2003). Responses of plants to conditions of hypoxia (i.e., low, but not zero, O_2) could involve sensing of changes in: (i) energy charge, (ii) cytosolic pH, (iii) cytosolic Ca^{2+} activity, (iv) levels of ROS and RNS (reactive oxygen and nitrogen species, respectively), (v) possibly O_2, and in the longer term, (vi) altered carbohydrates availability (see text). *ADH* alcohol dehydrogenase, *ATP* adenosine triphosphate, *H^+PP_iase* transmembrane proton carrier (proton pyrophosphatase), *PDC* pyruvate decarboxylase, *pH_{cyt}* cytoplasmic pH, *ROS* reactive oxygen species

quickly dismutate to hydrogen peroxide (Blokhina et al. 2003). The antioxidant defense system consists of redox-active small molecules and enzymes involved in either the direct detoxification of ROS or in regeneration of antioxidant molecules to their reduced states that can "mop up" ROS (Blokhina et al. 2003; Vartapetian et al. 2008; Colmer and Voesenek 2009).

Phytotoxins

ROL is the first line of defense against soil-generated phytotoxins. Chemically reduced elements such as Fe^{2+}, Mn^{2+}, and H_2S are converted to their oxidized counterparts in, and at the outer fringes of, an oxygenated rhizosphere

(Armstrong 1979; Armstrong et al. 1992; Colmer and Voesenek 2009). The overlapping oxygenated rhizospheres of closely spaced laterals may be particularly effective at protecting from phytotoxins by preventing initial production of reducing compounds and anaerobic digestion of organic matter, as well as encouraging an aerobic microflora and nitrogen transformations to nitrate (Colmer and Voesenek 2009). Subapical barrier formation in the exodermis is a second line of defense (Armstrong et al. 2000; Watanabe et al. 2013) and probably most necessary for the maturing parts of adventitious roots prior to lateral root emergence.

Future Challenges

Plant metabolism enabling survival of severe hypoxia/anoxia is a research area attracting much attention. In particular, sensing and signal transduction pathways are important to elucidate to understand wetland plant physiology, and these are also potential targets for use in crop improvement of flooding tolerance. Responses to submergence, of which altered O_2 supply is one important factor, involves perception of several environmental and plant changes and various transduction pathways (Voesenek et al. 2006). Responses to low O_2 in plant tissues could involve sensing of changes in: (i) energy charge, (ii) cytosolic pH, (iii) cytosolic Ca^{2+} activity, (iv) levels of reactive oxygen and nitrogen species (eg., NO), (v) possibly O_2, and (vi) altered carbohydrate availability (Greenway and Gibbs 2003; Licausi and Perata 2009). Recently, responses of expression of some genes to low O_2 availability have been shown to involve degradation of specific transcription factor regulatory proteins (Gibbs et al. 2011; Licausi et al. 2011). The various possible sensing and signaling processes in O_2-deficient cells and tissue need to be evaluated further to develop an integrated understanding (e.g., models) of the various regulatory pathways, for a range of sensitive (i.e., dryland) and tolerant (i.e., wetland) species.

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Symbioses: Assisting Plant Success in Aquatic Settings

46

Kevin J. Stevens, Bishnu R. Twanabasu, and Demetra Kandalepas

Contents

Abstract

Plants form associations with fungi in leaves, stems, and roots. Mycorrhizas develop in roots. Generally, plants benefit through increased nutrition in exchange for carbohydrates. Mycorrhizas are associated with increased tolerance to environmental stressors and competitive ability. While water quality and quantity affect mycorrhizas, several types are found in wetlands. Arbuscular mycorrhizas are the most widespread, being characterized by internal branched structures (arbuscules). Dark septate endophytes are a poorly studied but wide-ranging group that are seen as darkly colored hyphae on and within roots. Ectomycorrhizas are associated with tree species and while present in wetlands, their function is unclear. Orchids and ericaceous plants form mycorrhizas called orchid and ericoid mycorrhizas; their functions in wetland plants are also poorly studied. Fungi living in shoot tissues (shoot

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endophytes) are ubiquitous being found in terrestrial and wetland environments. While altered hydrology can affect wetland endophyte ecology, the impacts on vegetation are currently unknown.

Keywords
Symbiosis · Mycorrhiza · Shoot endophytes · Fungi · Roots · Translocation · Macronutrients

Introduction

Many plant species form intimate associations with fungi. The associations may physically take place in leaves, stems, and roots. Mutualistic associations occurring in roots are termed mycorrhizas (myco = fungus, rhiza = root). These associations are formed by the majority of plant species and involve a wide range of fungal groups (Smith and Read 1997). In above ground tissues, stems, and leaves, the fungal partners usually reside within the host tissues and are referred to as shoot endophytes (endo = inside, phyte = plant). The role of mycorrhizal fungi and shoot endophytes in wetlands is poorly understood. However, with increased recognition of wetland importance and greater attention being paid to wetland processes, their prevalence and their *potential* contributions to wetland plant growth and community structure are receiving greater attention.

Mycorrhizal Associations

Arbuscular Mycorrhizas Of the various types of mycorrhizal associations, arbuscular mycorrhizas (AMs) are the oldest and most widespread (Smith and Read 1997). Evidence of arbuscular mycorrhizal associations has been found in fossils of the earliest vascular plants (Taylor et al. 1995). The majority of vascular plant families and over 85% of vascular plant species form arbuscular mycorrhizal associations with soil-dwelling fungi belonging to the phylum Glomeromycota (Schüßler et al. 2001). Outside of a host root, the fungus persists either as spores or as a network of thin, thread-like structures called hyphae, which grow throughout the soil (Fig. 1). In the presence of a suitable host, the fungal hyphae grow toward the host root following a trail mapped out by signaling compounds released by the host. Once a hypha contacts the epidermal surface of the host plant root, a specialized structure, the appressorium, is formed that facilitates the entry of the fungus into the host root (Garriock et al. 1989). The fungus, upon breaching the epidermis, enters the cortex of the root and proliferates. Within the root the fungal partner forms highly branched structures, arbuscules, or in some cases hyphal coils, within cortical cells (Peterson et al. 2004). It is largely through these structures that the exchange of materials occurs between the fungus and the plant. The plant provides the fungus, an obligate symbiont, with carbohydrates, while the fungus provides the plant with minerals and water. The enhanced mineral nutrition in mycorrhizal plants confers a greater capacity to withstand environmental stressors such as drought, contaminant

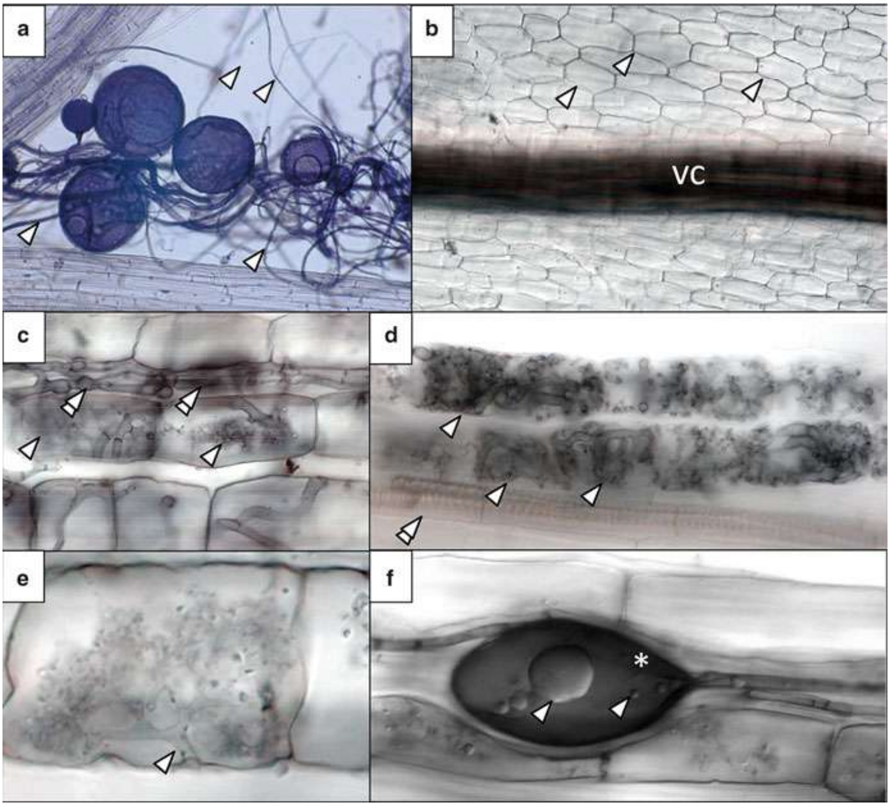

Fig. 1 Arbuscular mycorrhizal fungi. Arbuscular mycorrhizal fungi are the oldest and most widespread mycorrhizal symbiosis. Outside of the host root, the fungal partner produces thread-like extraradical hyphae (**a**; *arrowheads*) that forage for nutrients and contribute to soil stability. Larger, globular spores are produced that serve as fungal propagules allowing the fungus to survive inhospitable environmental conditions and provide a mechanism for dispersion to new areas. Clearing of plant roots by high temperature and caustic solutions break down the cellular contents, leaving only plant cell walls and fungal hyphae (**b**). In a noncolonized root, the lignified vascular cylinder (**b**; *VC*) is visible. Only the cell walls (**b; arrowheads**) remain of cortical cells (**b; asterisk**). Following staining with a stain specific for mycorrhizal fungal cell walls (e.g., Chlorazol Black **e**), arbuscular mycorrhizal fungi show up as gray structures within the cortical cells of the host (**c–f**). Internal to the host plant cell wall, the fungal partner produces arbuscules (**c**, **d**, **e**; *arrowheads*), highly branched structures involved in the exchange of materials between the fungus and its host, and intracellular hyphae (**c**; *double arrowheads*). Spiral secondary cell wall thickenings (*double arrowheads*) of the host vascular tissue are evident in **d**. After entry through the cell wall, the trunk hyphae (**e**; *arrowhead*) branches dichotomously. Subsequent dichotomous branches give rise to the highly branched arbuscules. Storages structures, vesicles, (**f**; *asterisk*) are also produced within the host root. Lipid droplets (**f**; *arrowheads*), an energy reserve are evident within the vesicle. Note the noncolonized cortical cells above and faint arbuscules (*arrowhead*) below the vesicle (Images are reprinted from Aquatic Botany, 92/2, Kevin J. Stevens, Misty R. Wellner, Miguel F. Acevedo, Dark septate endophyte and arbuscular mycorrhizal status of vegetation colonizing a bottomland hardwood forest after a 100 year flood, Pages 105–111. Copyright 2010 with permission from Elsevier)

toxicity, extremes in temperature, and resistance to microbial disease. The improved nutritional status of colonized plants also results in greater competitive ability (Fig. 1).

Early researchers predicted that AM would not be present in wetland plants because AM fungi are aerobic organisms requiring oxygen for respiration. Since wetlands are often anaerobic, it was believed that AM fungi could not survive in these habitats. The absence of AMs in wetlands was supported by an often reported trend towards reduced AM colonization levels in plants sampled along a water gradient from low to high water availability. However, as more researchers turned their attention to wetlands and wetland plants, it was recognized that AM are prevalent in wetlands. Arbuscular mycorrhizal fungi colonize plants in diverse wetland habitats including prairie potholes (Wetzel and van der Valk 1996), marshes and fens (Bohrer et al. 2004), cypress swamps (Kandelepas et al. 2010), northern peat bogs (Thormann et al. 1999), and bottomland hardwood forests (Stevens et al. 2009).

AM associations are highly responsive to environmental conditions. AM colonization levels are affected not only by the amount of water availability (Stevens and Peterson 1996) but also by the timing, frequency, and duration of flooding/drought. Furthermore, AM colonization levels are influenced by water quality. As in terrestrial habitats, AM colonization tends to decrease with increasing soil phosphorus availability (Stevens et al. 2002; White and Charvat 1999). As oxygen is depleted in flooded soils, soil redox potential decreases. This change in water chemistry impacts the availability of soil nutrients. The bioavailability of the macronutrient phosphorus increases with decreasing soil redox potential. In some cases, it may be this increased phosphorus availability, rather than water availability, that leads to reduced levels of colonization in flooded soils (Stevens et al. 2002). Increasing soil/water salinity (Carvalho et al. 2003), concentrations of heavy metals (Leyval et al. 1997), and urban contaminants including the widespread antimicrobial triclosan (Twanabasu et al. 2013a, b) can also lead to reductions in colonization levels. Given that increased urbanization and climate change have been, or are, predicted to influence water quantity, availability, and quality, impacts on mycorrhizal communities are likely.

The importance of AM fungi in wetland ecosystems is unclear. Given their ability to acquire and transport nutrients to their host plants, they play a role in nutrient cycling. Fungal hyphae are orders of magnitude smaller in diameter than host root systems and have the capacity to bind soil particles. Soil binding is further facilitated by the production of glomalin, a glycoprotein that essentially "glues" soil particles together (Wright and Upadhyaya 1998; Rillig 2004); hence, AM fungi play a role in soil stability. The contribution of AM to plant performance varies. AM-inoculated plants often show differences in patterns of resource allocation between shoots and roots and differences in shoot and root morphology compared to noncolonized plants. This response is species-specific and the magnitude of the response differs even among closely related wetland species. Although mycorrhizal interactions are largely assumed to be mutually beneficial, the advantages to the plant host can vary under different environmental conditions and the nature of the relationship may range from mutualistic to parasitic under different conditions

(Johnson and Graham 2013). The species specific nature of AM associations and differential effects of environmental characteristics on colonization levels suggests that changes in AM community dynamics may lead to changes in wetland plant community structure.

Dark Septate Endophytes (DSE) Dark septate endophytes (DSE), also known as dark septate fungi (DSF), are recorded in the fossils of the aquatic angiosperm, *Eorhiza arnoldii*, indicating an association with wetland plants that extends back to the Eocene period (Klymiuk et al. 2013). DSEs are darkly pigmented ascomycetous fungi that colonize host roots by proliferation of their septate hyphae either through intercellular spaces or from cell-to-cell in host root tissues (Peterson et al. 2008) (Fig. 2). Fungal hyphae, extending from DSE propagules in the soil, contact the host root and form a network of mycelia over the root surface. Hyphae on the surface enter the host tissue via epidermal cells, root cap cells, or wounds caused by emergence of lateral roots (Peterson et al. 2004). Intraradical hyphae (hyphae within the root) proliferate and enter host epidermal and cortical cells. The hyphae may become melanized and form clusters of inflated, round, thick-walled cells called microsclerotia within cortical cells (Jumpponen and Trappe 1998). These microsclerotia contain polysaccharides, proteins, and polyphosphates (Yu et al. 2001), which are believed to enable them to remain in the older roots and act as an inoculum source for the colonization of new roots (Peterson et al. 2004, 2008).

DSE are thought to access carbon, nitrogen, and phosphorus in litter and soil, which are bound to organic compounds and not immediately available to plants. Therefore, like other mycorrhizas, DSE may enhance the performance of their hosts by improving host plant mineral nutrition. While there is limited experimental data

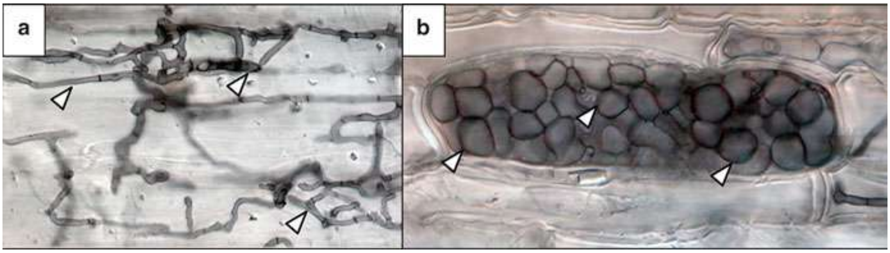

Fig. 2 Dark Septate Endophytes. Dark septate endophytes are visualized in the same manner as arbuscular mycorrhizal fungi (clearing and staining of the roots) and can often be found cooccurring with them. Hyphae within the host root, intraradical hyphae (*A; arrowheads*) enter the epidermal and cortical cells of the root. Exchange of materials between the host plant and fungus occurs through the intraradical hyphae. Within some cells, the hyphae may become melanized and produce microsclerotia, clusters of round thick-walled cells (*B; arrowheads*). Microsclerotia contain storage compounds (polysaccharides, proteins, and polyphosphates) and are thought to act as an inoculum source, facilitating the colonization of new roots (Images are reprinted from Aquatic Botany, 92/2, Kevin J. Stevens, Misty R. Wellner, Miguel F. Acevedo, Dark septate endophyte and arbuscular mycorrhizal status of vegetation colonizing a bottomland hardwood forest after a 100 year flood, Pages 105–111. Copyright 2010 with permission from Elsevier)

supporting this view, studies have shown DSE inoculation to improve plant performance compared to noninoculated plants when assessed in terms of root, shoot, and total biomass (Haselwandter and Read 1982; Newsham 1999). By improving access to carbon, nitrogen, and phosphorus, compounds essential to growing plants, DSE may be particularly beneficial during seedling establishment (Peterson et al. 2004; Solaiman and Hirata 1997; Stevens et al. 2011).

DSE are ubiquitous in terrestrial ecosystems from tropical forests (Rains et al. 2003) to alpine vegetation (Read and Haselwandter 1981). In North America, DSE have been found in wetland habitats from northern boreal wetlands in Canada to coastal wetlands of Louisiana (Wilson et al. 2004; Kandalepas et al. 2010) and from wetland types including peat lands, bogs, fens, lakes, streams, bottomland hardwood forests, and degraded marshes (Weishampel and Bedford 2006; Kandalepas et al. 2010; Kai and Zhao 2006; Kohout et al. 2012). Kandalepas et al. (2010) found DSE colonization was more prevalent than AM colonization in a degraded marsh in coastal Louisiana. They found all 18 species of the marsh plants colonized by DSE. Similarly 31 of 37 wetland species in a bottomland hardwood forest were colonized by DSE (Stevens et al. 2009), while only one species in 24 hydrophytes from lakes and 3 species among 17 stream hydrophytes were colonized by DSE (Kai and Zhao 2006). DSEs are found more commonly in monocots compared to dicots (Kandalepas et al. 2010; Weishample and Bedford 2006); however, additional studies are required to identify if this trend is seen on a larger scale and in a greater number of wetland habitats.

Orchid Mycorrhizas The family Orchidaceae is the largest angiosperm family. Plants in this family may grow in soil, as epiphytes, on other plants, or on rock surfaces. Many orchid species grow in wet habitats. Orchids produce very minute seeds devoid of endosperm (Fig. 3). Consequently, they possess insufficient nutrient reserves for seed germination and seedling growth and depend upon a fungal host for nutrient acquisition during germination and early seedling development (Peterson et al. 1998). Mycorrhizas have been extensively studied in terrestrial orchids; however, wetland orchids have generally been overlooked. Some recent studies have found that wetland orchids do harbor mycorrhizal fungi. The presence of mycorrhizal fungi has been noted by several authors in the bog orchid genus *Habenaria* (Cowden and Shefferson 2013; Stewart and Zettler 2002). In addition, eight orchid species from wet treeless habitats in Hungary and an *Epipactis thunbergii* in man-made wetlands in Hiroshima, Japan (Cowden and Shefferson 2013; Illyes et al. 2009) were found colonized by symbiotic fungi. In areas where the establishment of wetland orchids is an integral component of wetland creation or remediation efforts, Stewart and Zettler (2010) suggest *in vitro* germination and inoculation of cultured wetland orchids with mycorrhizal fungi to improve transplant success. The effects of environmental conditions on orchid mycorrhiza are poorly understood; however, evidence suggests that the saturated soils and flooded conditions are not an impediment to orchid mycorrhizal colonization. Illyes et al (2009) found orchid species colonized by symbiotic fungi from the fungal genera *Epulorhiza*, *Sebacinaceae,* and *Ceratobasidiaceae* in wetland habitats with water regimes ranging from standing water to well-drained areas.

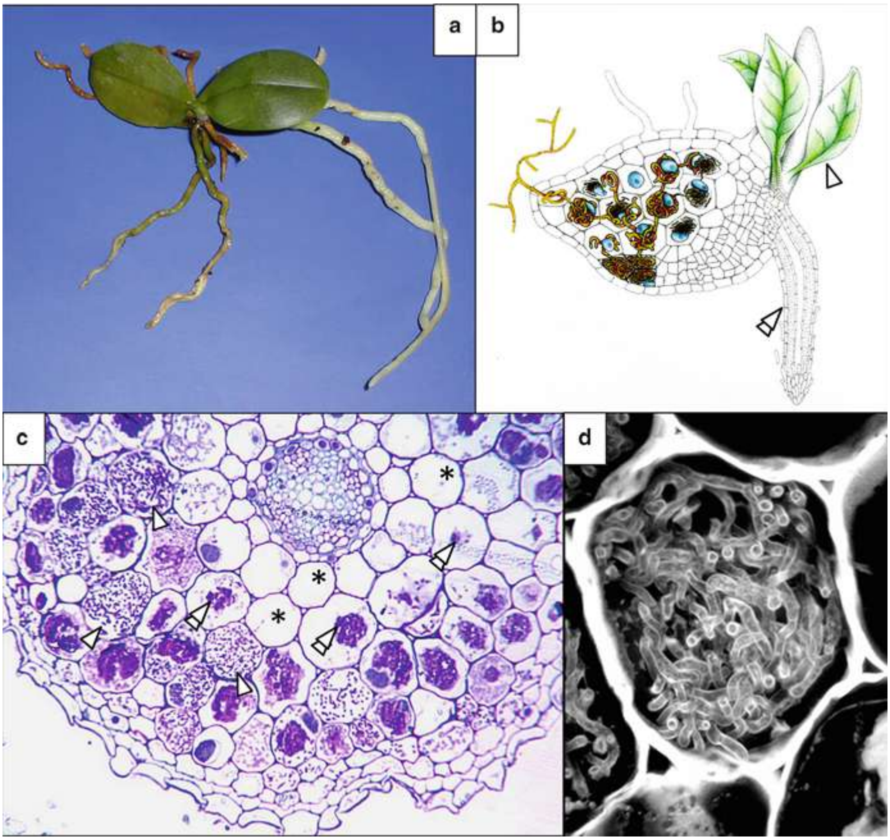

Fig. 3 Orchid mycorrhiza. While stems, leaves, and roots are clearly visible on a mature orchid plant (*A*), orchids are unique in that these structures are absent in the seed. Orchids produce numerous tiny seeds, lacking the storage tissue endosperm and containing an undifferentiated embryo lacking shoots and roots. Colonization of the seed by a fungal partner is necessary for the provision of carbohydrates needed for the seedling to develop. Following colonization, the embryo develops into a protocorm that further develops into a seedling. The shoot (*arrowhead*) and root system (*double arrowhead*) are produced by the protocorm (*B*). Within parenchyma cells of the protocorm, the fungal hyphae coils forming structures called peletons. Extensive colonization occurs in the roots (*C*). Cortical cells containing purple stained peletons contrast sharply with noncolonized cortical cells (*asterisks*). Peletons are short-lived and roots contain both viable (*C; arrowheads*) and degraded (*C; double arrowheads*) peletons. The highly coiled nature of the peleton is evident in an image obtained from a laser scanning confocal microscope (LSCM) image (*D*). By obtaining several optical slices through a specimen, the LSCM is able to construct three-dimensional images that reveal features not readily apparent with conventional light microscopy (Images are reprinted from Mycorrhizas: Anatomy and Cell Biology Images CD. R. Larry Peterson, Hugues B. Massicote, Lewis H. Melville, Forest Phillips. Copyright 2006 with permission from NRC Press)

Ericoid Mycorrhiza A unique mycorrhizal association is formed by some members of the diverse plant order Ericales and some ascomycetous fungi. Species in the Ericales include the economically valuable *Vaccinium* spp. (cranberry and blueberries) and several wetland species including *Andromeda polifolia* (bog rosemary), *Rhododendron canadense* (rhodora), *and Ledum groenlandicum* (Labrador tea). The roots involved in forming ericoid mycorrhizas are referred to as “hair roots” due to their very small (<100 um) diameter (Fig. 4). Anatomically, the hair roots consist of an epidermis, a very much-reduced cortex consisting of two layers, an outer hypodermis and inner endodermis, and a central core of vascular tissue comprised of small tracheary elements and sieve elements (Peterson et al. 2004). Colonization begins when fungal hyphae reach the root epidermal surface. Here, the fungal hyphae may branch and form a mantle-like structure or form “runner hyphae” that branch and facilitate the colonization of epidermal cells. Colonization of the host is restricted to enlarged root epidermal cells. Entry into an epidermal cell may be preceded by the formation of an appressorium although this does not occur in all cases. Once the epidermal cell wall has been penetrated, each hypha forms a coiled or branched structure, the hyphal complex, within the boundaries of the host cell wall. As with arbuscular mycorrhizal fungi, the hypha does not penetrate the plasma membrane (Perotto et al. 1995). The exchange of materials between the plant and fungus occurs through the hyphal complexes.

The role of ericoid mycorrhiza in wetland plant species has received little attention; however, information on the role of ericoid mycorrhizae in heath vegetation provides an indication of potential wetland importance. The fact that ericaceous plants are found in nutrient poor soils suggests that ericoid mycorhiza play a role in host nutrient status. The extent to which the extraradical mycelia of ericoid mycorrhizae extends beyond the host root system is hampered by the nature of peaty soils in which they often grow; however, measurements of <1 cm have been reported in sandy soils (Read 1984). Ericoid mycorrhiza can either produce the necessary enzymes or mobilize organic and inorganic forms of phosphorus (Smith and Read 2008). In heathlands, ericoid mycorrhiza facilitate nitrogen uptake absorbing nitrogenous compounds such as nitrate, ammonium, and amino acids and translocating them to the host. In addition, ericoid mycorrhiza are able to produce enzymes needed to access nitrogen bound up in more complex organic compounds such as pectin, cellulose, hemicellulose, and lignin found in plant cell walls and chitin found in fungal cell walls and invertebrate exoskeletons (Read 1996). Ericoid mycorrhizae also provide a measure of protection against the heavy metals aluminum, arsenic, cadmium, copper, and zinc (Bradley et al. 1982; Denny and Ridge 1995). It remains to be seen if these attributes are also expressed in wetland habitats.

Ectomycorrhizas While arbuscular mycorrhizal fungi are associated primarily with herbaceous vegetation, ectomycorrhizas are often associated with tree species and, although less frequently, shrubs and herbaceous plants (Fig. 5). Ectomycorrhizal associations involve an estimated 5000–6000 basidiomycetous and ascomycetous fungi (Smith and Read 2008). These fungi include many economically important mushroom-producing and truffle-producing taxa. Similar to arbuscular mycorrhiza, ectomycorrhizal fungi produce an extensive network of hyphae extending out from the host root system that is involved with the acquisition of water, nutrients, and

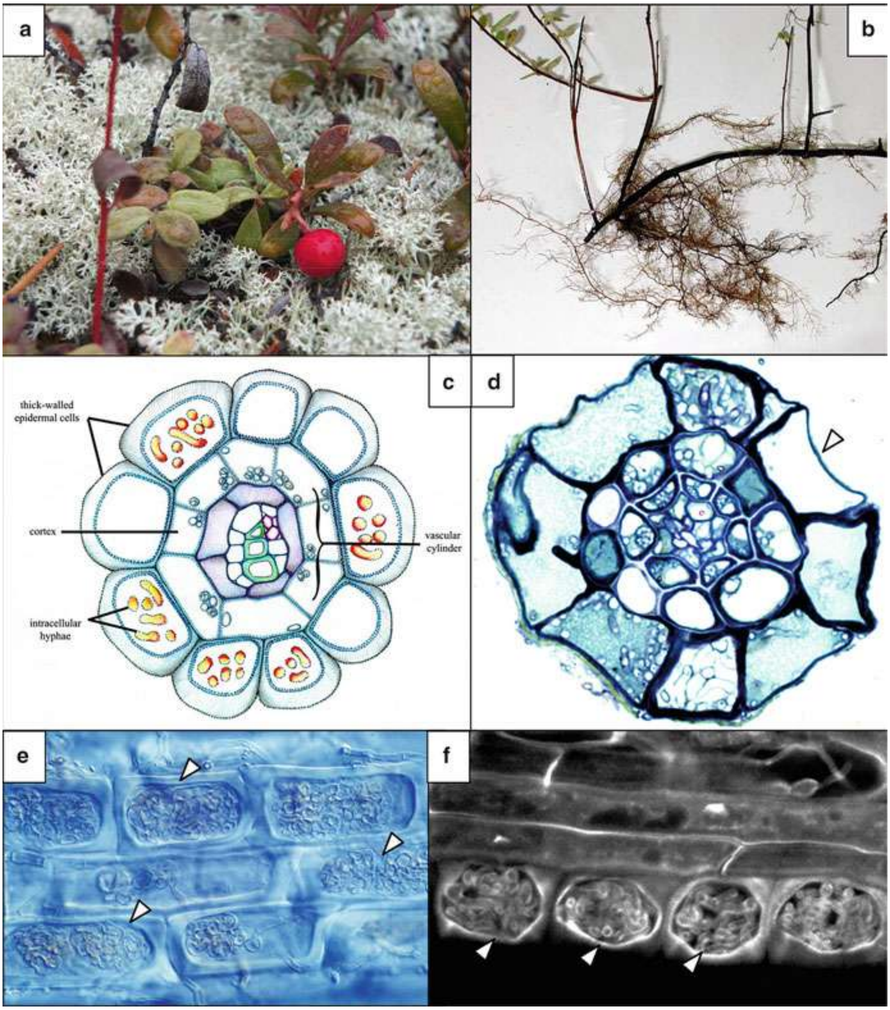

Fig. 4 **Ericoid Mycorrhiza**. Members of the order Ericales, including bog/fen species such as cranberry (*Vaccinium* sp; *A*) and Labrador tea (*Ledum groelandicum*; *B*) form ericoid mycorrhiza with ascomycetous fungi. The fine roots of ericaceous plants that are colonized are considerably reduced and often less than 100 um in diameter. The cortex consists of only two cell layers while the vascular cylinder consists of few xylem tracheary elements and phloem sieve elements (*B* and *C*). Colonization is restricted to the epidermal cells. Note the extensive colonization of epidermal cells compared to a noncolonized epidermal cell (*D; arrowhead*). The extensive hyphal coiling that occurs within the epidermal cells is evident in longitudinal sections (*E and F; arrowheads*). Note that the image in *E* was obtained with transmitted light microscopy while the image in *F* was obtained with laser scanning confocal microscopy (Images B–F are reprinted from Mycorrhizas: Anatomy and Cell Biology Images CD. R. Larry Peterson, Hugues B. Massicote, Lewis H. Melville, Forest Phillips. Copyright 2006 with permission from NRC Press)

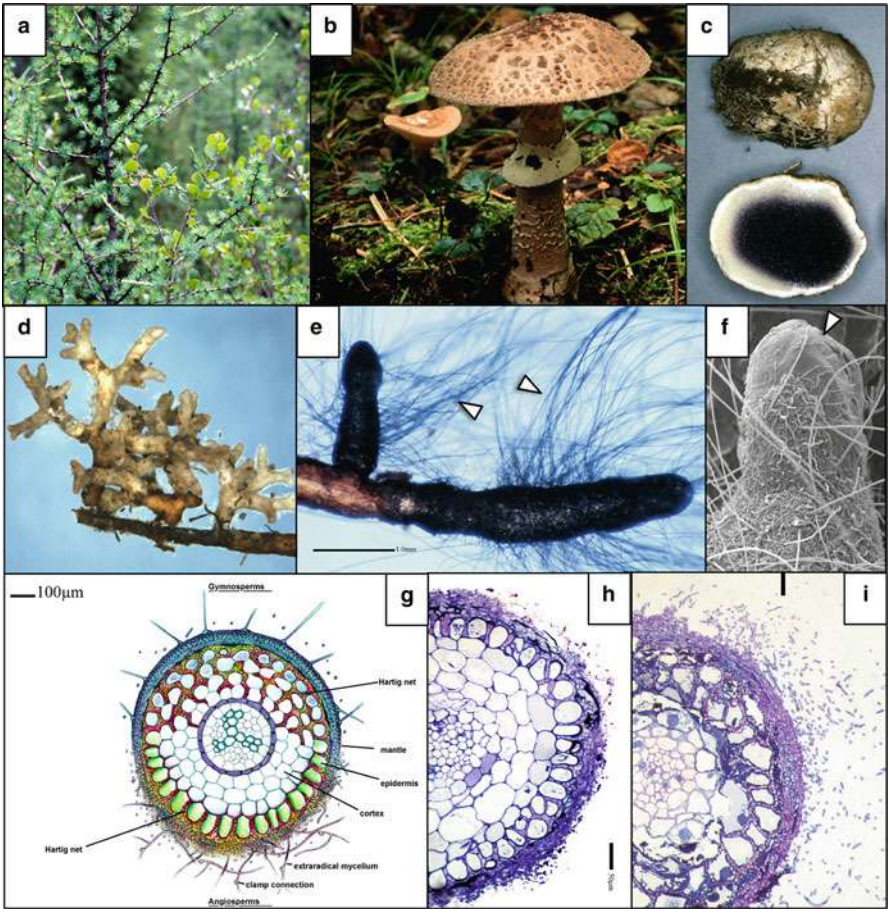

Fig. 5 **Ectomyorrhiza**. Ectomycorrhizal associations form between tree species such as tamarack (*A*; *Larix laricina*) and members of fungal taxa Basidiomycota and Ascomycota. The fungal taxa include mushroom forming species (*B*) and fungi that produce belowground fruiting bodies (*C*). Note that in *C*, the subterranean fruiting body has been cut in half to reveal the inner dark sporogenous tissue. Colonized roots often differ morphogically from noncolonized roots and typically display a dichotomous (y-shaped) branching pattern resulting in clusters of roots (*D*). Hyphae will envelop the roots forming a “mantle.” The color, thickness, density, and morphology of hyphae in the mantle differ among fungal taxa. For example, compare the light mantle hyphae in *D* to the dark mantle hyphae in *E*. Also note the extensive extraradical hyphae (*arrowheads*) emanating from the mantle in *E*. An image obtained from a scanning electron microscope further illustrates the extent of the mantle. Compare the sparsely colonized root tip (*arrowhead*) to the more densely covered region directly below the tip (*arrowheads*). Internal to the mantle, a network of fungal hyphae, the Hartig net, develops between cells of the host root (*G*–*I*). In angiosperms, this layer forms only between epidermal cells (*G* and *H*) whereas in gymnosperms the Hartig net extends throughout the cortex (*G* and *I*) (Images are reprinted from Mycorrhizas: Anatomy and Cell Biology Images CD. R. Larry Peterson, Hugues B. Massicote, Lewis H. Melville, Forest Phillips. Copyright 2006 with permission from NRC Press)

their subsequent transportation to the host plant. Extraradical hyphae, hyphae extending out from the host root, may occur singly or in groups of interwoven and interconnected hyphae called rhizomorphs. Modifications of hyphae within the rhizomorphs permit them to function in the rapid, high volume transport of materials from individual peripheral hyphae to the host. Two structures distinguish ectomycorrhizal fungi from other mycorrhiza, the presence of a mantle and Hartig net. During initial stages of colonization process, when an extraradical hypha contacts the epidermal surface of a compatible host root, the hyphae begin to branch and spread over the root surface (Peterson and Farquhar 1994). The resulting hyphal layer surrounding the root is called the mantle. The extent of epidermal root surface covered by the mantle varies but frequently the entire root surface is covered by hyphae that are several layers thick. The layers of the mantle may have unique morphological characteristics and be further classified as inner, middle, and outer mantle. A Hartig net develops from hyphae of the inner mantle. This net is a specialized structure that is involved in the exchange of materials between the plant and its fungal partner. In angiosperm species, the Hartig net develops around only epidermal cells and does not enter the cortex. In conifer species, however, the Hartig net includes several layers of cortical cells in addition to the epidermis (see Peterson et al. 2004 for details of ectomycorrhiza structure).

Many tree genera that are frequently found in wetlands (e.g. *Alnus*, *Salix*, *Pinus*, *Picea*) form ectomycorrhizal associations. However, we are unaware of any studies that have ascertained the role of ectomycorrhizae in wetland plants. Studies that have compared responses of colonized and noncolonized plants to environmental stressors under wetland conditions are lacking. Initial observations suggested that ectomycorrhizal colonization was reduced in wet soils (Slankis 1974), possibly due to anaerobic conditions, and resulting changes in the availability of soil-derived nutrients and toxins. This reduction in colonization with increasing water availability led to an investigation of ectomycorrhizal colonization as an indicator of hydrologic conditions for wetland delineations (Vasilas et al. 2004). This trend towards a reduction in colonization with increasing water availability may not necessarily pertain to all ectomycorrhizal species equally. Robertson et al. (2006) in an investigation of ectomycorrhizal communities of black spruce (*Picea mariana*) occurring in upland and wetland habitats in central British Columbia, Canada, found that while ectomycorrhizal species richness was greater in upland compared to wetland areas, each habitat type studied (upland and wetland) supported approximately 20 morphologically distinct mycorrhizal types. Furthermore, while some ectomycorrhizal fungal groups were more frequently found in upland compared to wetland habitats, this was not the case for all fungi and some were found with equal frequency in both wetland and upland areas.

Shoot Endophytes

Foliar endophytes, fungi that live within leaf tissue without causing any symptoms of disease to their hosts, may be a key component to maintain healthy wetland plant communities. Foliar endophytes are ubiquitous, and they have been found in all plants that have been examined to date. These cryptic fungi, long overlooked in the literature, are thought to be beneficial to their plant hosts, as they not only live within plant tissue without causing any signs of disease but also have been found to increase their hosts' fitness and competitiveness. Primarily, these fungi have a mitigating effect on plant responses to extreme environmental demands, such as extreme temperatures, high salinity, flooding, drought, and herbivore pressure (Bacon 1993; Malinowski et al. 1997; Arechavaleta et al. 1992; Clay et al. 1993; Marks and Clay 1996; Clay 1988; Belesky and Malinowski 2000; Van Bael et al. 2012; Colbentz and Van Bael 2013).

Shoot endophytes are a diverse group and their patterns of diversity and community composition have been attributed to many biotic and abiotic factors. In general, endophytes that are horizontally transmitted become more diverse with increasing leaf age (Arnold and Herre 2003), allowing time for more endophytes from surrounding plants to colonize younger plants. Endophyte diversity also increases with decreasing latitude (Arnold and Lutzoni 2007), a trend that has been observed for other taxa (Pianka 1966). Indeed, endophytes in the tropics have been shown to be "hyperdiverse" (Arnold et al. 2000), contributing substantially to fungal biodiversity. Arnold et al. (2000) were able to isolate dozens of isolates from single leaves. Endophyte colonization by any particular endophyte species, however, depends on host identity and physiology, as well as the interaction between host identity and the environment (Redman et al. 2001; Ahlholm et al. 2002; Gange et al. 2007; Hashizume et al. 2008; Hoffman and Arnold 2008). Host identity and the interaction to its environment may influence leaf chemistry, which affects colonization by any particular endophyte (Arnold and Herre 2003). Other factors that affect colonization by endophytes include plant physiology (Redman et al. 2001) and external leaf characteristics such as trichome density (Valkama et al. 2005). Environmental factors that have been implicated in influencing endophyte biodiversity and community composition are precipitation (Elamo et al. 1999), acid rain (Helander et al. 1993), soil moisture (Blodgett et al. 2007), soil nutrient content (Eschen et al. 2010), temperature (Hashizume et al. 2010), and altitude (Hashizume et al. 2008). In addition, stand density (Saikkonen et al. 1998) and plant diversity also affect endophyte diversity (Arnold et al. 2000).

The advantages that plants experience in the presence of endophytes may depend on the composition of endophyte communities colonizing the plant. Also, the composition of endophytes within plants after a disturbance may differ from that prior to the disturbance. Damage to leaves alters endophytic composition from a few host-specific endophytes to many cosmopolitan endophytes (Fail and Langenheim 1990), and endophyte species richness decreased in stressful conditions (Blodgett et al. 2007; Kandalepas 2012). The new endophytic communities within damaged plants may generate new dynamics within a plant host, which could either increase plant resistance to subsequent damage or decrease plant resistance to subsequent stress, herbivory, and disease. Also, new endophytic dynamics could alter an individual's competitive ability or fitness.

The ecology of shoot endophytic fungi is poorly studied. What is known is mostly from upland and agricultural studies (but see Redman et al. 2011; Kandalepas 2012). Few studies have sought to quantify shoot endophytes in wetland species but those that have (Kumaresan and Suryanarayanan 2001; Kandalepas 2012) have demonstrated that foliar endophytes are prevalent in wetlands. Many of the environmental characteristics that affect shoot endophyte ecology are altered either by anthropogenic activities or by hydrological changes associated with climate change. For example, canal construction in southeastern Louisiana wetlands has resulted in rapid erosion and, together with logging ditches, saltwater intrusion has occurred further inland. Levees and spoil banks from oil and gas canals have created impoundments, in which water first becomes anoxic and then toxic. Changes in water quality (salinity, nutrient levels) and quantity are well documented following hurricanes as is the significant physical damage or loss of vegetation. It is important to determine which endophytes exist in natural wetland systems and whether their community structure changes with the continued anthropogenic activity and climate change (Day et al. 2007). By understanding such dynamics, managers may be able to use fungi to promote the health of coastal plants under this additional pressure.

Conclusions

Many wetland plant species form intimate associations with shoot and/or root fungal endophytes. In some cases, these associations have existed for millennia. While the role and importance of endophytes in wetlands is largely unknown, there is evidence to suggest that endophytes play important roles in nutrient acquisition, herbivore defense, tolerance of soil and water contaminants, tolerance of altered hydrology (flooding and drought), and plant/plant competitive interactions. Other roles are likely to be found. Environmental changes associated with increased anthropogenic disturbance and climate change, such as altered water quality and changing hydrological patterns, may affect several aspects of endophyte ecology. Additional research is needed to fully understand the contributions that endophytes make to wetland plant ecology and ultimately how they impact valued wetland ecosystem services. Whereas the limited information possessed to date is often based on studies manipulating one environmental factor, future studies must include an assessment of the interplay among several environmental factors and endophyte ecology.

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Section VII

Overview of Wetland Management

Robert J. McInnes

Overview of Wetland Management

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Robert J. McInnes, Mark Everard, and Royal C. Gardner

Contents

Abstract

Wetlands are dynamic areas, open to influence from natural and human factors. In order to maintain the way that wetlands function, their biological diversity, and the benefits that they provide to human society, it is essential to understand their management requirements. Management can take many forms. Human history is littered with examples of unsustainable wetland management. However, in the

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latter part of the twentieth century attempts have been made to reconcile the potentially conflicting needs of a multitude of threats to wetlands including urbanization, pollution and intensive agriculture and the wider ecosystem services provided by the residual wetland areas. More sustainable wetland management techniques are slowly being introduced to reverse wetland loss and degradation and to optimize benefits for human society.

Keywords
Management · Ecosystem services · Wildlife · Legislation · Regulation

Introduction

Wetlands are dynamic areas, open to influence from natural and human factors. In order to maintain the way that wetlands function, their biological diversity, and the benefits that they provide to human society, it is essential to understand their management requirements. However, management can take many forms. Human history is littered with examples of unsustainable wetland management. For instance, the Fens of Eastern England once covered an area of approximately 500 km^2. During the seventeenth century the preeminent Dutch drainage engineer Cornelius Vermuyden undertook the task of draining the Great Fen in Cambridgeshire. Two new channels, the Old Bedford River and the Forty Foot Drain, and the drainage channel known as the New Bedford River were constructed. The managed drainage of the Fens facilitated the intensification of agriculture. However, the drying of the land caused the carbon-rich peat soil to mineralize and shrink rapidly due to oxidation, lowering much of the land to a level below that of the newly excavated drainage channels. By the end of the seventeenth century much of the land reclaimed by Vermuyden was again waterlogged and would remain so until the arrival of steam-powered pumps in the nineteenth century. Pumped drainage continued well into the twentieth century. However, in the latter part of the twentieth century attempts have been made to reconcile the potentially conflicting needs of intensive agriculture and the wider ecosystem services provided by the residual wetland areas. More sustainable wetland management techniques are slowly being introduced to reverse wetland loss and degradation and to optimize benefits for human society. Therefore, the Fens of Eastern England can be seen to demonstrate the different ends of the wetland management spectrum.

Wetland Management

Wetlands play crucial roles in supporting biodiversity, the functioning of landscapes, the cycling of water and other matter, and human wellbeing. This is today most commonly summed up in terms of ecosystem services, recognizing that management of the environment and the sharing of its benefits is a matter of societal choice with economic and wider spatial and temporal implications. Sustainable management of wetlands then is an essential basis for sound ecological, resource stewardship and equitable priorities.

The diverse benefits provided by wetlands stem from the many functions that they perform. For example, wetland hydrological functions contribute to the buffering of river and groundwater flows, habitat structure and ecosystem processes purify water and air as well as sequestering carbon and aiding fishery recruitment, while chemical processes cycle nutrients. The functioning of wetlands is the basis of their capacity to produce a range of ecosystem services.

Many wetland ecosystem services have been exploited by people, both directly and indirectly. For example, wetlands directly support important fisheries and wildfowling opportunities for subsistence, trade, and recreation. Indirectly, wetland processes also purify pollutants from land use as well as domestic and industrial waste streams. The topography and fertility of wetlands also render them suitable for conversion for agricultural uses, albeit that this commonly degrades the wetland system effectively “mining” it for short-term gain and exploiting the fertility of soils that may have built up over long timescales.

Frequently, direct and indirect benefits provided by wetlands are exploited narrowly, overlooking the diversity of other supportive services that they provide and their capacity to support a diversity of human interests in both immediate and longer-term timescales. A wide range of “negative externalities” arise from this narrowly framed use of wetlands, including, for example, liberation of sequestered carbon, disruption of hydrology, increased soil erosion, and loss of habitat for wetland species. This failure to value and include into decision-making many of the benefits that wetlands provide is a major driver of the erosion of the wetland resource. Wetlands and water then are among the most profoundly and rapidly degrading habitats globally (MA 2005).

Management of Wetlands for Ecosystem Services

There are a number of uniting themes across all of aspects of wetland management. First and foremost is the importance of managing wetlands on an integrated basis, reflecting the fact that all categories of ecosystem services and the character and functioning of the wetlands that produce them are fully integrated. Also, these subsidiary reviews suggest that wetlands have to be considered within their socio-economic context, including the security of resources underpinning a diversity of livelihoods and the equity issues inherent in their management. “Wise use” provisions of the Ramsar Convention respect the need to conserve the natural character and functions of wetland systems, including the interests of all their beneficiaries.

The balance of services and of the human interests that they support differs across wetland types. Coral reefs support valuable coastal fisheries and buffer the land/water interface from storm and tidal energy; mangrove systems too provide buffering from tsunamis and other extreme events as well as providing a diversity of food and fuel wood, while upland bogs store and purify water buffering flows of fresh water that may be important to people living in downstream catchments distant from river sources. The diversity and distribution of wetland types across landscapes is important for providing the wide range of ecosystem services securing human wellbeing. However, this wider valuation of wetland diversity and distribution is not always well or adequately reflected in conventional area-based nature conservation designations.

Conflicts Over Wetland Management

It has been widely suggested that water will become as significant a source of conflict in the future as it has been in the past. This view rather overlooks the fact that the Six-Day War of 1968 between Israel and Jordan was in reality a conflict over the flows of the Jordan River, a vital resource in this arid zone of the world, and that the ensuing annexation of the Golan Heights and other territories was as much about catchment control as pure land acquisition. Saddam Hussein's internal war against insurgent Marsh Arabs in 1990s Iraq too saw the annexation of water as a weapon of war, undermining the culture of the region. The sharing of water too is a source of friction between India and Pakistan in terms of allocation of the flows of the Indus River, just one example of many conflicts over international catchments. The "Cod Wars" of the 1960s, when Iceland unilaterally imposed exclusion zones to prevent foreign trawlers overexploiting cod stocks, demonstrate the extension of resource conflicts to coastal and open seas.

However, the world commission on Dams also highlights in the final of its seven Strategic Priorities the importance of "Sharing rivers for peace, development and security." This underlines wider recognition of the view that the sharing of water is a major driver of the making and maintenance of peace, including a detailed review of "southern African hydropolitical complex" which provides heartening evidence that international agreements on water sharing were a catalyst for dialogue, enduring agreements (even throughout periods of conflict) and also a mechanism for negotiating peace between countries at war (Turton 2005).

Collaborative sharing and comanagement of wetland systems may then be crucial for the security agenda, extending wider consensus about its central role in supporting international development.

Frameworks, Drivers, and Approaches for Delivering Successful Wetland Management

There are many drivers and approaches available to deliver success for wetland management. The single most important factor in wetland management is to define clear objectives from the start of the management process. Where there are important features, such as the threatened or rare species or the maintenance of vital natural resources which maintain human wellbeing, it is essential to define distinct management objectives for each one of these features. In other words, whoever is responsible for wetland management must be clear about what it is they are trying to achieve.

The ability to deliver on the objectives of site management will always be influencing to some extent by a range of factors. It is essential that all important factors should be identified and that their impacts on the wetlands, and particularly on those important features identified in the objectives, are considered. Some of the factors which influence successful wetland management may occur within the site boundary; however, others operate at a variety of scales, for instance, at the river basin or coastal zone scale. Similarly, some of the factors may be social or political in nature, such as changes in legislation or regulatory frameworks. All of these factors need to be considered in order to deliver successful wetland management.

The identification of clear objectives and understanding of the factors which may influence their delivery should help to resolve potential conflicts. However, within any wetland site management framework it is essential that there is room for conflict resolution if such a situation arises. In order to inform both the proactive site management and the potential for reactive management of conflicts, it is essential that a clear sight monitoring program is established and maintained. It is essential to know, and be able to demonstrate to other stakeholders and interested parties, that the objectives of being achieved, or more importantly where the factor, is compromising the achievement.

Good wetland management planning should be able to identify and describe the variety of actions required to deliver on the objectives. They should be informed by the monitoring program and should be dynamic in nature. Any wetland management plan should not just identify and describe actions; they should also estimate the cost required to implement the appropriate activities to deliver upon the objectives. Continuity of affective management and monitoring is essential. Baselines may change over time as the prevailing hydrological or social conditions evolve; therefore objectives need to be continuously revisited and appropriate monitoring and management protocols maintained to deliver against the external changes.

Good wetland management planning should also clearly demonstrate how the process is being effective and efficient. This should include a valuation not just for ecological site objectives but also for social and cost-related benefits. Consideration also needs to be given to compliance with local, national, and international obligations which the site might hold. The participation and involvement of stakeholders, including local communities and indigenous peoples, should be considered essential to ensure the effective and efficient wetland management is being achieved.

Who Are Wetland Managers?

Wetland management is a process. As such, many individuals, institutions, or stakeholders may be involved in that process. Therefore wetland managers can take many forms and will vary from site to site and from country to country. The wetland managers may include individuals who are responsible for day-to-day activities, such as managing livestock, monitoring and controlling hydrological regimes, undertaking the habitat and vegetation management, or engaging with or supervising activities involving members of the public on the site. Wetland managers may also include government officials responsible for implementing local, national, or international legislation or regulation, or they may include private landowners or tenants who are responsible for long-term management of a wetland area.

In many parts of the world, wetland management is intrinsically linked with the provision of food, water, and other resources for local communities and stakeholders. For instance, Lake Chilika on the east coast of India covers some 1,165 km^2 during the monsoon season, is an internationally important wetland system which hosts a wide range of biodiversity, including over one million wintering migratory birds, and is one of only two lagoons in the world to support the Irrawaddy dolphin. Following rapid degradation from 1950 to 2000, the Lake Chilika system suffered major declines in its fisheries, the proliferation of invasive weeds, and an overall reduction in the lake area. However, significant efforts have been made since the 1990s to reverse the degradation and to enhance the management of this important wetland ecosystem. Local beneficiaries of the restoration and sustainable management of the lake include 140,000 individuals and local communities who partake in both the management of the fisheries and partly engage in tourist activities. In this situation, the local wetland managers are also the beneficiaries of their own activities.

Future Challenges

There are many challenges involved in delivering successful and sustainable wetland management. As pressures on vital wetland resources increase, the need to understand how wetlands work and the management required to maintain the benefits which flow from wetlands increases. A genuine challenge to wetland managers is to understand the range of benefits which any wetland delivers, be they expressed as numbers of threatened species, as the number of households supplied with fish, or as the insurance value of properties protected from flooding. Only when these benefits have been identified can the appropriate objectives for the site be established and consequently the necessary monitoring and management prescriptions implemented. The failure to adequately understand the management objectives of any wetland site will increase the potential for conflicts and the risk of wetlands degradation and loss.

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Systems Scale Thinking for Wetland Management

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Mark Everard

Contents

Abstract

Systems thinking explores whole systems, which behave in ways that may not be predictable from analysis of their constituent parts. Wetlands and other ecosystems comprise complex interactions of living and nonliving components and so must be treated as integrated systems of great complexity producing a range of ecosystem service outputs. Their socioeconomic context is also highly significant in understanding the benefits of wetlands and their vulnerability to human interventions, making systemic thinking essential for effective wetland management.

Keywords

Systems · Emergent properties · Ecosystem services · Complexity · Sustainable management · Interdependence · Socioecological system · Natural infrastructure · Resilience

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Introduction

Systems thinking explores whole systems, which behave in ways that may not be predictable from analysis of their constituent parts. Wetlands and other ecosystems comprise complex interactions of living and nonliving components and so must be treated as integrated systems of great complexity producing a range of ecosystem service outputs. Their socioeconomic context is also highly significant in understanding the benefits of wetlands and their vulnerability to human interventions, making systemic thinking essential for effective wetland management.

The consequences of nonsystemic management, maximizing production of one or just a few target services while overlooking others, may be a major causative factor of wetland decline. It is therefore important to consider all aspects of wetlands in management decisions, including their context within landscapes.

Narrowly framed policy and perspectives, often backed up by “ring-fenced” budgets, contribute to management of wetlands for restricted purposes, which may contribute to ongoing erosion of the quality and quantity of wetland resources. It is therefore essential to develop a more far-seeing policy base that addresses all of the ecosystem services provided by wetlands, an economic approach that values them appropriately for inclusion in decision-making processes, and an awareness of the many values of wetlands to different elements of society.

Systems Thinking and Wetlands

Systemic understandings of wetlands reveal the complexity of these ecosystems and their many, often historically overlooked, values to society. This holistic perspective provides an essential basis for informed and potentially sustainable management.

Systems Thinking

The evolution of systems thinking, particularly since the 1970s, has unlocked new understandings of how complex systems operate and provided insights into their more effective management in fields as diverse as medicine, transport, spatial planning, climate and weather prediction, and ecosystem management. A systems approach is based on understanding the properties of systems as a whole and the relationships of their components, distinct from a reductionist approach of breaking them down into constituent parts as a means to generate understanding.

Patterns and relationships within complex systems – from the interactions of amino acids in proteins to those of cellular organelles, players in a football team, or ecosystems including human-ecosystem interactions – cannot necessarily be deduced by (reductive) analysis of their constituent parts in isolation. By definition, a system has emergent properties that exceed “the sum of the parts,” such as the catalytic properties of enzyme molecules, the capacity for consciousness arising from the mass of human brain cells, or the beneficial “ecosystem services” produced by processes occurring between ecosystem constituents in wetlands or other habitats. Addressing whole dynamic systems, and the principles that govern them, aids understanding and strategic decision-making about systems as a whole.

A systems approach is central to sustainable development, which is founded on respect for the interdependent relationships not only within ecosystems, within which all elements are indissoluble, interdependent, and essential for the production of ecosystem services, but also the wider socioecological system affecting their use and continued production.

Wetlands as Systems

Wetland ecosystems, as indeed within all habitats, are in dynamic relationship with flows of energy, matter, living organisms, and human exploitation and management activities. So, while wetlands have to be addressed as systems, we have also to be aware that systems themselves are hierarchical, dependent on wider systems including the geographical and human contexts within which the wetlands occur. These flows and influences have a profound impact on ecosystem functioning and the production of ecosystem services. Diminution of the quality and quantity of wetland habitats correspondingly reduces their potential to support human wellbeing, though the net costs of significant wetland losses observed across the world have rarely been accounted for in decisions relating to the rather narrower commercial benefits of their conversion for agricultural, infrastructure, and other human purposes.

With the rise of ecosystem services as a pedagogic and management paradigm, the multiple values of the diverse services provided by wetlands and other ecosystems have come to greater prominence. Indeed, there is growing recognition, through urban management initiatives such as “Green infrastructure” and SuDS (or Sustainable Drainage Systems), that the many beneficial services provided by natural and man-made wetland systems are of themselves a form of critical infrastructure that has been for too long overlooked in decision-making (Shiva 2008).

Recognition of the multiple values of nature and their progressive inclusion within the mainstream of policy-making across government activities is being increasingly seen as a priority, for example, in the case of the UK’s cross-government June 2011 White Paper on the natural environment (HM Government 2011). These multiple values have indeed been long recognized by the Ramsar Convention through its evolving definition of “wise use” of wetlands.

Systems scale thinking for wetland management includes understanding the role of wetlands in providing a wide range of ecosystem services beneficial to society, including the resilience of the ecosystems themselves and the interdependencies between those exploiting wetland services. A well-designed or integrated system, including, for example, an ecosystem shaped over long periods of evolutionary pressures, can absorb these forces and still maintain system functionality; conversely, a poorly designed or integrated system cannot absorb external forces, causing the system to collapse (Meadows 2008). Ecosystem resilience thereby provides a bulkhead against climate change and other environmental perturbations.

Systemic and Nonsystemic Management

Problems arise when ecosystems are managed nonsystemically, for example to maximize production of just one or a few target services to the exclusion and at net cost to other services and the integrity of the productive system (such as intensive agricultural production). As widely observed elsewhere, this has been the fate of wetlands across much of this planet, more commonly the maximization of some traded services (mainly provisioning services) resulting in widespread degradation of wetlands and their capacity to provide a wealth of nontraded (mainly regulatory, cultural, and supporting) services.

Conversely, optimization of wetland management to provide a range of ecosystem services may simultaneously protect or enhance multiple benefits for society. This diversity of benefits may be expressed over a range of scales including global (carbon sequestration stabilizing the climate), national (supporting valued biodiversity and providing amenity and tourism opportunities), catchment (floodwater attenuation and the buffering of river floods), or localized (breaking down heat islands or forming soils).

Systemic Governance

It is well known that “siloed” policies and management actions, addressing narrow goals in isolation, are contributing to the ongoing decline of the quantity and quality of the wetland resource. Wetland protection and functioning, as for other ecosystem resources, are too often treated as “conservation” issues. This anachronistic view posits wetland protection as a restraint to “growth,” ignoring the contribution that wetlands make to society. It also tends to target only protected areas, rather than the wetland functions that occur often in small pockets across wider landscapes.

Systemic management therefore requires a policy base that recognizes and integrates all of the many values potentially provided by wetlands and other ecosystems. Examples include incorporation into relevant policy areas the benefits of wetland functions for air quality enhancement, ameliorating urban and rural flooding, breaking down heat islands, harboring the predators of crop pests, provision of amenity, recreation and educational resources, the production of fiber for fuel, thatch and other economic resources, habitat for fishery recruitment and biodiversity, landscape quality, and the buffering of flows protecting water resources. Policy areas benefitting from these services need to fully integrate the values that wetlands provide, recognizing their dependency on this natural infrastructure and, conversely, their vulnerability if it is lost. Thus, wetland values must necessarily become integrated across all policy areas, rather than as at present commonly regarded by them as constraints on a more technocentric model of development.

A more far-seeing policy-base addressing all of the services provided by wetlands necessarily also requires an economic approach that values them appropriately for inclusion in decision-making and an awareness of the many values of wetlands to different elements of society. Today, inherited siloed perspectives are all too often reinforced by "ring-fenced" budgets within the constraints of single government bodies or directed at only narrowly framed outcomes. Appropriate valuation of ecosystem services, whether in monetary terms or by other means, is then essential if the full importance of ecosystems is to be incorporated into policy-making, economics, and other decision-making systems.

Systemic perspectives, policies, and economics then need to be incorporated into pragmatic operational tools supporting "real world" decision-making, bringing the functions of wetland systems into the mainstream of decision-making by reflecting the multiple values that they provide for all in society. Significant obstacles to the achievement of this goal have yet to be overcome, including, for example, the distribution of benefits and costs of wetland protection including the "profits foregone" of landowners holding back from lucrative practices (such as intensive farming or tree planting) in order to protect publically beneficial ecosystem services (such as floodwater attenuation, carbon sequestration, or support for wildlife). Novel market-based instruments such as "payments for ecosystem services" (PES) may have a role to play here in bridging these gaps.

The human system and its governance remains a major impediment to more sustainable and systemic wetland management. The term "policy resistance" describes the inherent resistance of the establishment to allow changes to affect governance systems; people seem to prefer to live with a familiar but flawed system than to allow changes that might cause uncertainty and instability (Meadows 2008).

Future Challenges

Major challenges are inherent in the legacy of narrowly framed legislation, budgets, and expectations. This is exacerbated by a perception of wetlands either as "wastelands" or else as protected sites that inhibit economic development, when the reality is that wetlands and their services represent important natural capital underpinning future human wellbeing and potential for development. The multiple values of the diverse ecosystem services provided by wetlands benefiting all in society need to be brought into the mainstream of legislation, regulation, market instruments, and societal awareness. Barriers erected by the distribution of benefits and costs of both narrowly exploitative or, conversely, multibenefit management need also to be overcome through the installation of appropriate incentive and reward systems.

This includes recognizing the value of many different types of wetlands distributed across landscapes and not just the relatively few larger wetlands within areas designated as nature conservation priorities.

Finally, "policy resistance" inhibiting societal change needs to be overcome if the benefits of wetland are to be fully recognized and brought into the mainstream of decision-making. Education about and better recognition of the societal values of wetlands may play a major role in this transition.

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Section VIII

International Framework for Wetland Conservation and Wise Use

Robert J. McInnes

Framework of International Conventions

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Royal C. Gardner

Contents

Abstract

A convention is a written agreement between two or more countries that is governed by international law. Such an agreement might be formally called a treaty, convention, protocol, or other name, but the name is not critical. The important point is that the countries that are parties to the agreement intend to be bound by its provisions. The convention most relevant to wetlands is the Convention on Wetlands of International Importance especially as Waterfowl Habitat, which is commonly referred to as the Ramsar Convention.

Keywords

International Law · Ramsar Convention · Multilateral environmental agreement

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Convention Basics

A convention is a written agreement between two or more countries that is governed by international law. Such an agreement might be formally called a treaty, convention, protocol, or other name, but the name is not critical. The important point is that the countries that are parties to the agreement intend to be bound by its provisions. The convention most relevant to wetlands is the Convention on Wetlands of International Importance especially as waterfowl habitat which is commonly referred to as the Ramsar Convention (www.ramsar.org).

The text of a convention is developed through negotiations. Since the beginning of the twentieth century, most convention texts have followed a fairly consistent format. The text of a convention typically begins with a preamble describing the contracting parties and their joint objectives in executing the agreement. Often the preamble is structured around a single, long sentence which is formatted into multiple paragraphs to improve readability. Each subsequent paragraph normally begins with a verb (recalling, recognizing, urging, etc.).

The sovereign titles or the full names of the contracting parties are usually included in the preamble, along with the full names and titles of their representatives. Often a standard clause is included to describe how their representatives have communicated (or exchanged) their full powers (i.e., the official documents appointing them to act on behalf of their respective countries) and found them in good or proper form.

After the preamble it is normal for there to be a series of numbered articles. These articles usually contain the substance of the actual agreement into which parties are joining. Each article is usually a single paragraph. In longer convention texts, the articles may be assembled or grouped under specific headings. Modern treaties, regardless of subject matter, normally contain articles governing the deposition of the final authentic copies of the treaty and describing how potential disputes will be resolved.

The end of a treaty, the closing protocol, is normally signaled by a clause like "IN WITNESS WHEREOF, the parties undersigned," followed by the words "DONE at," with details of where the treaty was executed and the date(s) of its execution. The date is typically written in its most formal, longest possible form. For example, the Ramsar Convention states: "DONE at Ramsar this 2nd day of February 1971, in a single original in the English, French, German and Russian languages, all texts being equally authentic which shall be deposited with the Depositary which shall send true copies thereof to all Contracting Parties." If the treaty is executed in multiple copies in different languages, that fact is always noted, as in the case of the Ramsar Convention, and is followed by a stipulation that the versions in different languages are equally authentic.

Bilateral and Multilateral Conventions

A convention between two countries is sometimes referred to as a *bilateral agreement*; a convention with many countries or parties is often called a *multilateral agreement*. Because of their subject matter, some conventions are also known as *multilateral environmental agreements*, or MEAs. Examples of MEAs include the Ramsar Convention; the Kyoto Protocol, which deals with greenhouse gas emissions; and the Basel Convention, which covers the transboundary movement of hazardous wastes. Several MEAs have a biodiversity focus. This cluster of biodiversity-related MEAs have separate but partially overlapping mandates and include:

- Ramsar Convention on Wetlands (Ramsar)
- Bonn Convention on Migratory Species (CMS) and its Agreements
- Convention on International Trade in Endangered Species (CITES)
- World Heritage Convention (WHC)
- Convention on Biological Diversity (CBD)
- International Treaty on Plant Genetic Resources (ITPGR)
- International Plant Protection Convention (IPPC)

Development of a Convention

Conventions can arise as a result of underlying events, such as dispute between states, or with regard to common concern. The development and implementation of a convention can be a lengthy process. For example, in the early 1960s, in light of increasing concerns about losses of wetland habitat and associated declines in waterfowl populations, a number of nongovernmental organizations (NGOs), governments, and wetland and waterbird ecologists recognized the need and potential for an international governmental agreement on wetlands. After over 8 years of discussions, meetings, and negotiations, in 1971 governmental representatives from 18 countries (with observers from other countries, intergovernmental organizations (IGOs), and NGOs also present) met in the Iranian city of Ramsar, on the coast of the Caspian Sea, to seek to finalize the terms of the treaty. Thus, "Ramsar" is not an acronym; the Convention takes its name from the location in which it was negotiated or signed just like the Kyoto Protocol or Basel Convention. On February 2, 1971, the treaty was concluded (i.e., the negotiators agreed on the final form of the text) and it was opened for signature. By its terms, the Ramsar Convention would enter into force (become effective) once seven countries agreed to become parties. The Ramsar Convention entered into force in December 1975. For a full account of the development and early history of the Convention, see Matthews (1993).

Ratification of a Convention

Conventions can be considered as "self-executing," insofar that a party puts the treaty and all its obligations in action upon accession. Other conventions may be non-self-executing and require primary legislation, through a change to domestic law, to fulfill the ascribed obligations. Like many other conventions, the Ramsar Convention text establishes several procedural options for a country to become a Ramsar party. A country can join the treaty either by a notification of accession, signature of the Convention subject to ratification (followed by ratification), or signature without reservation of ratification. Each country becoming a Ramsar party can choose which procedure to follow, according to its own ratification procedures. In the United States, for example, the US Senate must ratify a treaty, approving it by at least a two-thirds vote. The United States became a Ramsar party during the Reagan administration when the US Senate ratified the Ramsar Convention in 1987. Whichever procedure is followed, the notification of joining the Convention must be sent to the United Nations Educational, Scientific, and Cultural Organization, which is the Convention's legal depository organization. The Convention then enters into force in that country 4 months after accession or ratification.

Convention Bodies

Most conventions call for representatives of the parties (and others) to gather and meet at regular intervals to assess progress and exchange information. Often these events are referred to as a *conference of the parties* (COP) or a *meeting of the parties* (MOP). As with any treaty regime, there is a host of different entities that work together to promote a convention's objectives. The COP (or MOP) is typically the primary decision- and policy-making body of a convention. Depending on the convention, actions at such gatherings may be adopted by consensus or by vote. In the Ramsar context, for example, the COP meets every 3 years to assess how the Convention is being implemented and to consider resolutions and recommendations related to wetland conservation. It is a long-upheld Ramsar tradition that COP decisions are made by consensus.

Because decisions must be made and guidance provided between COPs, another important convention entity is the *standing committee*, which may serve as the COP's intersessional governance body. In the Ramsar Convention, the Ramsar COP elects the members of the standing committee on a regional basis, with the number of representative member parties for each Ramsar region being determined by the number of parties in that region.

Most conventions establish a *secretariat* or some other body charged with coordinating the day-to-day activities of the convention. Many MEAs also provide for the establishment of a *scientific advisory body*. These bodies, such as Ramsar's Scientific and Technical Review Panel (STRP) or the Convention on Biological Diversity's Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA), provide advice and guidance to the conventions, the parties, and other stakeholders.

The Legal Effect of an International Convention

A fundamental principle of international law is *pacta sunt servanda*: promises or agreements must be kept. This principle applies to MEAs such as the Ramsar Convention and its parties. But many MEAs' duties are written in a manner that affords the parties flexibility and discretion in how to implement them. Accordingly, in some cases, such as in the Ramsar Convention context, the obligations are often considered “soft law,” espousing aspirational goals, but not dictating binding legal obligations.

Note the legal effect of a convention within a particular country varies widely. The extent to which a convention or treaty applies directly in a country depends in part on whether that country subscribes to a monist or dualist approach to international law (Bruch 2006). A monist approach generally means that “international law is part of the domestic law of the country” (Bruch 2006). This means in some countries that an international convention is viewed as controlling or overriding domestic or national law. In other monist countries, an international convention has the same authority as a statute or legislative decree, while in other countries domestic laws trump conventions. Despite these variations within monist countries, an important distinction is when such a country joins a convention that agreement “is directly applicable” if its provisions are sufficiently clear. In contrast, countries that have a dualist approach to international law consider international law to be separate from domestic law. Consequently, an international legal instrument, such as the Ramsar Convention, does not immediately affect a dualist country’s domestic legislation. In order for the convention to apply within that jurisdiction, the dualist country would have to enact implementing legislation, also known as the “act of transformation” (Bruch 2006). In very broad terms, civil law countries follow the monist approach, while common law countries use the dualist approach. Some countries, such as the United States and New Zealand, follow a mixed approach (Shelton 2011).

The view that the Ramsar Convention or other MEAs is soft law does not mean, however, that it is without effect on a party’s domestic laws and internal decision-making processes. In some countries, such as Australia, the Ramsar Convention and the designation of sites have been used to augment the national government’s authority over environmental issues (Finlay-Jones 1997). In other countries, Ramsar guidance has prompted the development of comprehensive wetland policies and strategic plans (Bowman 2002; Gardner 2003). A case out of the Netherlands Antilles illustrates that Ramsar resolutions can even affect site-level permitting decisions (Verschuuren 2008).

In 2006, the government of Bonaire, the local authority of the Dutch territory of Netherlands Antilles, authorized a resort to be built adjacent to Het Lac, a Ramsar site. The governor-general annulled the granting of the permit because the environmental impact assessment failed to satisfy the commitments in Resolution VIII.9 adopted by the parties in 2002. Bonaire appealed to the Dutch Crown, contending that Ramsar resolutions are not legally binding. In November 2007 Queen Beatrix issued a royal decree upholding the annulment reasoning that “resolutions, decisions and guidelines accepted unanimously by the Conference of Parties to the Convention, of which the Netherlands is a signatory, must be considered part of the national obligations under the Convention” (Newton 2007).

Not all parties accept the legal theory that consensus-approved resolutions at conferences of the parties create binding domestic obligations. There is case law to the contrary in the United States regarding the Montreal Protocol on Substances that

Deplete the Ozone Layer, for example (NRDC v. EPA 2006). Regardless of whether a convention's resolutions are considered binding law or an expression of goals, they can influence the actions of the parties and other stakeholders.

Future Challenges

International conventions, and especially those in the area of the environment as described by the MEAs, are an important component of global governance and vital mechanism for the protection of natural resources such as wetlands. In the future, compliance with the objectives of these international conventions and agreements will be essential if governments are to deliver on sustainable development and the wise use of wetlands (Gardner and Davidson 2011).

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Biodiversity-Related Conventions and Initiatives Relevant to Wetlands

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Abstract

Although the Ramsar Convention on Wetlands is the primary multilateral intergovernmental environmental agreement (MEA) addressing inland and coastal wetlands, the scope and mandate of a number of other such MEAs also cover issues relevant to wetland conservation and wise use and the conservation of wetland-dependent species. Efforts are underway to establish ways and means of streamlining multiple conventions' implementation and enhancing synergies. The Ramsar Convention has established bilateral mechanisms for such inter-convention collaboration, which is most well established with the Convention on Biological Diversity (CBD) and the Convention on Migratory Species (CMS). A number of mechanisms have also been established for coordination between the various bodies of the biodiversity cluster of conventions.

Keywords

Ramsar convention · Multilateral environmental agreements (MEAs) · Biodiversity-related conventions · Convention on Biological Diversity (CBD) · Convention on Migratory Species (CMS) · African-Eurasian Migratory Waterbird Agreement (AEWA) · Millennium Ecosystem Assessment · IPBES

Introduction

Whilst the Ramsar Convention on Wetlands is the primary multilateral intergovernmental environmental agreement (MEA) addressing inland and coastal wetlands, the scope of a number of other such MEAs also cover issues relevant to wetland conservation and wise use. The landscape of an increasing number of global (and regional) multilateral environmental agreements (MEAs) – each with different, separately negotiated mandates that overlap to varying degrees – has become more complex over the last two decades. There is increasingly wide recognition of the need to identify ways and means of ensuring that global mechanisms and on-the-ground implementation of multiple biodiversity-related conventions become more consistent, streamlined, and coordinated. There is much ongoing debate within United Nations processes about options for International Environmental Governance (IEG) reform within the United Nations system, and there have been a variety of suggestions for the "clustering" or grouping of biodiversity-related MEAs.

While this debate continues, MEAs have been working together in a variety of practical ways of increasing their cooperation and improving collaborative implementation at global and national levels, including the framework of bilateral memoranda of cooperation and joint work plans and multilateral networks such as the UNEP-initiated Environment Management Group (EMG), the Biodiversity Liaison

Group (BLG) of biodiversity convention secretariats, and the Chairs of the Subsidiary Advisory Bodies of the Biodiversity Conventions (CSAB).

It is generally recognized that the cluster of global biodiversity-related intergovernmental agreements consists of six conventions/agreements, each of which has separate but partially overlapping mandates. These are the:

Ramsar Convention on Wetlands (Ramsar)
Bonn Convention on Migratory Species (CMS) (and its agreements)
Convention on International Trade in Endangered Species (CITES)
World Heritage Convention (WHC)
Convention on Biological Diversity (CBD)
International Treaty on Plant Genetic Resources for Food and Agriculture (ITPGRFA)

This chapter outlines the current state of cooperation (a) bilaterally between other conventions and Ramsar, focusing on CBD, CMS, and WHC, and (b) multilaterally on cooperation and information-sharing mechanisms in the biodiversity convention cluster. The Ramsar Convention is recognized as having been at the forefront of developing and implementing practical mechanisms for achieving such synergies.

Information on the full range of cooperation agreements between Ramsar and other organizations is summarized in Ramsar Convention Secretariat (2012) and supported by texts of memoranda of cooperation and joint work plans.

One major impediment to harmonization and streamlined synergies between MEAs is a lack of communication and consultation at national scale between the respective focal points of each MEA. Although parties to the biodiversity conventions have adopted repeated calls for such national-level coordination, this is not consistently in place. This can, and does, lead to situations where a country has one clear position on an issue in one MEA process, but a different position on the same issue in the process of another MEA, potentially leading to different outcomes in formal MEA COP decisions. This has been particularly marked in recent years in MEA debates on climate change and REDD+ (reducing emissions from deforestation and forest degradation) and on invasive alien species (IAS) and can lead to the same governments agreeing different decisions on the same issues in different MEAs.

Bilateral Cooperation Mechanisms on Wetlands in the Biodiversity Cluster of Conventions

The Convention on Biological Diversity (CBD) (http://www.cbd.int/)

Cooperation between Ramsar and CBD is the most well established and developed of such convention collaborations on biodiversity and is recognized as a model for how such streamlining can be done under existing multi-convention arrangements, recognizing that each MEA has distinct but overlapping scope and mandate (Davidson and Coates 2011). It has developed progressively and incrementally over the last 20 years.

The Ramsar–CBD partnership began with a Memorandum of Cooperation in January 1996 and in the same year with the CBD's recognition by its contracting parties (governments) of the Ramsar Convention as its "lead implementation partner" on wetlands (CBD Decision III/21), i.e., implementation across the scope of Ramsar's wetland coverage of inland, coastal, and human-made wetlands. Thus, implementation of the commitments under the Ramsar Convention is simultaneously an implementation of CBD commitments across the range of its thematic and crosscutting programs of work, including the ecosystem approach, for wetlands (see Finlayson et al. 2011 for discussion of the relationships between the CBD "ecosystem approach" and Ramsar's "wise use" of wetlands).

Subsequently, collaborative implementation has been guided by five successive Joint Work Plans, from 1998 onward, and with the current fifth plan being for 2011–2020, the updating of the Memorandum of Cooperation in 2005, and repeated recognition of the importance of such collaboration between MEAs in successive CBD COP decisions and Ramsar COP Resolutions.

Collaboration has moved from these affirmations of common interest, through recommending or adopting guidance prepared by one convention for use by the other, such as CBD's impact assessment guidelines adopted by Ramsar with interpretation for their application in the Ramsar context, to joint review and development of programs and guidance. Examples include joint preparation and publication of technical reports requested by parties and Ramsar's lead in undertaking the 2010 review and revision of the CBD program of work in the biological diversity of inland waters, which provided the basis for the adoption of CBD COP10 Resolution X/28 on inland water biodiversity.

Other recent areas of work requested of Ramsar by the CBD include further development of criteria for identifying sites of international importance for wetland biodiversity (cf. Ramsar Site designation criteria) and developing a framework for joint reporting on inland waters and other wetlands. For the latter, this already has been largely achieved through the parallel development of complementary national reporting formats by the two conventions. Ramsar's format largely concerns implementation process reporting while the CBD's national reports largely require ecological outcome reporting in relation to achievements toward the 2010 biodiversity target and now the 2020 Aichi Biodiversity targets.

In addition, CBD's COP10 (2010) decision on inland waters biodiversity invited the Ramsar Secretariat and its Scientific & Technical Review Panel (STRP), working with the CBD Secretariat, to establish an expert working group to review available information and provide key policy relevant messages, on maintaining the ability of biodiversity to continue to support the water cycle – work which is currently underway as a major assessment of the roles of different ecosystems in the water cycle.

The same decision also requested the CBD Secretariat to analyze information in the CBD's Fourth National Reports on the status and trends of wetlands and drivers of change in wetlands to strengthen mutual information flow between the two conventions and to inform the planned Ramsar "state of the world's wetlands" report.

However, despite these significant areas of collaboration, there continue to be several constraints and limitations in the extent of the Ramsar–CBD cooperation achieved to date and prospects for the future.

First, most achievements have so far been at the global scale, through the collaborative work of the secretariats and the conventions' scientific and technical subsidiary advisory bodies (Ramsar's STRP and the CBD's Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA)). Although called for in both CBD and Ramsar's decisions on collaboration between conventions, there has been much less consistency in the extent of national-scale communication and collaborative implementation including between the conventions' respective national focal points. This in turn has been recognized as a key precondition to establish harmonized national reporting, as has been repeatedly called for in discussions on synergies UNEP-WCMC 2009).

Second, and despite the terms of CBD Decision III/21, CBD parties and their decisions have subsequently focused on Ramsar collaboration mainly in the context of only the CBD program of work on inland waters. CBD parties, in their separate development and adoption of their other biome and crosscutting programs of work, have largely ignored the relevant implementation role of Ramsar, notably in coastal and nearshore marine systems (with Ramsar coverage extending across the whole coastal zone to a depth of 6 m of permanent inundation) and in protected areas in relation to the designation and management of Wetlands of International Importance (Ramsar Sites). CBD COP10 (2010) finally gave some recognition to the relevance of Ramsar's lead implementation role for these matters, in its protected areas and marine and coastal biodiversity programs of work. However, these and other CBD programs of work continue to pay little attention to Ramsar's long-standing implementation experience on such matters.

Third, until very recently, and at least in part due to inadequate adoption of ecosystem approaches, CBD processes and decisions have failed to recognize the crucial and crosscutting relevance of water and water management if there is to be any chance of achieving successful biodiversity conservation. This includes not just managing water to maintain ecosystems (including wetlands) but also the key role that wetlands and other ecosystems play as the natural infrastructure which maintains the water cycle and so provides water security for both ecosystems and people and the delivery of most ecosystem benefits, including the many needs and uses of water which are central to human welfare.

These issues are at the core of the Ramsar Convention's text and are a major focus of its implementation, particularly since 1996 and the adoption of Resolution VI.23 on *Ramsar and Water*. This was to some extent recognized at CBD COP10 in

October 2010, particularly in its Decision X/28 on inland water biodiversity, but also in the *Strategic Plan for Biodiversity 2011–2020*, which recognized the paramount importance of water as an ecosystem service. This now lays an improved framework, facilitating enhanced collaboration among all the biodiversity MEAs, which have all committed to supporting implementation of this strategic plan. The improved recognition of water as an ecosystem service also significantly increases resonance between the *Strategic Plan for Biodiversity*, and hence the MEAs, and the significant broader social, economic, and political interest in water.

Convention on Migratory Species (CMS) (http://www.cms.int/)

The CMS mandate covers all taxa of species which are regarded as migratory, under a geopolitical definition of "Migratory species" which regularly cross national boundaries on a regular and predictable basis: "Migratory species" means the entire population or any geographically separate part of the population of any species or lower taxon of wild animals, a significant proportion of whose members cyclically and predictably cross one or more national jurisdictional boundaries" (CMS text Article 1.1 a).

The Ramsar Secretariat and the CMS Secretariat first signed a memorandum of understanding in February 1997. It sought to ensure cooperation between the two secretariats in the fields of joint promotion of the two conventions; joint conservation action; data collection, storage, and analysis; and new agreements on migratory species, including endangered migratory species and species with an unfavorable conservation status. This was renewed as a Memorandum of Cooperation (MoC) in 2012. This renewed MoC makes provision for all relevant CMS daughter agreements and memoranda to develop bilateral Joint Work Plans with Ramsar, under the framework of the MoC, although no such further plans have yet been concluded.

A three-way Joint Work Plan between the secretariats of the CMS, the African-Eurasian Migratory Waterbird Agreement (AEWA), and the Ramsar Convention was signed in April 2004. A subsequent second Joint Work Plan (2012–2014) between Ramsar and CMS only was agreed between the secretariats in 2012.

Recent areas of collaborative implementation activity are summarized in CMS Secretariat (2011). These have included:

- Joint participation in the Task Forces on Avian Influenza and Wildlife Diseases
- Development of policy for migratory flyway conservation (see, e.g., Global Interflyway Network 2012)
- Participation by CMS in Ramsar's Regional Strategy for the Conservation and Sustainable Use of High Andean Wetlands
- Joint Ramsar Advisory Missions (RAM) to Ramsar Sites under threat which are important for migratory waterbirds

There a number of agreements and memoranda established through the CMS which are directly relevant to the conservation and wise use of migratory wetland-dependent species of fish, birds, and mammals and whose implementation contributes also to Ramsar goals and strategies. These include:

Agreements

- Wadden Sea Seal Agreement
- Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS)
- Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS)
- African-Eurasian Migratory Waterbird Agreement (AEWA)
- Agreement on the Conservation of Albatrosses and Petrels (ACAP)
- Agreement on the Conservation of Gorillas and Their Habitats

Memoranda of Understanding

- Conservation Measures for the Aquatic Warbler (*Acrocephalus paludicola*)
- Conservation Measures for the Siberian Crane
- Conservation and Management of Marine Turtles and Their Habitats of the Indian Ocean and South-East Asia (IOSEA)
- Conservation Measures for Marine Turtles of the Atlantic Coast of Africa
- Conservation of High Andean Flamingos and Their Habitats
- Conservation Measures for the Eastern Atlantic Populations of the Mediterranean Monk Seal (*Monachus monachus*)
- Conservation of Cetaceans and their Habitats in the Pacific Islands Region
- Conservation Measures for the Slender-Billed Curlew
- Conservation of the Manatee and Small Cetaceans of Western Africa and Macaronesia
- Conservation Measures for the Ruddy-Headed Goose (*Chloephaga rubidiceps*) (between the Argentine Republic and the Republic of Chile)
- Conservation of Migratory Sharks
- Conservation Measures for the West African Populations of the African Elephant (*Loxodonta africana*)

For more information on each of these agreements and memoranda, see the CMS Family Portal on: http://www.migratoryspecies.org/.

For most of these agreements and memoranda, there are not yet any formally established cooperation mechanisms with the Ramsar Convention. The most well-established cooperation is between Ramsar and AEWA (http://www.unep-aewa.org/). All species covered by AEWA are "wetland dependent" and so also covered by the provisions of the Ramsar Convention and Ramsar's established mechanisms such as for international waterbird flyway cooperation, and the designation of Ramsar Sites for waterbirds also supports AEWA implementation.

A major collaborative project between AEWA, Ramsar, BirdLife International, and Wetlands International on African-Eurasian migratory waterbird flyways – the "Wings over Wetlands (WOW) UNEP-GEF African-Eurasian Flyways Project" – was successfully completed in 2010 and was designed to increase knowledge and capacity for flyway-scale waterbird conservation throughout the region. Its outputs include an online Critical Site Network (CSN) tool, the establishment of regional training centers, and a comprehensive flyways training kit. For further information and access to these tools, see: http://www.wingsoverwetlands.org/.

At the conclusion of the project, the four main project partners agreed to continue their collaboration to develop a follow-on programmatic partnership based on the *WOW Project* for flyway-scale conservation of migratory waterbirds and the wise use of wetlands and signed a Memorandum of Cooperation (MoC) on the "Partnership for the Conservation of Migratory Waterbirds and their Habitats (Wings Over Wetlands)," during the AEWA 15th Anniversary Symposium in The Hague, the Netherlands, in June 2010.

AEWA's 5th Meeting of the Parties (MOP5) in 2012 strongly recognized the important role that implementing Ramsar mechanisms contributes to AEWA (see AEWA 2012a, b) and in particular the role that designating a comprehensive network of Ramsar Sites for migratory waterbirds can support flyway-scale conservation.

The World Heritage Convention (WHC) (http://whc.unesco.org/)

A considerable number of World Heritage Sites inscribed under the World Heritage Convention (both "natural" sites and mixed "natural/cultural" sites) are or include wetlands. Ramsar is consulted (through IUCN in their capacity as advisors to the World Heritage Committee on natural sites) on any new World Heritage Site nominations by governments and which include coastal or inland wetlands.

A memorandum of understanding was signed between the Ramsar Secretariat and the World Heritage Centre in May 1999. The Ramsar Secretariat and the World Heritage officer in charge of natural sites maintain a close working relationship with a view to:

- Promoting nominations of wetland sites under the two conventions
- Reviewing reporting formats and coordinating the reporting about shared sites
- Contributing to both conventions' training efforts
- Coordinating fundraising initiatives concerning shared sites
- Encouraging the establishment of joint national committees

World Heritage and Ramsar have also worked closely on joint expert advisory missions in recent years to a number of joint sites, including Ichkeul (Tunisia), Djoudj and Diawling (Senegal and Mauritania), and Lake Srebarna (Bulgaria). The two conventions maintain a list of wetland sites common to World Heritage and Ramsar.

There are, however, some continuing disconnects between the WHC and Ramsar mechanisms which could be valuably closed in terms of improving synergies. These include that:

- Not all wetland WH Sites have also been designated as Ramsar Sites, an example being Australia's Great Barrier Reef World Heritage Site which also wholly qualifies also for Ramsar Site designation
- The placing by the WHC of a site on the list of "World Heritage Sites in Danger" does not necessarily mean that the same government also invokes Article 3.2 reporting under the Ramsar Convention, which it should. Conversely, a Ramsar Party making an Article 3.2 report for a joint Ramsar/World Heritage Site does not always lead to consideration by the WHC of placing the site also on the WH Sites in Danger List.

The Convention on International Trade in Endangered Species (CITES) (http://www.cites.org/)

A considerable number of globally threatened plant and animal species covered by The Convention on International Trade in Endangered Species (CITES) the mandate of CITES are wetland dependent, for example, some tropical peat swamp forest trees, dugong and manatees, sturgeon, and many species of globally threatened waterbirds. Thus, CITES actions and decisions to control international trade in such species directly contribute to the conservation of wetlands and their ecological character at the species level. However, there are not as yet any formal cooperation agreements in place between Ramsar and CITES processes.

The International Treaty on Plant Genetic Resources for Food and Agriculture (ITPGRFA) (http://www.planttreaty.org/)

The 2001 International Treaty on Plant Genetic Resources for Food and Agriculture aims at:

- Recognizing the enormous contribution of farmers to the diversity of crops that feed the world
- Establishing a global system to provide farmers, plant breeders, and scientists with access to plant genetic materials
- Ensuring that recipients share benefits they derive from the use of these genetic materials with the countries where they have been originated

The treaty's work is relevant to the Ramsar Convention particularly in relation to the maintenance of the ecological character of the many wetlands, including Ramsar Sites, which are used for agriculture, and the coverage under Ramsar of human-made wetlands, notably rice paddies. There are not as yet any formal cooperation agreements in place between Ramsar and the ITPGR. However, there a formal cooperation agreements between FAO (which hosts ITPGR) and the International Water Management Institute (IWMI, which is a formal Ramsar International Organisation Partner (IOP)) which could potentially support ITPGR and Ramsar collaborations.

Other Conventions and Agreements Relevant to Wetlands

United Nations Convention to Combat Desertification (UNCCD) (http://www.unccd.int/en/Pages/default.aspx)

Although not formally treated as part of the biodiversity cluster of MEAs, there is considerable complementarity between UNCCD and Ramsar. While wetlands are crucially important everywhere, they are even more vital in arid and semi-arid lands. The Ramsar Secretariat was present at the first UNCCD Conference of the Parties in October 1997, where it distributed to the delegates an information document on "Wetlands in Arid Zones." In December 1998, during the second UNCCD Conference of the Parties, the secretary-general of the Ramsar Convention and the Executive Secretary of the CCD signed a Memorandum of Cooperation between the secretariats to help increase communication between them, coordinate efforts, and avoid duplication. Subsequent practical cooperation between the secretariats has focused on common issues in Africa but has been developing only slowly so far.

The International Whaling Commission (IWC) (http://iwc.int/home)

The International Whaling Commission (IWC) was set up under the International Convention for the Regulation of Whaling which was signed in Washington, DC, on 2 December 1946. The preamble to the convention states that its intention is to provide for the proper conservation of whale stocks and thus make possible the orderly development of the whaling industry.

The main duty of the IWC is to keep under review and revise as necessary the measures laid down in the Schedule to the Convention which govern the conduct of whaling throughout the world. These measures, among other things, provide for the complete protection of certain species, designate specified areas as whale sanctuaries, set limits on the numbers and size of whales which may be taken, prescribe open and closed seasons and areas for whaling, and prohibit the capture of suckling calves and female whales accompanied by calves. The compilation of catch reports and other statistical and biological records is also required. In addition, the commission coordinates and funds conservation work on many species of cetaceans. This includes work to reduce the frequency of ship strikes, to coordinate disentanglement events, and to establish Conservation Management Plans for key species and populations.

Many species of cetaceans (great whales) covered by the IWC depend on coastal wetlands (sensu Ramsar) at different stages of their migrations and lifecycles, and so decisions by the IWC can affect Ramsar interests and commitments. However, to date Ramsar has not been involved in IWC processes, and there is opportunity for developing synergies between the two instruments.

The Antarctic Treaty (http://www.ats.aq/index_e.htm)

The conservation of Antarctic wetlands is a special case. The Antarctic Treaty and related agreements, collectively known as the Antarctic Treaty System (ATS), regulate international relations with respect to Antarctica. For the purposes of the treaty system, Antarctica is defined as all of the land and ice shelves south of 60°S latitude. The treaty, entered into force in 1961 and currently having 50 parties, sets aside Antarctica as a scientific preserve, establishes freedom of scientific investigation, and bans military activity on that continent.

Antarctic agreements relevant to the conservation and wise use of wetlands and wetland-dependent species include:

- Agreed Measures for the Conservation of Antarctic Fauna and Flora
- The Convention for the Conservation of Antarctic Seals
- The Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR)

The Protocol on Environmental Protection to the Antarctic Treaty prevents development and provides for the protection of the Antarctic environment through five specific annexes on marine pollution, fauna and flora, environmental impact assessments, waste management, and protected areas.

Annex 5, on protected areas, of the Protocol on Environmental Protection provides that "Any area, including any marine area, may be designated as an Antarctic Specially Protected Area to protect outstanding environmental, scientific, historic, aesthetic or wilderness values, any combination of those values, or ongoing or planned scientific research."

There are currently 71 Antarctic Specially Protected Areas, many of which have been designated for their importance to wetland-dependent species covered by the Ramsar Convention, such as penguins, and mammals, such as seals.

Under the Antarctic Treaty, the Antarctic is not the sovereign territory of any one country but rather as a shared resource. There has been considerable debate in Ramsar as to whether this means that Ramsar Sites can, or cannot, be designated in Antarctica, since Article 4.1 of the convention text provides for designation only within the sovereign territory of a contracting party. This matter was debated at Ramsar COP9 in 2005 following the submission by the government of Switzerland of a draft resolution on designating Ramsar Sites in Antarctica. At that time, parties upheld the view that Ramsar Sites could not be designated in Antarctica, and the draft resolution was withdrawn.

However, this debate continues, with consideration of whether such designations could be made under the convention provisions of Article 5 on international cooperation.

Arctic Council/Conservation of Arctic Flora and Fauna (CAFF) (http://www.caff.is/)

The activities of the Arctic Council of ministers are conducted through six working groups, one of which is on the Conservation of Arctic Flora and Fauna (CAFF). CAFF's activities include the Circumpolar Biodiversity Monitoring Program (CBMP) – an international network of scientists, governments, indigenous organizations, and conservation groups working to harmonize and integrate efforts to monitor the Arctic's living resources, and the preparation of the periodic *Arctic Biodiversity Assessment*, the most recent being CAFF (2013).

Since the Arctic is of major importance for its wetlands, and crucial for the breeding and survival of Arctic wetland-dependent birds (notably migratory waterbirds) and mammals such as seals, walrus, and cetaceans, there is much common ground between CAFF and Ramsar. In recognition of this, the CAFF and Ramsar signed a Resolution on Cooperation (ROC) during Ramsar COP11 in 2012. The ROC recognizes the mutual importance of Arctic wetlands to both organizations and highlights potential opportunities to collectively build and share knowledge, create awareness, and enhance capacity for understanding change in these important ecosystems.

Multilateral Cooperation Mechanisms in the Biodiversity Cluster of Conventions

The Environment Management Group (EMG) (http://www.unemg.org/)

The EMG is a United Nations (UN) system-wide coordination body on environment and human settlements. It was established in 2001 through UN General Assembly resolution 53/242 in July 1999. The EMG membership consists of the specialized agencies, programs, and organs of the United Nations including the secretariats of the multilateral environmental agreements. Although the Ramsar Convention Secretariat is not UN administered, given that the convention itself is a UN-recognized treaty, Ramsar participates fully in the EMG.

The group is chaired by the Executive Director of United Nations Environment Programme (UNEP) and supported by a secretariat provided by UNEP. The secretariat is located in Geneva, Switzerland. The EMG identifies issues on the international environmental agenda that warrant cooperation and finds ways of engaging its collective capacity in coherent management responses to those issues. The EMG works through technical meetings, Issue Management Groups, and task forces. Representatives of intergovernmental bodies, civil society, and international nongovernmental organizations can be invited to contribute.

Liaison Group of Biodiversity-Related Conventions (BLG) (http://www.cbd.int/blg/)

The Liaison Group of Biodiversity-Related Conventions (BLG) meets regularly to explore opportunities for synergistic activities and increased coordination and to exchange information. Meetings involve the senior secretariat officials of each convention.

The mandate for establishing the Liaison Group of Biodiversity-Related Conventions was set out by the Parties to the CBD, in CBD COP7 Decision VII/26. Subsequently, through CBD COP9 Decision IX/27, CBD Parties underlined the important role of the Liaison Group of Biodiversity-Related Conventions, encouraged more regular meetings, and invited the group to identify options for improved implementation of and cooperation among biodiversity-related conventions.

Furthermore, through CBD COP10 Decision X/20, the Conference of the Parties to the CBD further underlined the need for strengthened effectiveness and improved harmonization of the biodiversity-related conventions. Paragraph 11(a) requested the biodiversity-related convention to review and, where necessary, update working arrangements, and at the Second Retreat of Biodiversity-Related Conventions held in Geneva on 4 September 2011, a new *modus operandi* was agreed and signed by all six biodiversity-related conventions.

Chairs of the Scientific Advisory Bodies of the Biodiversity-Related Conventions (CSAB) (http://www.cbd.int/csab/)

In 2006, the fifth meeting of the Liaison Group of the Biodiversity-Related Conventions (BLG) considered ways to strengthen the cooperation among the scientific advisory bodies of the Biodiversity-Related Conventions and to further improve the scientific advice available to these bodies. The BLG agreed to call a meeting of chairs of the scientific and technical bodies or advisory bodies of the conventions, together with representatives of the Convention Secretariats, the GEF Scientific and Technical Advisory Panel (STAP), and the United Nations Environment Programme (UNEP).

This first meeting of the CSAB took place in Paris in July 2007. The CSAB has convened a further five times, its most recent meeting being in October 2013. CSAB meetings generally take place associated with a meeting of the scientific advisory body of one or other of the conventions. Ramsar is represented in CSAB meetings by the chair of its Scientific & Technical Review Panel (STRP).

CSAB has considered a range of scientific and technical issues and topics of common relevance to the biodiversity-related conventions, with a view to ensuring improved consistency of the presentation of such matters to each convention and avoiding duplication of effort. These have included:

- Cooperation on climate change and biodiversity
- The 2010 Biodiversity Target and a coordinated approach for the work beyond 2010
- The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets
- Management of information and knowledge
- Complementarities and gaps in scientific and technical guidance developed under the conventions
- A multi-convention collaboration on ecosystem restoration
- The role of convention scientific advisory bodies in the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)
- Harmonization of species nomenclature

United Nations Information Portal on Multilateral Environmental Agreements (InforMEA) (http://www.informea.org/)

The MEA Information and Knowledge Management (IKM) Initiative brings together Multilateral Environmental Agreements (MEA) to develop harmonized and interoperable information systems for the benefit of parties and the environment community at large. The initiative is facilitated by the United Nations Environment Programme (UNEP).

InforMEA is an MEA IKM initiative which is designed to enhance web-based access to information about, and decisions made by, all multilateral environmental agreements, including the biodiversity cluster of conventions.

InforMEA harvests COP decisions and resolutions, news, events, MEA membership, national focal points, national reports, and implementation plans from MEA secretariats and organizes this information around a set of agreed terms.

The MEA IKM initiative currently includes 43 international and regional legally binding instruments from 18 secretariats hosted by three UN organizations and the International Union for Conservation of Nature (IUCN). The initiative invites and welcomes the participation of observers involved with MEA data and information, such as the Environment Management Group (EMG), the European Environment Agency (EEA), the Food and Agriculture Organization of the United Nations (FAO), the IUCN, the UNEP-World Conservation Monitoring Centre (UNEP-WCMC), and the International Institute for Sustainable Development (IISD).

Ramsar has been closely involved in the MEA IKM Initiative since its inception and has been working to ensure that InforMEA includes access to all relevant Ramsar materials. However, owing to compatibility issues with the structure of the Ramsar Convention website, not all such information about Ramsar is currently accessible through InforMEA. The recent redevelopment of the Ramsar Convention website is anticipated to resolve this.

UN-Water (http://www.unwater.org/)

UN-Water is the United Nations interagency coordination mechanism for all freshwater- and sanitation-related matters. UN-Water was formalized in 2003 by the United Nations High Level Committee on Programmes. It provides the platform to address the crosscutting nature of water and maximize system-wide coordinated action and coherence. UN-Water promotes coherence in, and coordination of, UN system actions aimed at the implementation of the agenda defined by the Millennium Declaration and the World Summit on Sustainable Development as it relates to its scope of work. Through UN-Water the United Nations acts as "One UN."

The scope of UN-Water's work encompasses all aspects of freshwater, including surface and groundwater resources and the interface between fresh- and seawater. It includes:

- Freshwater resources – both in terms of their quality and quantity, development, assessment, management, monitoring, and use (e.g., domestic uses, agriculture, and ecosystem requirements)
- Sanitation – both access to and use of sanitation by populations and the interactions between sanitation and freshwater
- Water-related disasters, emergencies, and other extreme events and their impact on human security

The main purpose of UN-Water is to complement and add value to existing programs and projects by facilitating synergies and joint efforts, so as to maximize system-wide coordinated action and coherence as well as effectiveness of the support provided to member states in their efforts toward achieving time-bound goals, targets, and actions related to its scope of work as agreed by the international community, particularly those contained in the Millennium Development Goals and the Johannesburg Plan of Implementation (World Summit on Sustainable Development).

UN-Water full members are only UN entities, including those responsible for major funds and programs, specialized agencies, regional commissions, conventions, etc. Other organizations outside of the UN system are partners of UN-Water. Unlike Ramsar's full participation in the UN Environment Management Group (EMG), the Ramsar Convention is treated only as a UN-Water partner, which considerably limits the convention's ability to contribute its expertise on wetlands and water fully to UN-Water processes.

UN-Water coordinates, through its World Water Assessment Programme (WWAP), the preparation of the now annual and thematic "World Water Development Report" (WWDR) that focuses on different strategic water issues each year and aims to provide decision-makers with the tools to implement sustainable use of our water resources. It also includes regional aspects, hotspots, examples, and stories, making the report relevant to a broad range of readers, at different levels and in different geographical areas. The most recent edition, WWDR5 focused on "Water and Energy" (WWAP 2014), WWDR6 on "Water for a Sustainable World" (WWAP 2015) and WWDR6 on "Water and Jobs" (WWAP 2016). Future WWDRs will include in 2017 "Wastewater: the untapped resource" and, of particular relevance to the Ramsar Convention, in 2018 "Nature-based solutions for water".

The Millennium Ecosystem Assessment (MA) (http://www.millenniumassessment.org/en/index.html)

The Millennium Ecosystem Assessment (MA) was called for by the United Nations Secretary-General Kofi Annan in 2000. Initiated in 2001, the objective of the MA was to assess the consequences of ecosystem change for human well-being and the scientific basis for action needed to enhance the conservation and sustainable use of those systems and their contribution to human well-being. The MA involved the work of more than 1,360 experts worldwide. Their findings, contained in five technical volumes and six synthesis reports, provide a state-of-the-art scientific appraisal of the condition and trends in the world's ecosystems and the services they provide (such as clean water, food, forest products, flood control, and natural resources) and the options to restore, conserve, or enhance the sustainable use of ecosystems.

The authorizing environment for the MA was then primarily through the biodiversity-related conventions and in particular the Ramsar Convention, the CBD, and the UNCCD. Ramsar experts were closely involved throughout the MA process, in its design, its governance, and lead expert contributions to its reports.

This resulted in the production by the MA of several "synthesis reports," each targeted at different convention audiences, including:

- Ecosystem Services and Human Well-Being: Wetlands and Water Synthesis (for Ramsar) (http://www.millenniumassessment.org/documents/document.358.aspx.pdf)
- Ecosystems and Human Well-Being: Biodiversity Synthesis (for CBD) http://www.millenniumassessment.org/documents/document.354.aspx.pdf
- Ecosystems and Human Well-Being: Desertification Synthesis (for UNCCD) (http://www.millenniumassessment.org/documents/document.355.aspx.pdf)

The MA's codification of "ecosystem services" (defined as the benefits nature provides to humankind) has been particularly relevance to Ramsar processes and adopted guidance. It has led to the adoption of a redefinition of the Ramsar terms of "wise use" and the "ecological character of wetlands" (Ramsar Convention 2005) and formed the basis for the ecosystem service components of describing ecological character (Ramsar Convention 2008) and those parts of the Information Sheet on Ramsar Wetlands (RIS) – 2012 revision concerning the ecosystem services delivered by Ramsar Sites (Ramsar Convention 2012).

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (http://www.ipbes.net/)

IPBES was established in April 2012, as an independent intergovernmental body open to all member countries of the United Nations. The members are committed to building IPBES as the leading intergovernmental body for assessing the state of the planet's biodiversity, its ecosystems, and the essential services they provide to society.

IPBES provides a mechanism recognized by both the scientific and policy communities to synthesize, review, assess, and critically evaluate relevant information and knowledge generated worldwide by governments, academia, scientific organizations, nongovernmental organizations, and indigenous communities. This involves a credible group of experts in conducting assessments of such information and knowledge in a transparent way. IPBES is unique in that it will aim to strengthen capacity for the effective use of science in decision-making at all levels. IPBES will also aim to address the needs of Multilateral Environmental Agreements that are related to biodiversity and ecosystem services and build on existing processes ensuring synergy and complementarities in each other's work.

The Ramsar Convention (Secretariat and Scientific & Technical Review Panel) has been closely involved in IPBES processes since its inception in 2008, including its design and the development of its work program. The chair of the STRP participates fully in the IPBES Multidisciplinary Expert Panel (MEP), which also includes members who are also appointed members of the STRP.

IPBES is currently implementing the first phases of its work program, which includes several assessments directly relevant to Ramsar issues. These include thematic assessments on:

- Pollinators, pollination, and food production
- Land degradation and restoration
- Invasive alien species and their control
- Sustainable use and conservation of biodiversity and strengthening capacities and tools

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Ramsar Convention on Wetlands: Scope and Implementation

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Contents

Abstract

The Ramsar Convention on Wetlands is the longest established of the modern global intergovernmental environmental agreements (often known as multilateral environmental agreements (MEAs)). The text of the Convention was opened for signature in the town of Ramsar, Islamic Republic of Iran, on 2 February 1971. The Convention was developed in the 1960s as a response to increasing concerns about accelerating conversion and destruction of wetlands and the impact of this on both people and biodiversity, especially waterbirds. The Convention's aim is to "stem the progressive encroachment on and loss of wetlands now and in the future," and it expresses its Contracting Parties' confidence "that the conservation of wetlands and their flora and fauna can be ensured by combining far-sighted national policies with coordinated international action." The Convention is implemented through the three "pillars" of its strategic plan: the wise use of all wetlands, the designation and management of Wetlands of International Importance (Ramsar sites), and inter-national cooperation – including on shared wetlands, river basins, and populations of migratory waterbirds. The 45-year growth of the Convention is outlined, as are the Convention's bodies and

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their roles and the extensive suite of the Convention's formally adopted technical implementation guidelines.

Keywords
Ramsar · Wetlands · Contracting parties · Conference of contracting parties · Standing committee · Scientific & technical review panel · Resolutions · Technical guidelines

The Ramsar Convention on Wetlands

For further information about the Ramsar Convention, see *The Ramsar Manual* (Ramsar Convention Secretariat 2013) and the Convention's web site on www.ramsar.org. Parts of this article are derived from Gardner and Davidson (2011) and Finlayson et al. (2013).

The Ramsar Convention on Wetlands is the longest established of the modern global intergovernmental environmental agreements (often known as multilateral environmental agreements (MEAs)). The text of the Convention was opened for signature in the town of Ramsar, Islamic Republic of Iran, on 2 February 1971. Its full title is "the Convention on Wetlands of International Importance especially as Waterfowl Habitat" (UN Treaty Series No. 14583). The present text of the Convention, as subsequently amended by the Paris Protocol (1982) and the Regina Amendments (1987), is available on http://www.ramsar.org/sites/default/files/documents/library/current_convention_text_e.pdf. The United Nations Educational, Scientific and Cultural Organization (UNESCO) acts as the legal depositary of the Convention.

The Convention was developed in the 1960s as a response to increasing concerns about accelerating conversion and destruction of wetlands and the impact of this on both people and biodiversity, especially waterbirds. The origins and early history and evolution of the Convention are well described in Matthews (1993), and a summary of the Convention's history and progress on the occasion of its 40th anniversary is provided by Ramsar Convention Secretariat (2011).

The Convention's aim is to "stem the progressive encroachment on and loss of wetlands now and in the future," and it expresses the parties' confidence "that the conservation of wetlands and their flora and fauna can be ensured by combining far-sighted national policies with coordinated international action." Those responsible for drafting the Convention text were inspirational and farsighted: the adopted text recognized "the interdependence of Man and his environment"; "the fundamental ecological functions of wetlands as regulators of water regimes and as habitats supporting a characteristic flora and fauna, especially waterfowl"; and "that wetlands constitute a resource of great economic, cultural, scientific, and recreational value, the loss of which would be irreparable," the last being recently affirmed by Russi et al. (2013) in respect of the high economic values of many inland and coastal wetlands. Given the continuing conversion and loss of many wetlands worldwide, these are issues which remain as pertinent today as they were over 40 years ago.

The Convention's Mission, established in its strategic plan (for the fourth strategic plan, for 2016–2024, see Ramsar Convention 2015), is the "conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world." The strategic plan recognizes that to achieve this Mission, it is essential that vital ecosystem services, and especially those related to water and those that wetlands provide to people and nature through their natural infrastructure, are fully recognized, maintained, restored, and wisely used.

The Convention is implemented through the three "pillars" of the strategic plan (Ramsar Convention 2012): the wise use of all wetlands, the designation and management of Wetlands of International Importance (Ramsar sites), and international cooperation – including on shared wetlands, river basins, and populations of migratory waterbirds.

In 2005 the Convention redefined the wise use of wetlands as "the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development" (Ramsar Convention 2005; Finlayson et al. 2011). In turn, the ecological character of wetlands is "the combination of the ecosystem components, processes and benefits/services that characterize the wetland at a given point in time." This definition notes that within this context, ecosystem benefits are defined in accordance with the Millennium Ecosystem Assessment definition of ecosystem services as "the benefits that people receive from ecosystems" (Millennium Ecosystem Assessment 2003).

The Convention adopted a broad and inclusive definition of wetlands ranging from the mountains to the sea, covering all inland wetlands (including lakes and rivers), coastal and nearshore marine wetlands (including estuaries, mangroves, salt marshes, sea grass beds, and coral reefs) and human-made wetlands (including reservoirs, fishponds, and rice paddies). For the full range of wetland types covered under the Convention, see Ramsar Convention Secretariat (2010). The Convention's coverage of coastal and marine wetlands extends to those permanently inundated to a depth of six meters, so the Convention does not address deep-ocean ecosystems.

Convention Bodies

The Convention is governed by its government contracting parties, who meet every 3 years in a meeting of the Conference of the Contracting Parties (COP). The most recent meeting was COP12 in Uruguay in 2015. At each COP, contracting parties negotiate and adopt Resolutions which set the financial, policy, and technical implementation agenda and priorities for the subsequent triennium.

Intersessionally, Convention implementation is overseen on behalf of all contracting parties by the Standing Committee, composed of representative contracting parties appointed by the COP on a pro rata basis from each of the six Ramsar geopolitical regions (Africa, Asia, Europe, Neotropics, North America, and Oceania – see Ramsar Convention 2012). The Standing Committee meets approximately annually and takes decisions concerning the intersessional use of budgets and implementation and oversees and guides preparations for the COP.

The Scientific and Technical Review Panel (STRP), established in 1993, is the Convention's subsidiary body charged with preparing scientific and technical implementation advice and guidance at the request of the COP. At its core are (COP12 Resolution XII.5 http://www.ramsar.org/sites/default/files/documents/library/cop12_res05_new_strp_e_0.pdf) 18 members, appointed in their own right for their wetland-relevant expertise. A large number of other bodies are invited to participate as observer organizations (Resolution XII.5 Annex 2). Over the past 20 years, the STRP has prepared a substantial body of guidance to support implementation of different aspects of the Convention (see “Convention implementation guidance” below). For further information about the STRP’s work, visit its web platform on http://strp.ramsar.org/.

The CEPA (Communication, Education, Participation and Awareness) Oversight Panel, established in 2005, oversees and provides advice on the implementation of the Convention’s CEPA Programme for 2016–2024, Resolution XII.9 http://www.ramsar.org/sites/default/files/documents/library/cop12_res09_cepa_e_0.pdf.

The Ramsar secretariat, hosted by IUCN in Gland, Switzerland, is the administrative body of the Convention, with responsibilities including providing advice and support to contracting parties and organizing, and preparing documentation for, official Convention meetings. The secretariat is now notable among global MEAs in being directed solely by its contracting parties and not by a United Nations body or agency.

At national level, within each contracting party, the government appoints a Ramsar “administrative authority” which has responsibility for within-country and international cooperation implementation of the Convention. The administrative authority is expected to appoint an STRP National Focal Point and two CEPA National Focal Points (one governmental and one nongovernmental), to designate Ramsar sites, and to establish a National Ramsar (or wetlands) Committee to provide national coordination and implementation advice.

Growth and Evolution of the Convention

Since its establishment in 1971, the Convention has progressively grown and now (as at February 2014) has 168 contracting parties (Fig. 1). Each contracting party commits to the designation and management of a coherent and comprehensive national network of Wetlands of International Importance (Ramsar sites) – wetlands which qualify under one or more of nine designation criteria (see Ramsar Convention Secretariat 2010). The global network of Ramsar sites is (as at September 2016) 2,242 sites covering over 2.15 million km^2 of wetlands and associated habitats, with new site designations being made at twice the rate during the past 10 years as previously (Fig. 1). Information on all Ramsar sites is publicly available through the Ramsar Sites Information Service (RSIS) on http://ramsar.wetlands.org/. A major redevelopment of the RSIS, designed to handle data from the revised format of the Information Sheet on Ramsar Wetlands (RIS) adopted by COP11 (see Table 2), was launched in 2015.

A Convention budget (funded by annual contributions from contracting parties) and secretariat (then called the bureau – services previously provided by IUCN and the International Waterfowl and Wetlands Research Bureau (IWRB)) were not established until 1987, at COP3. Although both the budget and secretariat staffing

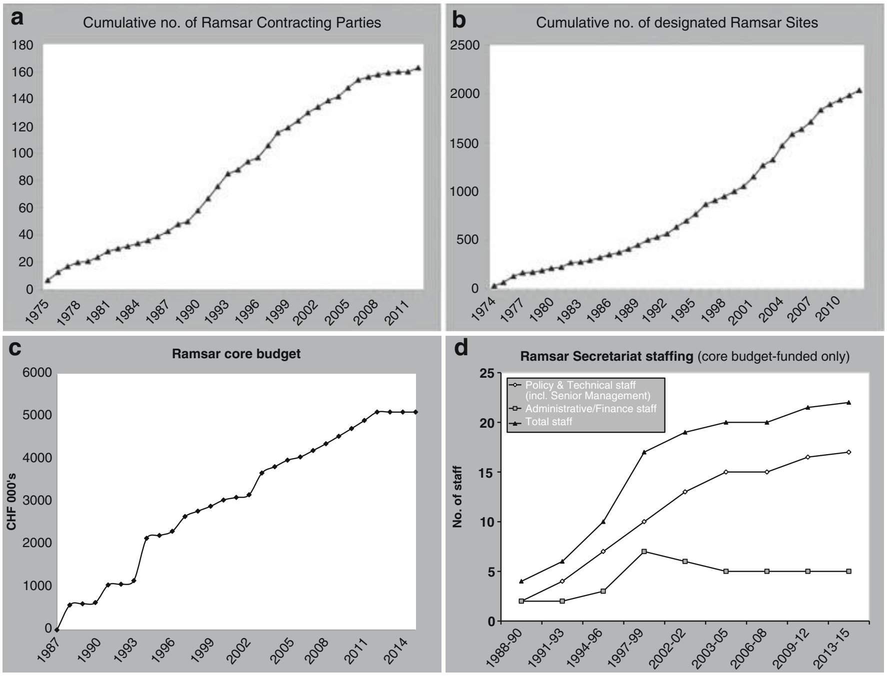

Fig. 1 Growth of the Ramsar Convention since its establishment in 1971: (**a**) number of contracting parties, (**b**) number of designated Wetlands of International Importance (Ramsar sites), (**c**) budget, and (**d**) secretariat staffing

have grown progressively since then, neither budget nor secretariat capacity has kept pace with Convention growth, and the budget has been flat lined (in absolute terms) since 2012 (Fig. 1).

In its early days, the Convention focused attention on the designation of Ramsar sites, particularly for waterbird populations. Since then the Convention has steadily evolved and developed to address its full originally intended scope: essentially all issues of wetlands and people, water, and ecosystems. In so doing, the Convention has recognized that delivering wise use depends on many cross-sectoral collaborations and partnerships and addressing social and cultural, as well as ecological, dimensions of wise use: many of the more recent decisions and guidance adopted by parties concern these approaches (see Tables 1 and 2). In particular the suite of wetlands and water-related guidance (Ramsar Handbooks 8–12) is the only such guidance on managing ecosystems for water and water for ecosystems adopted by governments for global implementation.

The Ramsar Convention has been at the forefront of developing and implementing practical synergies and collaborations between biodiversity-related Conventions and since 1996 has been recognized as the Convention on Biological Diversity's (CBD) lead implementation partner for wetlands. Collaboration is effected the Joint Work Plan (currently the fourth Joint Work Plan, for

Table 1 The Ramsar Convention "tool kit" of handbooks for the conservation and wise use of wetlands, fourth edition, 2010 (Available in English, French, and Spanish on http://www.ramsar.org/search?f[0]=type%3Adocument&f[1]=field_document_type%3A494&search_api_views_fulltext=)

Wise use	
Handbook 1	**Wise use of wetlands**
	Concepts and approaches for the wise use of wetlands
Handbook 2	**National wetland policies**
	Developing and implementing national wetland policies
Handbook 3	**Laws and institutions**
	Reviewing laws and institutions to promote the conservation and wise use of wetlands
Handbook 4	**Avian influenza and wetlands**
	Guidance on control of and responses to highly pathogenic avian influenza
Handbook 5	**Partnerships**
	Key partnerships for implementation of the Ramsar Convention
Handbook 6	**Wetland CEPA**
	The Convention's program on communication, education, participation, and public awareness (CEPA) 2009–2015
Handbook 7	**Participatory skills**
	Establishing and strengthening local communities' and indigenous people's
	participation in the management of wetlands
Handbook 8	**Water-related guidance**
	An integrated framework for the Convention's water-related guidance
Handbook 9	**River basin management**
	Integrating wetland conservation and wise use into river basin management
Handbook 10	**Water allocation and management**
	Guidelines for the allocation and management of water for maintaining the ecological functions of wetlands
Handbook 11	**Managing groundwater**
	Managing groundwater to maintain wetland ecological character
Handbook 12	**Coastal management**
	Wetland issues in integrated coastal zone management
Handbook 13	**Inventory, assessment, and monitoring**
	An integrated framework for wetland inventory, assessment, and monitoring
Handbook 14	**Data and information needs**
	A framework for Ramsar data and information needs
Handbook 15	**Wetland inventory**
	A Ramsar framework for wetland inventory and ecological character description
Handbook 16	**Impact assessment**
	Guidelines on biodiversity-inclusive environmental impact assessment and strategic environmental assessment
Ramsar sites designation and management	
Handbook 17	**Designating Ramsar sites**
	Strategic framework and guidelines for the future development of the List of Wetlands of International Importance

(*continued*)

Table 1 (continued)

Handbook 18	**Managing wetlands**
	Frameworks for managing Ramsar sites and other wetlands
Handbook 19	**Addressing change in wetland ecological character**
	Addressing change in the ecological character of Ramsar sites and other wetlands
International cooperation	
Handbook 20	**International cooperation**
	Guidelines and other supports for international cooperation under the Ramsar
	Convention on wetlands

Table 2 Further or revised scientific and technical implementation advice and guidance adopted by Ramsar COP11 in 2012

Resolution XI.7	Tourism, recreation, and wetlands
Resolution XI.8, annex 1	Ramsar Site Information Sheet (RIS) – 2012 revision[a]
Resolution XI.8, annex 2	Strategic framework and guidelines for the future development of the List of Wetlands of International Importance of the Convention on Wetlands (Ramsar, Iran, 1971) – 2012 revision[a]
Resolution XI.9	An integrated framework for avoiding, mitigating, and compensating for wetland losses
Resolution XI.10	Wetlands and energy issues
Resolution XI.11	Principles for the planning and management of urban and peri-urban wetlands
Resolution XI.12	Wetlands and health: taking an ecosystem approach
Resolution XI.13	An integrated framework for linking wetland conservation and wise use with poverty eradication
Resolution XI.14	Climate change and wetlands: implications for the Ramsar Convention on wetlands
Resolution XI.15	Agriculture-wetland interactions: rice paddy and pest control

[a]This revised RIS format, and its supporting strategic framework and guidelines, formally came into force on 1 January 2015

2011–2020). Thus, implementation of the commitments under the Ramsar Convention is simultaneously implementation of CBD commitments across the range of its thematic and cross-cutting programs of work, including the ecosystem approach, for wetlands. For further description of the progress and challenges of such MEA synergies, see Davidson and Coates (2011).

Convention Implementation Guidance

To support and guide governments and others in Convention implementation, since the early 1990s the Convention has progressively adopted a large suite of "wise use guidelines," most prepared through the work of the Convention's STRP. All such guidance is published thematically in the 20 volumes of the Ramsar Wise Use Handbooks, the most recent being the fourth edition, which includes all guidance adopted up to and including Ramsar's tenth Conference of Contracting Parties in 2008 (Table 1). Further, and revised, guidance was adopted by COP11 in 2012 (Table 2) and is available on http://www.ramsar.org/search?f[0]=type%3Adocument&f[1]=field_tag_body_event%3A366&f[2]=field_tag_body_event%3A415&f[3]=field_document_type%3A530&search_api_views_fulltext=.

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Ramsar Convention: Ramsar Site Designation Process

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David A. Stroud

Contents

Abstract

The designation of Wetlands of International Importance is one of the three "pillars" of the Convention on Wetlands of International Importance (Ramsar Convention). A primary motivation for the establishment of the Convention in the 1970s was the progressive loss of wetlands and negative consequences for biodiversity. One means of addressing this was to establish a mechanism by which states could commit to protect the most important wetlands in their territory (complemented by the wise use of all other wetlands). With sites designated under the World Heritage Convention, it is one of only two legal mechanisms for the global recognition of sites of international importance for biodiversity.

Keywords

Ramsar Convention · Ramsar Sites · Wetlands of International Importance · Designation · Protected areas

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Introduction

The designation of international importance is one of the three "pillars" of the Convention on Wetlands of International Importance (Ramsar Convention). A primary motivation for the establishment of the Convention in the 1970s was the progressive loss of wetlands and negative consequences for biodiversity (Ramsar Convention 1971; Matthews 1993). One means of addressing this was to establish a mechanism by which states could commit to protect the most important wetlands in their territory (complemented by the wise use of all other wetlands). With sites designated under the World Heritage Convention, it is one of only two legal mechanisms for the global recognition of sites of international importance for biodiversity (de Klemm and Shine 1993).

The network extent grows with each designation. Updated statistics are available on the Convention's home page (www.ramsar.org). As at 18 July 2016, there were 2,241 listed Ramsar Sites covering an area of 2,152,476 km^2, slightly greater than the extent of Saudi Arabia (the 12th largest country in the world).

Development of a Global Network of Ramsar Sites: Vision and Objectives

Initially, the strategic direction given to the development of the List of Wetlands of International Importance (or Ramsar Sites) was limited, beyond an obligation to designate a single Ramsar Site as one of the legal requirements of Convention accession (Article 2).

The sixth meeting of the Conference of the Contracting Parties (COP 6), however, urged Parties through the Convention's Strategic Plan 1997–2002 (Ramsar Convention 1996b) to "increase the area of wetland designated for the List of Wetlands of International Importance particularly for wetland types that are under-represented either at the global or national levels."

At the time of COP 7 in 1999, as the number of Ramsar Sites was approaching 1,000, the Convention first adopted the *Strategic Framework and guidelines for the future development of the List of Wetlands of International Importance* (Ramsar Convention 1999) and has amended and added to it regularly since then, most substantially at COP 11 (Ramsar Convention 2012b). Its purpose is to provide a clearer view, or vision, of the long-term outcomes which the Convention is seeking to achieve through the Ramsar List. Advice is also offered to assist Contracting Parties (CPs) take a systematic approach to identifying priorities for future designations, in order to create national networks of Ramsar Sites which, when considered at the global level, fulfill the vision for the Ramsar List, which is:

> To develop and maintain an international network of wetlands which are important for the conservation of global biological diversity and for sustaining human life through the maintenance

of their ecosystem components, processes and benefits/services. (In this context, the Convention defined “ecosystem benefits” in accordance with the Millennium Ecosystem Assessment definition of ecosystem services as “the benefits that people receive from ecosystems.”)

Such an international network of wetland sites has to be built from coherent and comprehensive national networks of Ramsar Sites. The extent to which this has been achieved is highly variable. Several countries have a large area (Bolivia, Canada, Chad, Congo, and Russian Federation all with >10,000,000 ha) and/or numbers of sites (UK [170] and Mexico [142]), others few (26 Contracting Parties just have a single Ramsar Site as at 22 September 2012).

The distribution of sites globally is highly uneven, with most designations having been made in Europe (Figs. 1 and 2) and fewest in Oceania. To a significant extent, this reflects patterns of accessions to the Convention – the earliest Contracting

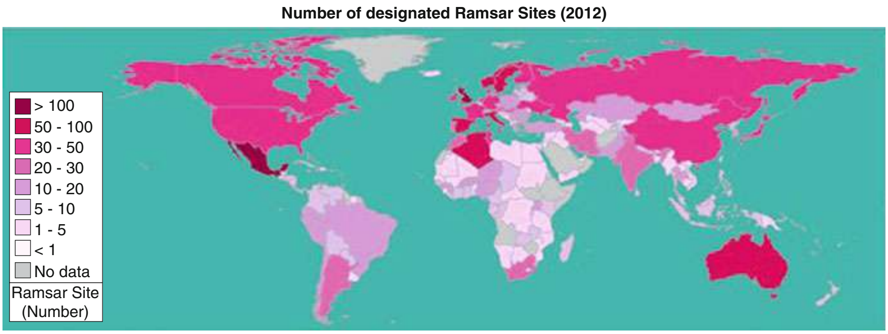

Fig. 1 Distribution of numbers of Ramsar Sites by Contracting Party as at 5 November 2012 (Data from www.ramsar.wetlands.org)

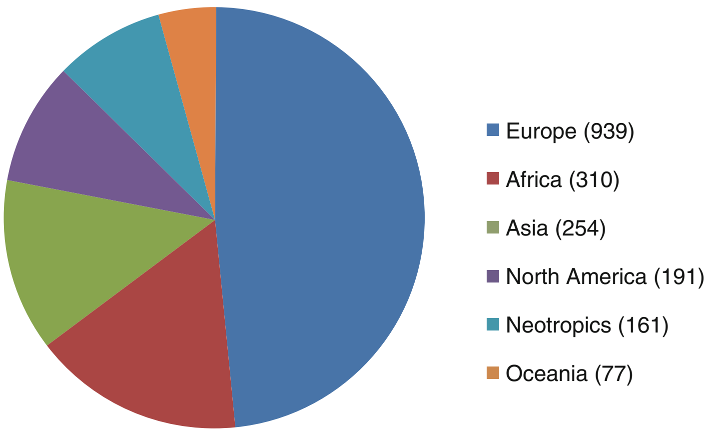

Fig. 2 Number of Ramsar Sites by region; data as at November 2011 (Data from www.ramsar.wetlands.org)

Parties joining the Convention were largely from Europe, with most recent accessions being from Oceania.

The Convention established (Ramsar Convention 2012b) the following five objectives to realize the vision for the Ramsar List (not in priority order):

1. To establish national networks of Ramsar Sites in each Contracting Party which fully represent the diversity of wetlands and their key ecological and hydrological functions;
2. To contribute to maintaining global biological diversity through the designation and management of appropriate wetland sites;
3. To foster cooperation among Contracting Parties, the Convention's International Organization Partners, and local stakeholders in the selection, designation, and management of Ramsar Sites;
4. To use the Ramsar Site network as a tool to promote national, supranational/regional, and international cooperation in relation to complementary environment treaties; and
5. To use national Ramsar Site networks to provide essential ecosystem services/ benefits, especially related to water, that contribute to human health, livelihoods, and well-being.

Designation and Listing

The designation of (at least one) wetland of international importance is one of the central obligations of CPs. Designation of "at least one wetland" (Article 2.4) is part of the accession process when states join the Convention. Thereafter, "each Contracting Party shall designate suitable wetlands within its territory for inclusion in the List of Wetlands of International Importance" (Article 2.1). It is this requirement to designate geographical territory on accession which constrains non-state parties (such as Regional Economic Integration Organisations for example the European Union) from becoming signatories to the Convention.

The process of "designation" of a Ramsar Site by the state party – in which it assumes legal responsibility for the wise use of the site – is a process distinct from the addition of that site to the Convention's List of Wetlands of International Importance. That formal "listing" by the Secretariat requires acceptance of a sufficient quality map and a supporting information sheet (RIS) submitted by the Contracting Party (see Sect. 9 of Ramsar Convention 2012b).

Typically, designation and listing occur together, but there are some sites designated by Parties which remain to be listed.

Ramsar Site Identification and Criteria

Throughout its evolution, the Convention has developed criteria for the designation of Ramsar Sites which have been kept under constant review (Matthews 1993; Stroud 2013). It has supplemented these with regularly updated guidelines to assist

Contracting Parties (CPs) in their interpretation and application of the criteria reflecting the development of conservation science (Ramsar Convention 2012b).

Essentially, wetlands may be defined as of international importance because they hold representative, rare, or unique wetland types (Criterion 1), rare ecological communities or species including at critical life cycle stages (Criteria 2–4), or important numbers of birds (Criteria 5 and 6), fish (Criteria 7 and 8), or other animal taxa (Criterion 9) (Box 1).

Ramsar Site Description

The organization and reporting of information about Ramsar Sites lie at the heart of the Convention's activities. This key need was specifically highlighted in the Final Act of the 1971 Conference in Ramsar, Iran, which stated that "the entries in the List of Wetlands of International Importance which is to be maintained under the Convention could usefully be supplemented by descriptions of the biotopes involved and an enumeration of the bird species especially in need of protection therein." (Carp 1972). The emphasis given to this information requirement at that time, before any consideration had been given to other processes supporting the Convention, was significant. It was not until COP 4, however, that a RIS was formally established as a means of collating this information (Ramsar Convention 1990), and its format has been progressively revised at subsequent COPs at a frequency that balanced the need for stability in established processes with both the development of the Convention's growing needs for better strategic overviews of the List of Ramsar Sites and the technological abilities to disseminate this key information to the wide spectrum of users.

The RIS is used to describe the site at the time of designation in sufficient detail to provide a baseline for subsequent monitoring to detect any changes to these ecological and hydrological attributes. Parties are requested to update the information in the RIS, either when there is an actual or potential change in ecological character or at periods not longer than 6 years (Ramsar Convention 1996a).

In 2012, COP 11 agreed a major restructuring of the format of the RIS and a move to develop systems that will allow for its electronic submission (Ramsar Convention 2012a). Aspects of the 2012 RIS format such as its clearer focus on species information, and – for the first time – the collection of simple information on ecosystem services provided by sites, have been acknowledged as potentially revolutionizing our understanding of both individual Ramsar Sites and the global network.

Obligations Following from Ramsar Designation

The act of designating (listing) a wetland as internationally important under the Convention is a first step along a conservation and sustainable use pathway, the endpoint of which is achieving the long-term wise (sustainable) use of the site.

Article 3.2 of the Convention determines that "each Contracting Party shall arrange to be informed at the earliest possible time if the ecological character of any wetland in its territory and included in the List has changed, is changing or is likely to change." Accordingly, the Ramsar Convention has developed the concept of "ecological character" for wetlands, which is defined (Ramsar Convention 2005a) as:

> Ecological character is the combination of the ecosystem components, processes and benefits/services that characterise the wetland at a given point in time.

Parties are expected to manage their Ramsar Sites in such a way as to maintain the ecological character of each site and, in so doing, retain those essential ecological and hydrological functions which ultimately provide its ecosystem services. Ecological character is therefore an indication of the "health" of the wetland, and changes to ecological character outside its natural range of variation may signal that uses of a site, or externally derived impacts on the site, are unsustainable and may lead to the degradation of natural processes and thus the ultimate breakdown of the ecological, biological, and hydrological functioning of the wetland.

Delisting

Guidance on the issue of the removal of a Ramsar Site from the international list has been provided by the COP as related to several scenarios.

The first relates to the provisions established under Article 2.5 which allow a party "...because of its urgent national interests, to delete or restrict the boundaries of wetlands already included by it in the List...." Typically, such provisions have been exercised where development or other activities in the national interest unavoidably conflict with the status of a Ramsar Site. Where such deletion or boundary restriction occurs, Article 4.2 requires that the Contracting Party "...should as far as possible compensate for any loss of wetland resources." Resolution VIII.20 (Ramsar Convention 2002a) gave general guidance for interpreting "urgent national interests" under Article 2.5 of the Convention and in considering compensation under Article 4.2.

Resolution VIII.22 (Ramsar Convention 2002b), however, recognized situations other than Article 2.5 provisions in which Ramsar Site boundaries may need refinement notably where:

(a) A Ramsar Site never met the criteria for designation
(b) Part or all of a Ramsar Site unavoidably loses the values, functions, and attributes for which it was included or was designated in error
(c) A Ramsar Site at the time of listing met the criteria, but, while its interests remain unchanged, it later fails to meet the criteria because of a change in those criteria or in the population estimates or parameters which underpin them

Resolution IX.6 (Ramsar Convention 2005b) provided guidance on principles and procedures for situations not foreseen in the Convention text concerning the loss or deterioration of the ecological character of listed wetlands under circumstances other than those addressed by Article 2.5.

Different National Implementation Concepts

One of the strengths of the Convention is its flexibility to a wide range of differing national circumstances – particularly manifest in the range of different models of national implementation of the Ramsar Site concept. At one extreme, some countries, for example, UK, tightly define boundaries of sites which are given a high level of protection (albeit for multifunctional uses) under national legislation and where legal protection regimes are established prior to the designation.

A contrasting approach, equally valid under the Convention, uses the designation to give status to landscape or catchment scale wetlands which may have no prior national legislative protection. In these cases, the Ramsar status is used to help resolve conservation or land-use issues as they arise. Such wide-scale Ramsar Sites may also contain other, more restricted protected sites with a higher level of national protection. For example, France has designated a number of large-scale Ramsar Sites containing more strictly protected and discrete Natura 2000 sites.

Sources of Further Information

A wealth of information on Ramsar Sites and their designation is available on the Convention's website. Considerable background information is given by Ramsar Convention Secretariat 2010 and Ramsar Convention 2012b. The Ramsar Sites Information Service (www.ramsar.wetlands.org) is the formal database of information on Ramsar Sites maintained by Wetlands International on behalf of the Convention and gives access to detailed information on listed sites and a wide range of summary statistics.

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Ramsar Convention: Transboundary Ramsar Sites

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Royal C. Gardner

Contents

Abstract

A transboundary Ramsar Site refers to "an ecologically coherent wetland [that] extends across national borders" when "the Ramsar Site authorities on both or all sides of the border have formally agreed to collaborate in its management, and have notified the [Ramsar] Secretariat of this intent." Such a cooperative management arrangement is one way that contracting parties to the Ramsar Convention may implement their duty of international cooperation, including the responsibilities related to Article 5 of the Convention, which requires contracting parties to consult with one another in the case of shared wetlands or water systems. A site's designation as a transboundary Ramsar Site acknowledges the existence of a cooperative management arrangement, but it is not a distinct legal status for the Ramsar Sites involved. Accordingly, such a designation does not create additional international legal obligations beyond those already imposed by the Ramsar Convention.

Keywords

Ramsar convention · Ramsar sites · Transboundary

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Introduction

A transboundary Ramsar Site refers to "an ecologically coherent wetland [that] extends across national borders" when "the Ramsar Site authorities on both or all sides of the border have formally agreed to collaborate in its management, and have notified the [Ramsar] Secretariat of this intent" (Ramsar Convention Secretariat 2006). Such a cooperative management arrangement is one way that Contracting Parties to the Ramsar Convention may implement their duty of international cooperation, including the responsibilities related to Article 5 of the Convention which requires Contracting Parties to consult with one another in the case of shared wetlands or water systems. A site's designation as a transboundary Ramsar Site acknowledges the existence of a cooperative management arrangement, but it is not a distinct legal status for the Ramsar Sites involved. Accordingly, such a designation does not create additional international legal obligations beyond those already imposed by the Ramsar Convention.

Transboundary Ramsar Sites

The first formal transboundary Ramsar Site occurred when Hungary and Slovakia notified the Ramsar Secretariat in 2001 of their intention to manage Hungary's Baradla cave system and its related wetlands and Slovakia's Domica in a cooperative manner. The Contracting Parties have flexibility in how to manage such sites; there is no prescribed management approach. As Table 1 indicates, as of September 2016 there were 18 formal transboundary Ramsar Sites. All these transboundary Ramsar Sites are located in Europe, except for Niumi-Saloum, which extends across the borders of Gambia and Senegal. The Ramsar Secretariat maintains a current list of transboundary Ramsar Sites on its website.

Table 1 Transboundary Ramsar Sites: collaborative international management of adjacent Ramsar Sites (Source: Ramsar Convention Secretariat 2014)

Contracting Parties	Individual Ramsar Sites and designation dates	Common TRS name (if any)	Instrument
Hungary	Baradla cave system and related wetlands (2001)	–	14 Aug. 2001
Slovakia	Domica (2001)		18 Jan. 2001
Hungary	Felsö-Tisza (Upper Tisza) (2004)	Upper Tisza valley	6 Nov. 2003
Slovakia	Tisa river (2004)		
Belgium	Vallée de la Haute-Sûre (2003)	Vallée de la Haute-Sûre	8 March 2004
Luxembourg	Vallée de la Haute-Sûre (2003)		

(*continued*)

Table 1 (continued)

Contracting Parties	Individual Ramsar Sites and designation dates	Common TRS name (if any)	Instrument
Austria	Donau-March-Thaya-Auen (1982)	Trilateral Ramsar Site Floodplains of the Morava-Dyje-Danube Confluence	30 June 2004
Czech Republic	Untere Lobau (1982)		
Slovakia	Mokrady dolního Podyjí (floodplain of lower Dyje river) (1993)		
	Moravské luhy (Morava floodplains) (1993)		
Estonia	Nigula Nature Reserve (1997)	North Livonian Transboundary Ramsar Site	27 Dec. 2007
Latvia	Sookuninga Nature Reserve (2006)		31 July 2006
	Northern bogs (Ziemelu Purvi) (2002)		
Hungary	Ipoly valley (2001)		2 Feb. 2007
Slovakia	Poiplie (1998)		
Belarus	Prostyr (2005)	Stokhid-Prypiat-Prostyr	4 Jan. 2008
Ukraine	Prypiat river floodplains (1998)		1 Feb. 2008
	Stokhid river floodplains (1995)		
Austria	Bayerische Wildalm and Wildalmfilz (2004)	Austrian-Bavarian Wildalm	7 Aug. 2008
Germany	Bayerische Wildalm (2007)		
France	Rhin supérieur/Oberrhein (2008)	Rhin supérieur/Oberrhein – Oberrhein/Rhin supérieur	28 Aug 2008
Germany	Oberrhein/Rhin supérieur (2008)	Niumi-Saloum	26 June 2008
Gambia	Niumi National Park (2008)		
Senegal	Delta du Saloum (1984)		21 Oct. 2008
Czech Republic	Krkonoská rašeliniště (Krkonose mountain mires) (2003)	Krkonose/Karkonosze subalpine peat bogs	17 Sep. 2009
			14 Sep. 2009
Poland	Subalpine peat bogs in Karkonosze mountains (2002)		
Austria	Neusiedler See-Seewinkel (1982)	Transboundary Ramsar Site Neusiedler See-Seewinkel – Fertö-Hanság ["Fertö-Hanság határon átnyúló ramsari területe," "Grenzüberschreitendes Ramsar-Gebiet Neusiedlersee-Seewinkel-Waasen"]	12 Nov. 2009
Hungary	Lake Fertö (1989)		
	Nyirkai-Hany (2006)		

(*continued*)

Table 1 (continued)

Contracting Parties	Individual Ramsar Sites and designation dates	Common TRS name (if any)	Instrument
Belarus	Kotra (2002)	–	20 July 2010
Lithuania	Cepkeliai (1993)		2 July 2009
Bulgaria	Srebarna (1975)	Lake Calarasi (Iezerul Calarasi) – Srebarna	15 April 2013
Romania	Lake Calarasi (Iezerul Calarasi) (2012)		
Bulgaria	Belene Islands Complex (2002)	Suhaia – Belene Islands Complex	15 April 2013
Romania	Suhaia (2012)		
Bulgaria	Ibisha island (2002)	Bistret – Ibisha island	15 April 2013
Romania	Bistret (2012)		
Denmark	*Vadehavet (Wadden Sea)* (Site no. 356, 1987)		
Germany	Hamburgisches Wattenmeer (Site no.501, 1990) Schleswig-Holstein Wadden Sea and adjacent areas (Site no. 537, 1991) Wattenmeer, Elbe-Weser-Dreieck (Site no. 80, 1976) Wattenmeer, Jadebusen & westlich Wesermündung (Site no. 81, 1976) Wattenmeer, Ostfriesisches Wattenmeer & Dollart (Site no. 82, 1976)	Wadden Sea	01 December 2015
The Netherlands	North Sea Coastal Area (Noordzeekustzone) (Site no. 1252, 2000) Waddenzee (Wadden Sea) (Site no. 289, 1984, now includes former Sites 195 Boschplaat and 196 Griend) Duinen en Lage land Texel (Site no. 2213, 2000) Duinen Vlieland (Site no. 2216, 2000) Duinen Terschelling (Site no. 2215, 2000) Duinen Ameland (Site no. 2212, 2000) Duinen Schiermonnikoog (Site no. 2214, 2000)		
Belarus	Olmany Mires Zakaznik (Site no. 1091, 2001)	Olmany - Perebrody mires	25 June 2014
Ukraine	Perebrody Peatlands (Site no. 1402, 2004)		29 December 2015

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54 Ecological Character Concept of the Ramsar Convention

Dave Pritchard

Contents

Abstract

The Ramsar Convention requires every Contracting Party to arrange to be informed about change or potential change in the ecological character of its listed Ramsar Sites. Definitions of "ecological character" and "change" were agreed in 1996, then revised in 1999 and 2005. These definitions emphasise the interrelationships between all the processes and attributes that characterise a given wetland, and the latest version includes ecosystem services provided by the wetland as part of the character. Maintenance of ecological character is now the conceptual basis for obligations under the Convention applying to all wetlands, whether listed or not; and extensive guidance and commentaries have been published to support implementation of this.

Keywords

Ecological character · Obligation · Maintenance · Adverse change · Ecosystem services · Definitions

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Introduction

The Ramsar Convention text, as adopted in 1971 (Matthews 1993), introduced the concept of the "ecological character" of wetlands in its Article 3.2, which required every Contracting Party government to "...arrange to be informed at the earliest possible time if the ecological character of any wetland in its territory and included in the List has changed, is changing or is likely to change as the result of technological developments, pollution or other human interference." ("The List" refers to the List of Wetlands of International Importance, usually known as Ramsar sites, created under the Convention.)

Measures to implement this requirement were elaborated by several of the early meetings of the Conference of Parties (COP), but it was not until the sixth meeting, in 1996, that "ecological character" was formally defined for the first time (it has been amended twice since then). COP6 Resolution VI.1 gave a "working definition, to be assessed further," as follows: "The 'ecological character' is the structure and inter-relationships between the biological, chemical, and physical components of the wetland. These derive from the interactions of individual processes, functions, attributes and values of the ecosystem(s)" (Ramsar Convention 1996).

"Change in ecological character" of a wetland was defined at the same time (again provisionally) as "the impairment or imbalance in any of those processes and functions which maintain the wetland and its products, attributes and values."

The Resolution and its annex added an explanation of how these definitions were to be interpreted. First, it emphasized that the working definitions were "relevant to the management of wetlands in general" (i.e., not only to Ramsar sites). It further noted that "change in the ecological character of a site is interpreted as meaning adverse change, in line with the context of Article 3.2 of the Convention [...]. The definition refers explicitly to adverse change caused by human activities. It excludes the process of natural evolutionary change occurring in wetlands. It is also recognised that wetland restoration and/or rehabilitation programmes may lead to favourable human-induced changes in ecological character."

Key terms used in the definitions ("processes," "functions," "values," "products," and "attributes") were themselves defined in the same decision, but in ways which have since been superseded by the definitions quoted below. The annex to the Resolution also provided "guidelines for describing and maintaining ecological character of listed sites."

Reworking of the Concept

Following deliberations by the Ramsar Scientific and Technical Review Panel (STRP), a first reworking of the definitions was agreed by the Parties at COP7 in 1999 (Ramsar Convention 1999). Resolution VII.10 adopted the following: "ecological character is the sum of the biological, physical, and chemical components of the wetland ecosystem, and their interactions, which maintain the wetland and its products, functions, and attributes"; and "change in ecological character is the impairment or imbalance in any biological, physical, or chemical components of the wetland ecosystem, or in their interactions, which maintain the wetland and its products, functions and attributes."

The STRP was then asked by COP8 in 2002 to "further review and, as appropriate, develop guidance and report to COP9 concerning identified gaps and disharmonies in defining and reporting the ecological character of wetlands, including, inter alia, harmonisation of definitions and terms in the guidance on inventory, assessment, monitoring and management of the ecological character of wetlands." Following this work, and in particular some conceptual alignment between the Convention and the Millennium Ecosystem Assessment (MA), COP9 in 2005 agreed the definitions that remain in use today (Resolution IX.1 Annex A: Ramsar Convention 2005).

In these, "**ecological character**" was defined as "**the combination of the ecosystem components, processes and benefits/services that characterise the wetland at a given point in time**." The definition is to be read with the accompanying notes which explain that "within this context, ecosystem benefits are defined in accordance with the MA definition of ecosystem services as 'the benefits that people receive from ecosystems'"; and "the phrase 'at a given point in time' refers to Resolution VI.1 paragraph 2.1, which states that 'it is essential that the ecological character of a site be described by the Contracting Party concerned at the time of designation for the Ramsar List, by completion of the Information Sheet on Ramsar Wetlands (as adopted by Recommendation IV.7)'."

"**Change in ecological character**" was defined as follows: "**For the purposes of implementation of Article 3.2, change in ecological character is the human-induced adverse alteration of any ecosystem component, process, and/or ecosystem benefit/service.**" The accompanying notes explain that "the inclusion of specific reference to Article 3.2 of the Convention text within the definition is designed to clarify the maintenance obligation for the ecological character of listed Wetlands of International Importance (Ramsar sites) under Article 3.2, and to note that such change concerns only adverse change caused by the actions of people. [. . .] For the purposes under the Convention, this definition therefore excludes the processes of natural evolutionary change occurring in wetlands and also excludes positive human-induced change. However, it should be noted that other actions adopted by the Convention, such as those concerning assessing the overall status and trends of wetlands and Ramsar sites, require information on all types of change in ecological character – positive and negative, natural and human-induced".

Technical Guidance

Subsequently, extensive guidance has been adopted on the operation of these concepts (Ramsar Convention Secretariat 2013). This includes guidance on the systematic description of the ecological character of individual wetlands as a baseline against which to assess change (annex to Resolution X.15) (Ramsar Convention 2008). This highlights, among other things, the fact that the concept of ecological character covers more than merely a list of the ecosystem's components, processes, and services, requiring also attention to the additional idea of what they represent *in combination*, since it is "a holistic rather than a reductionist concept." The annex to Resolution X.16 provides a "Framework for processes of detecting, reporting and responding to change in wetland ecological character," and further detail on these aspects is set out in a background paper (DOC.27) provided to COP10 in 2008 and in volume 19 of the *Ramsar Wise Use Handbooks* (4th edition, 2010).

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Wise Use Concept of the Ramsar Convention

55

Dave Pritchard

Contents

Abstract

The term "wise use", as it relates to wetlands, originates in the text of the Ramsar Convention adopted in 1971. Initially the term was not defined; but early discussions made clear that it expressed an aim to combine conservation and sustainable use of resources, making Ramsar the first Convention to embrace what later became the widely accepted concept of "sustainable development". A first formal definition of "wise use" was adopted in 1987 by the Convention's Parties, who then modified it in 2005. This later definition emphasises "maintenance of wetland ecological character" as the central tenet of the concept. A quantity of guidance material has been published to support the implementation of wise use for all wetlands.

Keywords

Ramsar convention · Wise use · Sustainable development · Ecosystem approaches · Definitions

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Introduction

The term "wise use," as it relates to wetlands, originates in the text of the Ramsar Convention adopted in 1971. Article 3.1 of the Convention requires Contracting Party governments to "... formulate and implement their planning so as to promote ... as far as possible the wise use of wetlands in their territory." It is thus an obligation that must be implemented through national-level measures; and it applies to all wetlands, whether specially protected or not.

Subsequent definitions and commentaries have concentrated on this provision for wise use of wetland ecosystems, largely ignoring the fact that the Convention also applies it to species populations ("wise use of migratory stocks of waterfowl," Article 2.6).

Developing the Concept of Wise Use

Initially the term was not defined; but early discussions made clear that it was intended to cover maintenance of the "ecological character" of wetlands, as a basis not only for nature conservation but also for sustainable development (Davis 1992). This made Ramsar the first Convention to seek to combine conservation and sustainable use of resources.

This thinking eventually became reflected in the first formal definition of wise use of wetlands adopted by the Conference of Parties at their third meeting (COP3) in 1987 (Davis 1993), when it was defined as "their sustainable utilization for the benefit of humankind in a way compatible with the maintenance of the natural properties of the ecosystem."

"Sustainable utilization" was defined at the same time as "human use of a wetland so that it may yield the greatest continuous benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations." This echoed the definition of "sustainable development" put forward in the same year by the UN World Commission on Environment and Development (Brundtland Commission) as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs."

Mission statements adopted by the Ramsar Convention have reinforced this connection, by emphasizing that conservation and wise use of all wetlands under the Convention is intended to be "a contribution towards achieving sustainable development throughout the world."

These definitions all tread delicately around the accommodation of both "constant state" and "progressive development" ideas of "use."

The terminology continued to evolve thereafter, in contexts such as the 1992 Convention on Biological Diversity's approach to "sustainable use" and the development of the concept of "ecosystem services" by the Millennium Ecosystem Assessment, completed in 2005. Ramsar Parties asked the Convention's Scientific and Technical Review Panel to "review the wise use concept, its applicability, and its consistency with the objectives of sustainable development" in light of these developments.

Based on the Panel's findings, COP9 in 2005 adopted Resolution IX.1, Annex A of which included an updated definition of wise use (Ramsar Convention 2005; Finlayson et al. 2011), as follows: "Wise use of wetlands is the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development."

The first clarifies that "ecosystem approaches" include, among others, those elaborated by the Convention on Biological Diversity and by the Helsinki and OSPAR Commissions. This highlights factors such as integrated management, stakeholders' participation in decision-making, transparency about trade-offs, and equitability of outcomes. Mechanisms applied under Ramsar such as Integrated River Basin Management, Integrated Water Resources Management, Integrated Coastal Zone Management, and Integrated Marine and Coastal Area Management had, in effect, already been exemplifying this approach. (Indeed, wise use is regarded as the longest-established example among intergovernmental processes of the implementation of an ecosystem-based approach to the conservation and sustainable development of natural resources.)

The second footnote explains that the phrase "in the context of sustainable development" is intended to recognize that "whilst some wetland development is inevitable and that many developments have important benefits to society, developments can be facilitated in sustainable ways by approaches elaborated under the Convention, and it is not appropriate to imply that 'development' is an objective for every wetland."

The successive definitions of "wise use" have brought it to equate both to "sustainable use" and to the "maintenance of the ecological character" of all wetlands (Finlayson et al. 2011). That latter phrase originated with the Convention's provisions for conservation of special sites, but since it in turn has been redefined in a way that now incorporates "ecosystem services" in the sense of the Millennium Ecosystem Assessment, a fusion of two historically distinct ideas has been achieved. While "services" had previously been seen as *flowing from* "character," they are now seen as an integral part of it. While "use" and "conservation" had previously been seen as different things, to be reconciled by a "balancing" notion of sustainability, wise use is now primarily about *maintaining the capability for use*, rather than *using, subject to a sustainability caveat*.

COP4 in 1990 adopted initial guidelines for the implementation of wise use, followed by the adoption of additional guidance at COP5 in 1993 (Ramsar Convention 1990, 1993). These were groundbreaking in their time, but have since largely been superseded by an increasingly comprehensive suite of other guidance on specific aspects of Ramsar implementation. Nonetheless, the landmark adoption of a "conceptual framework for the wise use of wetlands and the maintenance of their ecological character" in 1990 and the production of successive editions of 21 volumes of the Convention's "toolkit" of "Handbooks for the wise use of wetlands" testify to the continuing centrality of this concept in global wetland conservation (Ramsar Convention Secretariat 2010).

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Convention of Migratory Species (CMS) and Wetland Management

56

Robert J. McInnes and Nick C. Davidson

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Abstract

Migratory species, including waterbirds, cetaceans and turtles, face many challenges as they cross national borders. The Convention on the Conservation of Migratory Species of Wild Animals – *or as it is more commonly referred to as the Convention on Migratory Species* (*CMS*) *or the Bonn Convention* – is an environmental treaty that provides a global platform for the conservation and sustainable use of migratory animals and their habitats. The Convention was signed in 1979 entered into force in 1983. Since this date, the Convention's membership has grown steadily, and in 2015 it includes over 100 parties from Africa, Central and South America, Asia, Europe, and Oceania. CMS is the only global convention specializing in the conservation of migratory species, their habitats, and migration routes.

Keywords

Waterbirds · International cooperation · Transboundary management · Conservation

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Origins of the Convention

Migratory species, including waterbirds, cetaceans and turtles, face many challenges as they cross national borders (Kirby et al. 2008). The Convention on the Conservation of Migratory Species of Wild Animals – *or as it is more commonly referred to as the Convention on Migratory Species* (*CMS*) *or the Bonn Convention* (http://www.cms.int/) – is an environmental treaty that provides a global platform for the conservation and sustainable use of migratory animals and their habitats. The Convention was signed in 1979 in Bad Godesberg, a suburb of Bonn (hence the name), and entered into force in 1983 (Lyster 1989). Since this date, the Convention's membership has grown steadily, and in 2015 it includes over 100 parties from Africa, Central and South America, Asia, Europe, and Oceania. CMS is the only global convention specializing in the conservation of migratory species, their habitats, and migration routes. However, CMS complements and cooperates with a number of other international organizations, including the Ramsar Convention, nongovernmental organizations, and the private sector.

CMS brings together the States through which migratory animals pass (termed "range states") and establishes the legal foundation for internationally coordinated conservation measures throughout a migratory range. The CMS mandate covers all taxa of species which are regarded as migratory, under a geopolitical definition of "migratory" of species which cross national boundaries on a regular and predictable basis: "Migratory species" means the entire population or any geographically separate part of the population of any species or lower taxon of wild animals, a significant proportion of whose members cyclically and predictably cross one or more national jurisdictional boundaries" (CMS text Article 1.1 a).

Migratory species threatened with extinction are listed on Appendix I of the Convention. CMS parties work closely toward strictly protecting these animals; managing, conserving, or restoring the habitats where they live; understanding and working to remove obstacles to migration; and controlling other factors that might endanger them. Besides establishing obligations for each State joining the Convention, CMS promotes concerted action among the range states of many of these species. CMS and its daughter agreements determine policy and provide further guidance on specific issues, such as avian influenza or minimizing the risk of poisoning to migratory birds, through various strategic plans, action plans, resolutions, decisions, and guidelines.

International Cooperation and Protection

Many migratory species depend on trans-boundary cooperation for their protection. The migratory species that require or would significantly benefit from international cooperation are listed in Appendix II of the Convention. In order to successfully manage wetlands that are utilized by trans-boundary migratory species, the Convention encourages the range states to conclude global or regional agreements. In this respect, CMS acts as a framework Convention. The agreements may range from legally binding treaties (called agreements) to less formal instruments, such as memoranda of understanding, and can be adapted to the requirements of particular regions. Consequently, CMS influences the management and wise use of migratory wetland-dependent species of fish, birds, and mammals through a number of these established agreements and memoranda of understanding. These include:

Agreements
- Wadden Sea Seal Agreement
- Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea, and Contiguous Atlantic Area (ACCOBAMS)
- Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS)
- African-Eurasian Migratory Waterbird Agreement (AEWA)
- Agreement on the Conservation of Albatrosses and Petrels (ACAP)
- Agreement on the Conservation of Gorillas and Their Habitats

Memoranda of Understanding
- Conservation Measures for the Aquatic Warbler (*Acrocephalus paludicola*)
- Conservation Measures for the Siberian Crane
- Conservation and Management of Marine Turtles and Their Habitats of the Indian Ocean and South-East Asia (IOSEA)
- Conservation Measures for Marine Turtles of the Atlantic Coast of Africa
- Conservation of High Andean Flamingos and Their Habitats
- Conservation Measures for the Eastern Atlantic Populations of the Mediterranean Monk Seal (*Monachus monachus*)
- Conservation of Cetaceans and Their Habitats in the Pacific Islands Region
- Conservation Measures for the Slender-Billed Curlew
- Conservation of the Manatee and Small Cetaceans of Western Africa and Macaronesia
- Conservation Measures for the Ruddy-Headed Goose (*Chloephaga rubidiceps*) (between the Argentine Republic and the Republic of Chile)
- Conservation of Migratory Sharks
- Conservation Measures for the West African Populations of the African Elephant (*Loxodonta africana*)

Further information on each of these agreements and memoranda of understanding can be found on the CMS Family Portal on: http://www.migratoryspecies.org/.

One of the most well-established examples of cooperation through an agreement is the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA) (http://www.unep-aewa.org/). Developed under the framework of CMS, and administered by the United Nations Environment Programme (UNEP), AEWA brings together countries and the wider international conservation community, including the Ramsar Convention, in an effort to establish coordinated conservation and management of migratory waterbirds throughout their entire migratory range (Boere and Lenten 1998). Covering migratory waterbirds and their wetland habitats across Africa, Europe, the Middle East, Central Asia, Greenland, and the Canadian Archipelago, all the species covered by AEWA are "wetland dependent" and so are also covered by the provisions of the Ramsar Convention, and Ramsar's established mechanisms such as for international waterbird flyway cooperation, and the designation and management of Ramsar Sites for waterbirds.

The high relevance of the Ramsar Convention mechanisms to AEWA and the need for collaborative implementation of the two instruments were most recently recognized by the 5th Meeting of Parties to AEWA through its adoption of two decisions (AEWA 2012a, b).

Future Challenges

Migratory wetland-dependent species are a major component of faunal biodiversity. Not only do these species represent a significant proportion of the world's genetic variety, they also play an essential role in ecosystem functioning and dynamics through, for instance, the pollination of plants or the dispersal of seeds.

In its new strategic plan for migratory species (CMS 2014), the Convention identified a range of issues and future challenges for the conservation of migratory species, as follows.

Migratory species have their own special vulnerabilities. Migratory journeys expose them to heightened survival risks, and habitat requirements are often a complex mix of different components in breeding areas, nonbreeding areas, and the places in between. Concentrations of large numbers of individuals during specific periods at specific sites also increase the risk of serious impacts from negative pressures at those sites.

Barriers to migration pose special challenges, whether or not in the form of physical obstacles, which may cause direct mortality, or fragmentation of ecological resources disrupting movement from one place to another.

Conservation strategies therefore need to give holistic attention not only to populations, species, and habitats, but to the entire span of migratory routes and the functioning of the migration process. Many of the actions defined in this plan are accordingly directed toward "migratory systems," a concept which reflects the interdependent complexes of places, routes between places, populations, ecological factors, and temporal cycles involved.

The repeating cycles and trans-boundary ranges inherent to the phenomenon of migration are fundamental to the ability of the planet to support humankind and biodiversity overall. Migration is a key adaptation to natural rhythms and evolutionary changes; and by the same token both migratory species and their habitats can be affected/disrupted by human impacts, including climate change.

A great many migratory species are of major direct and indirect importance for people's food security and livelihoods. Many human communities rely on the regular influx of migratory animals: as a basis for subsistence; for economically and/or culturally important hunting, fishing, tourism, and recreation; or to maintain ecosystem function in a way that allows another resource to be harvested. Levels of use by one community can significantly affect the availability of the resource to communities in different, possibly distant, locations.

The conservation and sustainable use of migratory species is therefore a key contribution to wider aims of sustainable development and requires global attention.

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57 Convention on Biological Diversity (CBD) and Wetland Management

David Coates

Contents

Abstract
The Convention on Biological Diversity (CBD) was inspired by the world's growing commitment to sustainable development. Its objectives are the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of benefits arising from the use of genetic resources. The CBD arose from the United Nations Conference on Environment and Development. The convention currently has 193 parties, and, excepting the United States, all parties of the Ramsar Convention are also parties to the CBD. This section outlines the approach to wetlands under the CBD, development of collaboration between the Ramsar Convention on Wetlands and the CBD, and constraints to, and opportunities for further, incorporation of wetlands management under the CBD.

Keywords
Biological diversity · Conservation · UNCED · Ramsar Convention · Collaboration

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Introduction

The Convention on Biological Diversity (CBD) was inspired by the world's growing commitment to sustainable development. Its objectives are the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of benefits arising from the use of genetic resources (www.cbd.int). The CBD arose from the United Nations Conference on Environment and Development (the Rio "Earth Summit," 1992). The convention currently has 193 parties, and, excepting the United States, all parties of the Ramsar Convention are also parties to the CBD. This section outlines the approach to wetlands under the CBD, development of collaboration between the Ramsar Convention on Wetlands and the CBD, and constraints to, and opportunities for further, incorporation of wetlands management under the CBD.

Wetlands Under the CBD and Collaboration with the Ramsar Convention

"Biodiversity," under the CBD, is defined as the variability among living organisms from all sources including, inter alia, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species, between species, and of ecosystems (Article 2 of the Convention – http://www.cbd.int/doc/legal/cbd-en.pdf). On this basis, any biodiversity living in or associated with wetlands as well as wetlands themselves (as ecosystems) are included under the scope of the CBD. In recent years, and in parallel with shifts in broader dialogues, including under the Ramsar Convention, consideration of "biodiversity" has moved beyond considering "species, genes, and ecosystems" to include a focus on the role of biodiversity in underpinning **ecosystem services** (benefits derived from ecosystems by people), thus assisting with improved articulation of the relationship between the objectives of the CBD and sustainable development.

The CBD has adopted the Ramsar Convention definition of "wetland" (decision VII/4 para. 27 http://www.cbd.int/decision/cop/?id=7741) which frames how wetlands are considered under its mandate. Collaborative work between Ramsar and the CBD is recognized as a model for how such streamlining can be done under existing multilateral environmental agreements (**MEAs**), while recognizing that each MEA has distinct but overlapping scope and mandate (for the most recent, see CBD COP10 decision X/20, http://www.cbd.int/decision/cop/?id=12286, and Ramsar COP-10 Resolution X.11, http://www.ramsar.org/pdf/res/key_res_x_11_e.pdf). As early as 1996, the Ramsar Convention was recognized as the "lead implementation partner" on wetlands for the CBD (Convention on Biological Diversity COP3 199, decision III/21). In essence, implementation of the commitments under the Ramsar Convention is simultaneously implementation of the CBD.

The history of collaboration between the two conventions, including implementation, is reviewed by Davidson and Coates (2011). In practice, Ramsar-CBD collaboration has been progressive and incremental, moving from affirmations of common interest, through recommending/adopting guidance prepared by one convention for use by the other, for example, CBD's impact assessment guidelines adopted by Ramsar with interpretation for their application in the Ramsar context (Ramsar COP-10 Resolution X.17 http://www.ramsar.org/pdf/res/key_res_x_17_e.pdf), to joint review and development of programs and guidance, including joint preparation and publication of technical reports requested by parties, and Ramsar's lead in undertaking the 2010 review and revision of the CBD program of work in the biological diversity of inland waters, providing the basis for the adoption of CBD COP10 Resolution X/28 on inland water biodiversity (http://www.cbd.int/decision/cop/?id=12294). The Ramsar guidance on wetland management, including relevant resolutions of the Ramsar Convention, supporting information, and in particular the *Ramsar Handbooks* for the wise use of wetlands (http://www.ramsar.org/cda/en/ramsar-pubs-handbooks/main/ramsar/1-30-33_4000_0__), is therefore de facto guidance for the CBD.

Following its inception, the development of policies and guidance under the CBD was dominated by the identification and adoption of various thematic programs of work and crosscutting issues as mechanisms to identify specific actions that need to be taken in order to fulfill the objectives of the convention (http://www.cbd.int/programmes/). The thematic programs of work are "biome" or sector based and include agricultural, dry and subhumid lands, forest, inland waters (which the Ramsar Convention was instrumental in designing), islands, marine, and coastal and mountain biodiversity. There is no specific program of work on "wetlands" which are covered most immediately by the programs of work on inland waters (where all identifiable habitats are wetlands using the Ramsar definition) and marine and coastal biodiversity (where most habitats up to 6 m depth are wetlands, using the Ramsar definition, which essentially excludes open oceans). There is however recognition that wetlands are ubiquitous and can be found in dry and subhumid areas, forests, agricultural areas, mountains, and islands and are therefore also relevant to these areas (decision VII/4, paragraph 11, http://www.cbd.int/decision/cop/?id=7741). Wetlands are pertinent to all of the CBD's crosscutting issues (which include topics such as climate change, gender, invasive alien species, etc.) among which **Wetlands of International Importance** (**Ramsar Sites**) are of particular relevance the CBD's crosscutting activities on protected areas.

The Strategic Plan for Biodiversity (2011–2020), and its **Aichi Biodiversity Targets**, has become the accepted framework for implementation for the MEAs, including for the CBD and the Ramsar Convention (see Biodiversity-Related Conventions and Ramsar). This has enabled improved opportunities for more holistic approaches to wetlands under the CBD, where the programs of work are seen as tools to support broader objectives. The improved recognition of water as an ecosystem service, and the role of wetlands, and other ecosystems, as natural infrastructure in supporting this service, significantly increases opportunities for recognizing the role of wetlands under the CBD and other MEAs.

Future Challenges

Despite significant areas of progress, there continue to be several constraints and limitations in addressing wetlands under the CBD as illustrated by the Ramsar-CBD cooperation achieved in practice (Davidson and Coates 2011). Most achievements have so far been at the global scale, through the work of the secretariats and the conventions' scientific and technical subsidiary advisory bodies (Ramsar's Scientific & Technical Review Panel (STRP) and CBD's Subsidiary Body on Scientific, Technical & Technological Advice (SBSTTA)). Although called for in both CBD and Ramsar decisions on collaboration between conventions, there is as yet much less consistency in the extent of national-scale communication and collaborative implementation including between the convention's respective national focal points, which in turn has been recognized as a key precondition to establish harmonized national reporting (UNEP-WCMC 2009).

Despite the early recognition of the importance of an ecosystem approach to implementation (CBD decision III/21, 1996), CBD parties and their decisions have focussed on wetlands mainly in the context of the CBD program of work on inland waters. CBD parties, in their separate development and adoption of their other biome and crosscutting programs of work, have paid much less attention to the relevance of wetland management in notably coastal and nearshore marine systems and in protected areas in relation to the designation and management of Wetlands of International Importance (Ramsar Sites). Only recently has the CBD given improved recognition to the relevance of Ramsar's lead implementation role for these matters, in its protected areas and marine and coastal biodiversity programs of work (CBD COP10 Decision X/31 on Protected Areas, http://www.cbd.int/decision/cop/?id=12297 and Decision X/29 on marine and coastal biodiversity http://www.cbd.int/decision/cop/?id=12295). An assessment of the interconnectivity between the CBD inland waters and marine and coastal programs of work (SCBD 2011) revealed that there is remarkably little linkage on common issues across the two programs and that the compartmentalized approach of the biome programs of work, as a major approach to CBD implementation, appears to hinder and run counter to the application of CBD's overarching ecosystem approach, which explicitly requires landscape and seascape scale cross-sectoral and integrated approaches to conservation and sustainable use, whereas the approach is embodied in Ramsar's definition of "wise use" (Ramsar Convention Secretariat 2005). For further discussion of "wise use" in relation to CBD's "ecosystem approach," see Finlayson et al. (2011). This lack of coherence between CBD's programs of work, or rather the failure to systematically use the ecosystem approach and not programs of work as the point of entry, further exacerbates problems with selective cross-referencing to wetland management and the role and guidance of the Ramsar Convention. Recent attention to the Strategic Plan for Biodiversity (2011–2020) and its Aichi Biodiversity Targets should help address these, and other, challenges to the improved recognition in practice of the role of wetland management.

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Strategic Plan for Biodiversity (2011–2020) and the Aichi Biodiversity Targets

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David Coates

Contents

Abstract

The Strategic Plan for Biodiversity 2011–2020 was adopted at the tenth meeting of the Conference of the Parties in 2010. Its purpose is to promote effective implementation of action on biodiversity by all parties and stakeholders as well as providing a flexible framework for the establishment of national and regional Strategic Plan for Biodiversity. It is also recognized as the overarching flexible framework relevant to the objectives of all the biodiversity-related conventions. The Convention on Wetlands (Ramsar), for example, recognizes the important

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contribution of its strategic plan toward the achievement of the Strategic Plan for Biodiversity 2011–2020.

Keywords
Strategic Plan for Biodiversity · Biodiversity · Aichi Biodiversity Targets · Biodiversity-related conventions

Introduction: The Origins and Purpose of the Strategic Plan for Biodiversity 2011–2020

The Strategic Plan for Biodiversity 2011–2020 was adopted at the tenth meeting of the Conference of the Parties to the **Convention on Biological Diversity** in 2010 (SCBD 2010). Its purpose is to promote effective implementation of action on biodiversity by all parties and stakeholders as well as providing a flexible framework for the establishment of national and regional biodiversity-related targets. It is also recognized as the overarching flexible framework relevant to the objectives of all the **biodiversity-related conventions. The Convention on Wetlands** (**Ramsar**), for example, recognizes the important contribution of its strategic plan toward the achievement of the Strategic Plan for Biodiversity 2011–2020 (Ramsar Secretariat 2012).

The rationale for the Strategic Plan for Biodiversity 2011–2020 is that biological diversity underpins ecosystem functioning and the provision of ecosystem services essential for human well-being: it provides for food security, human health, the provision of clean air and water, contributes to local livelihoods, and economic development, and is essential for the achievement of the **Millennium Development Goals**, including poverty reduction. The **vision** for the plan is: “Living in Harmony with Nature” where “By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy plant and delivering benefits essential for all people.” Its **mission** is to “take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services, thereby securing the planet’s variety of life, and contributing to human well-being, and poverty eradication.”

The Aichi Biodiversity Targets

The achievement of the vision and mission of the Strategic Plan for Biodiversity 2011–2020 requires pressures on biodiversity be reduced, ecosystems restored, biological resources used sustainably and benefits of genetic resources shared in a fair and equitable manner, adequate financial resources be provided, capacities enhanced, biodiversity issues and values mainstreamed, appropriate policies be effectively implemented, and decision-making be based on sound science and the precautionary approach. Such principles are reflected in five strategic goals accompanied by 20 “Aichi Biodiversity Targets” (so named because they were adopted in Aichi Prefecture in Japan):

Strategic Goal A: Address the Underlying Causes of Biodiversity Loss by Mainstreaming Biodiversity Across Government and Society

Target 1
By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.

Target 2
By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.

Target 3
By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out, or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the convention and other relevant international obligations, taking into account national socioeconomic conditions.

Target 4
By 2020, at the latest, governments, business, and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of the use of natural resources well within safe ecological limits.

Strategic Goal B: Reduce the Direct Pressures on Biodiversity and Promote Sustainable Use

Target 5
By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.

Target 6
By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally, and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems, and the impacts of fisheries on stocks, species, and ecosystems are within safe ecological limits.

Target 7

By 2020 areas under agriculture, aquaculture, and forestry are managed sustainably, ensuring conservation of biodiversity.

Target 8

By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.

Target 9

By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.

Target 10

By 2015, the multiple anthropogenic pressures on coral reefs and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.

Strategic Goal C: To Improve the Status of Biodiversity by Safeguarding Ecosystems, Species, and Genetic Diversity

Target 11

By 2020, at least 17% of terrestrial and inland water and 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative, and well-connected systems of protected areas and other effective area-based conservation measures and integrated into the wider landscapes and seascapes.

Target 12

By 2020 the extinction of known threatened species has been prevented, and their conservation status, particularly of those most in decline, has been improved and sustained.

Target 13

By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socioeconomically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.

Strategic Goal D: Enhance the Benefits to All from Biodiversity and Ecosystem Services

Target 14

By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods, and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.

Target 15

By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15% of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.

Target 16

By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.

Strategic Goal E: Enhance Implementation Through Participatory Planning, Knowledge Management, and Capacity Building

Target 17

By 2015 each party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory, and updated national biodiversity strategy and action plan.

Target 18

By 2020, the traditional knowledge, innovations, and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the convention with the full and effective participation of indigenous and local communities, at all relevant levels.

Target 19

By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.

Target 20

By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011–2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for Resource Mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs assessments to be developed and reported by parties.

Wetlands and the Strategic Plan for Biodiversity 2011–2020 and Aichi Biodiversity Targets

By and large the Strategic Plan for Biodiversity 2011–2020 is not biome or sector based, and wetlands are therefore not specifically mentioned, with the exception of “coral reefs” in target 10, but the Strategic Plan for Biodiversity 2011–2020 and its Aichi Biodiversity Targets apply equally to wetlands. For example, the term "biodiversity" in the Strategic Plan and its Aichi Targets includes, of course, all wetland-dependant species, and likewise "ecosystem" includes wetlands as ecosystems. Because of the role of wetlands in underpinning **water-related ecosystem services**, they also play a crosscutting role. The Strategic Plan for Biodiversity 2011–2020 also recognized the paramount importance of water as an ecosystem service (in its footnote 15). CBD COP-10 also recognized the clear basis to strengthen attention to water across all relevant interests and programs of work of the convention (SCBD 2010), and CBD COP-11 (2012) recognized the importance of the water cycle to achieving most of the Aichi Biodiversity Targets, requiring that the water cycle be considered as a crosscutting theme when implementing the Strategic Plan for Biodiversity 2011–2020 (SCBD 2012).

The Strategic Plan for Biodiversity 2011–2020 lays an improved framework facilitating enhanced collaboration including between all of the biodiversity-related multilateral environment agreements (MEAs). The improved recognition of water as an ecosystem service also significantly increases resonance between the strategic plan, and hence the MEAs, and the significant broader social, economic, and political interest in water.

Further details of the relationship between the CBD, wetlands, and the Ramsar Convention are provided in contribution CBD and wetland management.

Future Challenges

There are significant constraints to implementation of the plan and the achievement of the targets. Approaches to, and policies for, land and water use remain significantly fragmented and sector-based interests still dominate. To a large extent, the Aichi Biodiversity Targets require collective actions implemented at the landscape scale. This remains a significant challenge and the strategic plan is weak on guidance as to how this can be better achieved. Most of all, the key drivers of biodiversity loss, and in this case wetlands loss and degradation, are indirect including population growth, unsustainable consumption and production patterns, and inappropriate economic models and approaches (which externalize ecosystem values and benefits). Unless these, and other drivers, are addressed, moving society toward improved sustainability of natural resources, including land and water, and more equitable resource use, it will not be possible to adequately reduce the direct drivers of biodiversity loss (such as pollution, water extraction, and habitat conversion). Assessments suggest that it is possible to meet human needs for resources on a more sustainable and equitable footing – but significant shifts in policies and approaches are required to do this. Recognizing the role of ecosystems in sustaining natural resource availability, in this case in particular regarding water, will be a significant step toward redressing the current imbalance between ecosystem and development interests.

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Transactional and Regional Legal Frameworks

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Sacha Kathuria and Kirk W. Junker

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Abstract

Wetlands are governed not only by domestic law, but by transnational and regional legal frameworks as well. These transnational and regional legal tools should be considered as part of the overall legal regime for wetlands. Sometimes domestic wetlands law exists in order to comply with transnational or regional agreements. This article sets out to describe transnational sources of law and the nature of global problems that they are meant to remedy. After identifying these sources of law are, important examples are provided. As in other areas of international law, custom and principle also provide important sources of law for protecting and maintaining wetlands.

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Introduction

While domestic law, enabled by the sovereignty of the state, may well establish norms through the legal instruments of statutes, regulations, and the ensuing permits the geographic reach of those norms cannot extend beyond the state's own borders. Thus, when it comes to wetland management, a wetland's expanse or recharging area over more than one state can lead to the need for cooperation between and among the states concerned. The agreement of states is often recorded in treaties, conventions, or other types of international agreements. Moreover, if one considers a wetland's effect on ecosystems beyond its immediate recharging area, wetlands, like rivers, may eventually be determined to be best managed as the common property of all mankind, just as the oceans and seas are already.

Transnational Frameworks

Before examining how transnational frameworks work with regard to wetland management, one must understand the nature of the framework system. The mechanism through which international frameworks are crafted is the treaty. Treaties "are described as conventions, agreements, protocols, covenants, pacts, etc." (Guruswamy 2007). It is also important to note that the name of an agreement between or among states "is not decisive" (Aust 2010). Thus, calling an agreement a "treaty," "convention," or other name does not determine whether it is binding on the signatory states.

This also applies to framework conventions. Like all other international legal agreements, one must read the intent of the states, as expressed in the document, to determine its legal nature. One might imagine the term "framework" being applied to any legal structure. However, as a term of art, legal "framework" most often refers to framework conventions. "There is no fixed model for framework agreements and the term does not have a technical meaning" (Bodansky, as cited by Matz-Lueck 2013). The term "treaty" "suggests the most formal kind of agreement" (Brierly 1963). They can be bilateral, regional, or multilateral. The Vienna Convention, commonly referred to as the "treaty on treaties" defines a treaty as "an international agreement, concluded between States in written form and governed by international law, whether embodied in a single instrument or two or more related instruments and whatever its particular designation" (Vienna Convention, Article 2 1969).

Treaties, "serve as international legislation, or they may have the character of contracts between members of the international community" (McCaffrey 2006). Multilateral treaties serve as law-making "machinery" for the purpose of adapting international law "to new conditions and in general for strengthening the force of the rule of law between states" (Brierly 1963). Treaties are like contracts in the sense that "they derive their validity from the agreement of the parties" (Jessup 1959). Signatories to treaties are expected to follow the principle of *pacta sunt servanda*, i.e., agreements are binding and once agreed to, should be ratified or otherwise carried out in good faith. However, what makes the treaty particularly unique is its reservation option, which allows countries signing to opt out of certain provisions. According to the Vienna Convention, the reservation option is "a unilateral statement, however phrased or named made by a State [. . .] whereby it purports to exclude or modify the legal effect of certain provisions of the treaty in their application to that State" (Vienna Convention, Article 2 1969). A reservation does not require acceptance by other contracting states "unless the treaty so provides" (Vienna Convention, Article 20 1969). Once signed, a treaty may be self-executing or it may require a ratification process, depending upon the constitutional process of each country (Jessup 1959).

The framework convention has become a common multilateral treaty form, together with more specific protocols, in accomplishing the goals of states, especially in the field of environmental law, to which one can say that wetlands law belongs. "The regulation of international issues by framework conventions is a relatively recent regulatory technique in international law and has mainly been employed in the field of international environmental law. Framework agreements are usually associated with the so-called 'framework convention and protocol approach' by which parties agree on a more general treaty, the framework convention, and more detailed protocols to fill out the room left by the legal framework for specific regulations" (Matz-Lueck 2009). But even if the form of framework conventions turns out to be similar, it is not so by some sort of agreement or requirement. Therefore, one must use an inductive method of observation and usage to understand what is meant by the term. A quick review of the United Nations' framework conventions would yield such large and well-known examples as The UN Framework Convention on Climate Change (UNFCCC), The World Health Organizations Framework Convention on Tobacco Control, and The Framework Convention for the Protection of National Minorities, but also the lesser-known and very specific 2004 Framework Convention instituting cooperation for transboundary development relating to the Esch-Belval project between the Government of the French Republic and the Government of the Grand Duchy of Luxembourg.

Transnational denotes a global character, going beyond the reach of a country or region. Therefore, transnational or multilateral frameworks are global agreements that can serve the interests of the international community. Unlike regional frameworks, they are not meant for any certain geographic or cultural group. Nor are they provincial. Rather, they address concerns of international importance. Conservation of wetlands is one such international concern. Wetlands are a source of "economic, cultural, scientific, and recreational value," and their demise "would be irreplaceable" (Ramsar Convention 1975). Though there are several transnational frameworks affecting wetland management, three of the most pertinent will be discussed here: the Ramsar Convention, the Convention on Biological Diversity, and the Convention on Migratory Species. This purpose here is not to provide an all-inclusive list of transnational frameworks. The purpose here is to provide an overview of how a few of the most significant transnational frameworks affect wetland management through their objectives, mechanisms, and enforcement.

The objective of a transnational framework is to address a particular issue affecting the international community. International cooperation and consensus with regard to environmental issues is of special importance because one country's actions may dramatically affect the environment of its neighbors or indeed the entire world. Wetlands are affected by a variety of transboundary environmental factors, from air quality, species diversity, and migration to desertification. The main themes throughout the treaties examined here are the conservation of wetlands, biodiversity, and migratory species.

Other Sources of International Law

Treaties are not the sole source of international law. A legal framework should be understood to include other sources of law. According to Article 38 of the International Court of Justice Statute, primary sources of international law not only include international conventions but also international custom and general principles of law (International Court of Justice Statute, Article 38 1945). Customary law results from two components. The first is regarded as objective and is evidenced by general and consistent state practices. The second is regarded as subjective and is evidenced by the belief in a sense of legal obligation which is also called the *opinio juris*. General principles are those that are common to major legal systems and may be invoked as supplementary rules of international law (Yearbook 1950). The importance of a transnational framework is not diminished even if all countries do not accede to it. As Stephen McCaffrey explains, "A treaty may attract state practice which, when accompanied by *opinio juris*, eventually crystallizes into a norm of customary international law that binds even states that are not parties to the treaty" (McCaffrey 2006). This means that even if a state has not acceded to a treaty affecting wetland management, it may nonetheless be bound by it through other forms of international law. In practice, in the case of custom, this can mean that when a conflict arises between states, a state can be bound by its past practices. It also means, in the case of principle, that a state can announce an intended form of legal behavior to which it can be bound, even if it has not signed an agreement with another state or states to do so.

As to the situations when domestic law and international law conflict, the Vienna Convention on the Law of Treaties, operating as the rule book of treaties, states in Article 27 that "[a] party may not invoke the provisions of its internal law as justification for its failure to perform a treaty. This rule is without prejudice to Article 46" (Vienna Convention, Article 27 1969). Article 46, in turn, provides that "(1) A State may not invoke the fact that its consent to be bound by a treaty has been expressed in violation of a provision of its internal law regarding competence to conclude treaties as invalidating its consent unless that violation was manifest and concerned a rule of its internal law of fundamental importance. (2) A violation is manifest if it would be objectively evident to any State conducting itself in the matter in accordance with normal practice and in good faith" (Vienna Convention, Article 46 1969).

The Convention on Wetlands of International Importance especially as Waterfowl Habitat, commonly referred to as the Ramsar Convention, is the most significant transnational legal framework directly addressing the wetlands. Signed in Ramsar, Iran on February 2, 1971 and entered into force on December 21, 1975, it seeks "to stem the progressive encroachment on and loss of wetlands" by "combining far-sighted national policies with co-ordinated international action" (Ramsar Convention 1975). Ramsar is the first treaty with the principal objective to protect a particular type of ecosystem (Bowman 1999). Ramsar defines wetlands as "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres" (Ramsar Convention, Article 1 1975). It also considers waterfowl birds ecologically dependent on wetlands (Ramsar Convention, Article 1 1975).

Biodiversity is an important concept not only to wetlands but also to all ecosystems. The Convention on Biological Diversity (CBD) took effect on December 29, 1993 and defines biodiversity as "the variability among living organisms from all sources [. . .] and the ecological complexes of which they are a part: this includes diversity within species, between species and of ecosystems" (CBD, Article 2 1993). Its purpose is to "conserve biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources" (CBD, Article 1 1993). Sustainable use is "the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity" (CBD, Article 2 1993).

As the Ramsar Convention indicates, migratory species such as waterfowl birds are an important component to the wetland ecosystem. The Convention on the Conservation of Migratory Species of Wild Animals (CMS or Bonn Convention) signed in Bonn, Germany and entered into force on November 1, 1983 is "to take action to avoid any migratory species becoming endangered" (CMS Convention, Article 2 1983). Migratory species are defined as "the entire population or any geographically separate part of the population of any species or lower tax of wild animals, a significant portion of whose members cyclically and predictably cross one or more national jurisdictional boundaries" (CMS Convention, Article 1 1983).

Achieving the Objectives of International Frameworks

Once a framework objective is agreed upon, countries need to decide how they will achieve their goals. Some frameworks are better than others in identifying mechanisms. While all frameworks state that something can be done to combat the problem at hand, often times they are vague and lofty. This may be partially due to the sheer size of participation in a transnational framework. Taking into account all the different challenges, viewpoints, and attitudes from around the world, the more generalized the mechanisms, the more free contracting parties are to interpret them to suit their needs. It is therefore up to the contracting parties of the transnational treaty to take concrete action to ensure objectives of the framework are met. Each contracting party can take steps domestically, such as through national legislation, public awareness, and education, to achieve framework objectives.

Ramsar seeks to attain its objectives through the "wise use" of wetlands. Wise use is the maintenance of ecological character, "achieved through the implementation of ecosystem approaches, within the context of sustainable development" (Ramsar Introductory 2008). Ramsar also requires contracting parties to designate wetlands within their respective territories to be on a List of Wetlands of International Importance (also called the Ramsar List) (Ramsar Convention, Article 2 1975). The contracting parties are to formulate plans to promote conservation of sites on the Ramsar List and to share information regarding any changes to the ecological character of any such sites. (Ramsar Convention, Articles 3 and 2 1975). They are also to consult with each other about implementing objectives especially as they pertain to international wetlands (Ramsar Convention, Article 5 1975).

Parties to the Convention on Biological Diversity (CBD) agree to cooperate with each other and to develop national strategies to attain the goals of the framework (CBD Articles 5 and 6 1993). Each country is expected to identify components of biological diversity, monitor them, identify processes that negatively impact conservation and sustainable use, and maintain and organize identification and monitoring activities, (CBD, Article 7 1993). The CBD specifically refers to *in situ* conservation in Article 8, which is the conservation of "ecosystems and natural habitats and the maintenance and recovery of viable populations of species in their natural surroundings" or "where they have developed their distinctive properties" (Article 2, CBD 1993). Contracting parties to the CBD are to establish systems of protected areas for *in situ* conservation and regulate, promote, rehabilitate, and prevent the eradication of such ecosystems (CBD, Article 8 1993). Likewise, in an effort to complement *in situ* conservation, contracting parties should also adopt measures for *ex situ* conservation by adopting measures for conservation. *Ex situ* conservation is the "conservation of components of biological diversity outside their natural habitats" (CBD, Article 8 1993). This means establishing and maintaining facilities for research, creating measures for the reintroduction of threatened species into their natural habitats, and by cooperating financially to facilities in developing countries (CBD, Article 9 1993).

Contracting parties to the CBD are to carry out these expectations by offering incentives in their respective countries to comply with sustainability and conservation, and provide research, training, public education, and awareness (CBD, Articles 11, 12, 13 1993). Each country is expected to take an assessment and minimize adverse impacts of projects, programs, and policies on biodiversity (CBD, Article 14 1993). In addition to developing national policies and legislation, contracting parties to the CBD agree to cooperate together on creating access to genetic resources, transferring technology, and exchanging information (CBD Articles 15, 16, 17, 18 1993). Moreover, they are expected to engage in biotechnological research and provide financial support to initiatives within their capabilities (CBD, Articles 19 and 20 1993).

The Convention on Migratory Species measures the success of conservation efforts by gleaning whether migratory species are maintaining themselves on a long-term basis, if their range is being reduced, the ability of sufficient habitat in the long-term, and whether the distribution and abundance of migratory species is consistent with suitable ecosystems and wise wildlife management (CMS Convention, Article 1 1983). Parties to CMS are to cooperate in research relating to migratory species, provide immediate protection for them, and conclude agreements covering the conservation and management of migratory species (CMS Convention, Article 3 1983). This can be done through the conservation and restoration of habitats of endangered species and the prevention of activities and situations that threaten migratory species (CMS Convention, Article 3 1983). Furthermore, "Range States" under CMS, that is, states that have jurisdiction over a range of migratory species, are expected to conclude agreements that benefit species and give them priority (CMS Convention, Article 3 1983). As used in the CMS Convention, "arguments" "means international agreements relating to the conservation of one or more migratory species" (CMS Convention, Article 1 1983).

While negotiating and developing a framework of any kind is difficult, perhaps the most challenging part is enforcing the agreement. This is especially true with environmental frameworks on a large, transnational scale. It is inevitable that some countries will be better than others at enforcement. Much of this has to do with wealth, threat perception, and the rule of law. Ultimately enforcement is up to each individual state.

Perhaps because it is an older treaty, Ramsar does not provide for strict enforcement mechanisms. Nevertheless, Ramsar has modernized itself through the issuance of a Management Guidance Procedure and the development of the Wetland Conservation Fund to help finance small conservation projects. The CBD and CMS take into account the possibilities of disputes between contracting parties. Failure to live up to terms of frameworks affects all parties. Annex II of the CBD provides for arbitration and conciliation procedures for contracting parties in case of a dispute. (CBD, Annex II 1993). Article 13 of the CMS calls for negotiation between contracting parties in case two or more parties are in a dispute. If the dispute is still not resolved, then by mutual consent, parties are advised to submit the dispute to the Permanent Court of Arbitration at The Hague. (CMS, Article 13 1983).

Transnational frameworks are an important part of international environmental law and indeed wetland management. They provide a forum for several diverse countries to combat some of the most difficult issues affecting humankind. While transnational frameworks may sometimes be vague in an effort to appease all parties, they nonetheless bring important problems to the forefront of international consciousness and help contracting parties develop solutions. Although enforcement is often elusive, dispute resolution efforts and negotiation provide a means of overcoming this persistent hurdle.

Regional Agreements

"Regions" is a legal category for public international law that has historically been understood to mean something more than one state, but less than all states (or approximately all states.) There are regional conventions for trade, exploitation of natural resources, prosecution of crime, and environmental protection. Thinking about regional agreements requires that one think about why such a species of agreement should exist. Why not solve wetlands problems through the establishment of the most wide-reaching norm possible, as would be done with a transnational agreement like the UNFCCC? The mantra of "think globally, act locally" addresses two polar positions of legal thinking. As an alternative to these polar positions of global and local, when should one think regionally or act regionally, specifically when it comes to wetlands protection, management, or creation? A first answer would be tied to the object of concern – the wetland itself. The recharging area of a wetland, like so many other natural phenomena of environmental law, may well be spread over more than one state. What is perhaps more interesting, however, is that states should want to come together because they recognize that the protection, management, and creation of wetlands is a concern common to all residents of the earth. It is not difficult to provide a catalogue, even if incomplete, of the extant regional wetlands agreements. Therefore, what can one say is common to the regional agreements that gives them their own character? A survey might show that they affect only interested states, whereas a transnational agreement might be signed and ratified by states which agree with the norms, but which do not directly experience the effects of those norms, because the problems covered have yet to arise in those states.

Regional agreements are typified, as their name says, by applying to a particular and limited geographic region. (See, for example The North American Wetlands Conservation Act 1989, The African-Eurasian Waterbird Agreement 1995, The African Convention on the Conservation of Nature and Natural Resources 2003, The Revised Southern African Development Community (SADC) Protocol on Shared Watercourse Systems 1985, revised 2000 [SADC Protocols are legally binding among members], The South Africa DC Protocol on Fisheries 2001, or The South Africa DC Protocol on Forestry 2002).

It is not only the case, however, that a regional agreement is drafted by and applies to a limited set of states for a limited purpose, such as wetland management and related issues. The European Union, for example, while unique in its vision of a modicum of shared sovereignty, is nevertheless held together by a set of regional agreements that when seen on the whole, create a complete government structure in all branches of government with competencies in nearly all areas of legislation. The European Union's legal instruments concerning wetlands include the European Union Habitats Directives Annex 1 (92/43/EC), The European Union Water Framework Directive (2000/60/EC), The European Union Birds Directive (2009/147/EC), The European Commission's Communication "Flood Risk Management: Prevention, Protection, Mitigation" of 12 July 2004, or Council Directive on the Conservation of Wild Birds (2009/147/EC). It should be here mentioned that for better or worse, it has been noted that the African Union, successor to the Organization of African Unity, has often copied European legal institutional structures and practices.

In addition there are European regional agreements that are not part of the Union's shared-sovereignty competencies and just happen to be entered into by a subset of states on the European continent. The European regional agreements that are not part of the European Union's competencies include The Barcelona Convention (signed February 16th, 1976, in force February 12th, 1978, and revised in Barcelona, Spain on June 10th, 1995 as the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean), the Bonn

Convention on the Conservation of Migratory Species of Wild Animals of 1983, the Convention on Cooperation for the Protection and Sustainable Use of the Danube River of 1994, the Programme on the Sustainable Development of the Rhine 2001, also known as “Rhine 2020,” The Agreement on Conservation of Seals in the Wadden Sea 1990, The Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas 1991, and The Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea, and Contiguous Atlantic Area 1996.

Chapter VIII of the United Nations Charter enables member states to enter into Regional Agreements. Without this explicit permission, one might have argued that a United Nations member state could act only in concert with all other United Nations member states. Within Chap. VIII, Article 52 and subsequent articles focus exclusively upon agreements concerning peace and conflict, however, and neither mention the environment or biodiversity in general nor wetlands in particular. To that end, Chap. VIII is largely designed to be permissive rather than restrictive; that is, it allows United Nations member states to enter into regional agreements.

Regional agreements are concluded with respect to political, economic, military, scientific, technical, and cultural matters. A literature review of regional agreements will demonstrate that much has been written on the nature, advantages, and disadvantages of regional trade agreements. The Organization of Economic Cooperation and Development notes that in the first 11 years of the World Trade Organization, the number of regional trade agreements notified to the World Trade Organization alone tripled from 58 to 188 in number (OECD 2013). The Organization of Economic Cooperation and Development divides the lessons that we can learn from regional trade agreements as follows: “[t]he first lesson is that many consequences of regional trade agreements activity bolster the case for a strengthened multilateral framework” (OECD 2013). According to the Organization of Economic Cooperation and Development, “[t]his applies particularly to the contribution of regionalism to divergence from the rules of the multilateral system, to the effects which the patchwork of regionalism can have on non-members of those agreements and to the role of regionalism in raising transaction costs for business” (OECD 2013). The second lesson, according to the Organization of Economic Cooperation and Development, is that there are features of regional approaches that complement multilateral agreements or even be drawn upon in designing strengthened multilateral rules. “The scope for complementarity arises from the contribution which regional initiatives can make towards harmonisation of rule making; the scope for drawing upon arises from the extent to which Regional Trade Agreements (RTAs) go beyond the WTO. Together, these two elements have yielded highly effective synergies between approaches at the regional and the multilateral levels” (OECD 2013).

Transnational and Regional Agreements

The interplay between transnational and regional wetland agreements has not been studied in such detail and the question as to whether the same observations the Organization for Economic Cooperation and Development made of trade agreements can be said to apply to wetlands agreements remains unanswered. At least from trade agreements one can observe that it is possible that regional agreements may both diverge from multilateral agreements and be used to strengthen multilateral agreements. Like transnational legal frameworks, regional legal frameworks are an important part of international environmental law, biodiversity protection, and wetland management. Some of the vagueness of transnational frameworks may be alleviated in regional frameworks by including only affected parties to the wetlands protection scheme.

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Waterbird Flyways and History of International Cooperation for Waterbird Conservation

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Nick C. Davidson and David A. Stroud

Contents

Abstract

Many waterbirds are migratory along flyways which connect their breeding, staging and non-breeding areas, and for conservation purposes can be considered under single-species migration systems, multispecies flyways and global regions for waterbird conservation management. There are different types of flyway-scale initiatives: facilitative, formal or legally binding; bilateral or multilateral. A wide range of outcomes are sought through international flyway cooperation, but there are continuing major challenges to achieving successful migratory waterbird conservation.

Keywords

Waterbirds · Flyway · International cooperation · Migration system · Ramsar convention · Convention on migratory species · African-Eurasian migratory waterbird agreement

This text is drawn largely from Boere and Stroud (2006), Boere et al. (2006), and Global Interflyway Network (2012), to which reference should be made for further information.

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Waterbird Flyways

Flyways can be considered at different scales.

Single species migration systems: the distributional extent of the annual migration of a species, or a population within a species, encompassing breeding, staging and nonbreeding areas. While often described as the flyways of the species concerned, such annual distributional ranges are better described as the migration system of the species concerned.

Multispecies flyways: as defined by the Ramsar Convention on Wetlands:"*A single flyway is composed of many overlapping migration systems of individual waterbird populations and species, each of which has different habitat preferences and migration strategies. From knowledge of these various migration systems it is possible to group the migration routes used by waterbirds into broad flyways, each of which is used by many species, often in a similar way, during their annual migrations*." This definition was developed by the International Wader Study Group (IWSG 1998) and subsequently adopted by the Ramsar Convention (Ramsar Convention 1999).

Research into the migrations of many wader or shorebird (Charadrii) species, for example, indicates that the migrations of waders can broadly be grouped into eight flyways (Fig. 1). However, it is important to note that these flyways apply only to waders/shorebirds and not to other waterbird taxa – although this graphic has been widely, but erroneously, used for all waterbirds, particularly by the media in relation to highly pathogenic avian influenza (HPAI) issues.

Global regions for waterbird conservation management: At a larger scale still are global regions containing species with similar migration systems that are the

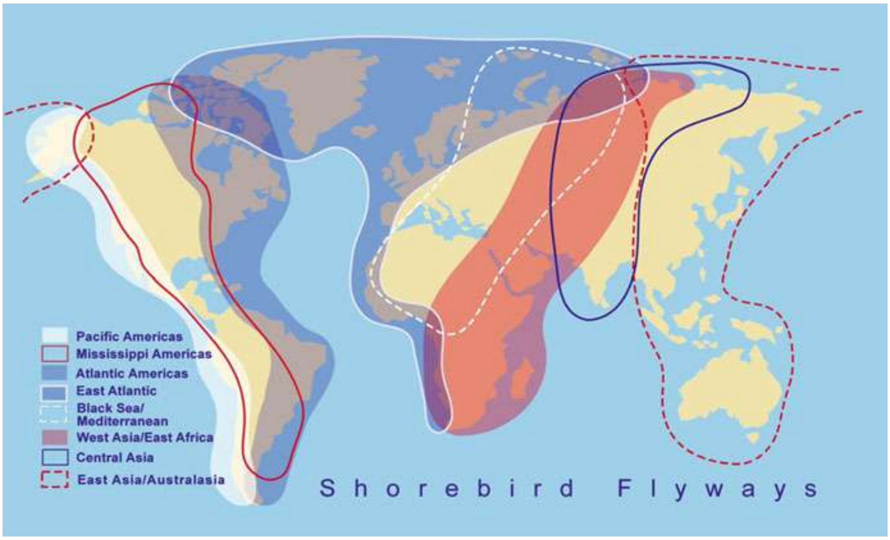

Fig. 1 Wader (shorebird) flyways (Redrawn from IWSG (1998))

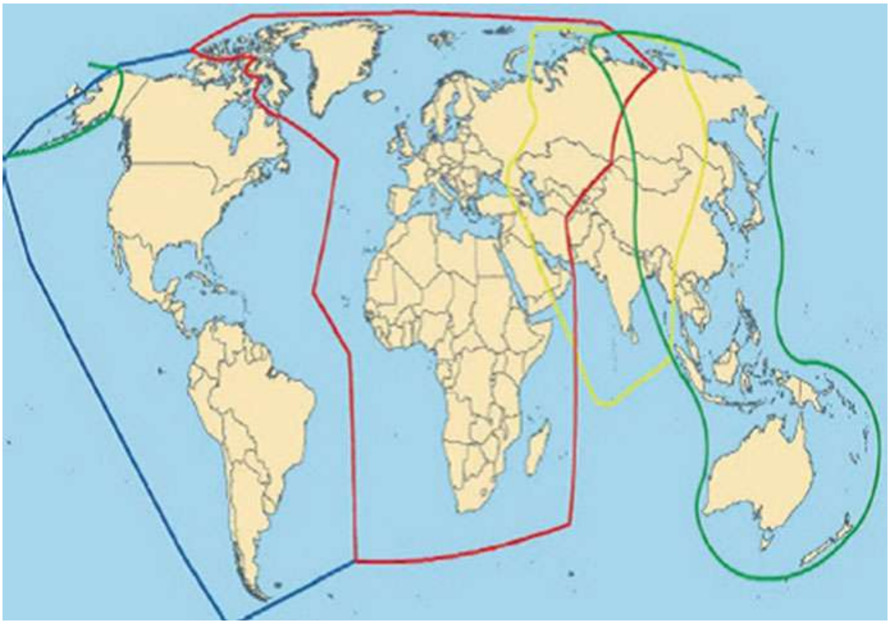

Fig. 2 The main "geopolitical" flyways for waterbird conservation (From Global Interflyway Network (2012))

subject (actual or potential) of shared international conservation activity, i.e., "geo-political flyways" (Fig. 2). Thus, the agreement area for the African-Eurasian Agreement on the conservation of migratory waterbirds (AEWA) is the area that contains the migration systems of all migratory waterbirds that occur in Africa and western Eurasia. A similar approach has been applied to the main flyway systems of the Asia-Pacific region. lt contains multiple flyways of different waterbird taxa, and its value is in terms of the political and governmental processes of international cooperation.

What Are Waterbirds?

Waterbirds are those species which are ecologically dependent on wetlands, for all or part of their annual cycle. As defined by Ramsar Convention (Ramsar Convention 1999), this can include any wetland bird species, but at the broad level of taxonomic order it includes especially:

- Penguins: *Sphenisciformes*
- Divers: *Gaviiformes*
- Grebes: *Podicipediformes*

(*continued*)

- Wetland related pelicans, cormorants, darters, and allies: *Pelecaniformes*
- Herons, bitterns, storks, ibises, and spoonbills: *Ciconiiformes*
- Flamingos: *Phoenicopteriformes*
- Screamers, swans, geese, and ducks (wildfowl): *Anseriforme*s
- Wetland-related raptors: *Accipitriformes* and *Falconiformes*
- Wetland-related cranes, rails, and allies: *Gruiformes*
- Hoatzin: *Opisthocomiformes*
- Wetland-related jacanas, waders (or shorebirds), gulls, skimmers, and terns: *Charadriiformes*
- Coucals*: Cuculiformes*
- Wetland-related owls: *Strigiformes*

Many species in other bird orders and families are also recognized as "wetland dependent" because of their ecological dependence on wetlands. Although there is no standard global list of such species, see BirdLife International (2002) and Crosby and Chan (2005) for lists of such species for Africa and Asia, respectively.

Several other terms are in wide use for different groups of waterbirds. These include:

- **Shorebirds**: A term increasingly used for species in the order Charadriiformes (excluding seabirds) such as plovers, sandpipers, oystercatchers, stilts and avocets. In some parts of the world these species are called **waders**.
- **Wading birds**: species of long-legged waterbirds such as herons and egrets, ibises, storks, cranes and flamingos. Confusingly, in North America these species are sometimes referred to as "waders."
- **Wildfowl**: species of ducks, geese and swans in the family Anatidae.
- **Waterfowl**: often now treated as a term synonymous with "waterbirds," but in North America usually covers only species in the order Anseriformes.

The terms wildfowl and waterfowl have hunting connotations, "fowling" being an old word for hunting birds, so these terms have often been used when referring to those species of waterbirds which are, or have been, quarry species.

For research and conservation purposes, waterbirds are often grouped in **guilds**: groups of species with similar ecology (most often feeding ecology) and morphology. There is no standard suite of waterbird guilds. Rather, each researcher has defined one or more such guilds based on the waterbird species composition at a wetland, but they typically include groupings such as:

(*continued*)

- *Herbivores or omnivores feeding in sheltered, shallow water* (e.g., ducks and swans)
- *Diving waterbirds* (mostly fish-eating: e.g., divers, cormorants, darters, pelicans, some seabirds)
- *Wading birds*: long-legged species feeding in shallow waters (e.g., herons and egrets, ibises, storks, cranes and flamingos, and some shorebirds such as stilts and avocets)
- *Omnivores and insectivores* feeding in dense vegetation on wetland margins (e.g., coots, gallinules, and rails)
- *Species feeding on shorelines and in sheltered, shallow waters*, often in tidal areas (e.g., shorebirds)
- *Fish-eating and scavenging waterbirds,* feeding along the shoreline and on the surface of shallow waters (e.g., gulls and terns)
- *Wetland-dependent raptors* (e.g., Osprey, sea-eagles, some harriers)

Waterbird Flyway Conservation Initiatives

There are different categories of flyway initiative. They may be facilitative, formal, or legally binding; some are bilateral, others multilateral. In general, the development of different waterbird flyway "initiatives" has followed a similar pathway, illustrated in Fig. 3.

Flyway initiatives have a long history, some dating back to the early twentieth century, although most have been developed since the early 1970s (Fig. 4).

What do such initiatives try to achieve? Generally a range of outcomes, including:

- High-level acknowledgment of shared international heritage/resources
- Agreement by governments on common conservation objectives and goals (leading to common policies and laws (e.g., lists of protected species) and joint actions)
- Establishing international standards for national conservation action
- Data and information exchange
- Formal transboundary cooperation (e.g., for Ramsar Sites)
- Joint working/surveys/monitoring
- Joint funding of international scale initiatives

Working with governments, international and national NGOs have played, and continue to play, key roles in flyway initiatives, including:

- *Initiating* (e.g., Ramsar Convention, Convention on Biological Diversity)
- *Supporting* (e.g., Ramsar's International Organization Partners) through providing access to science

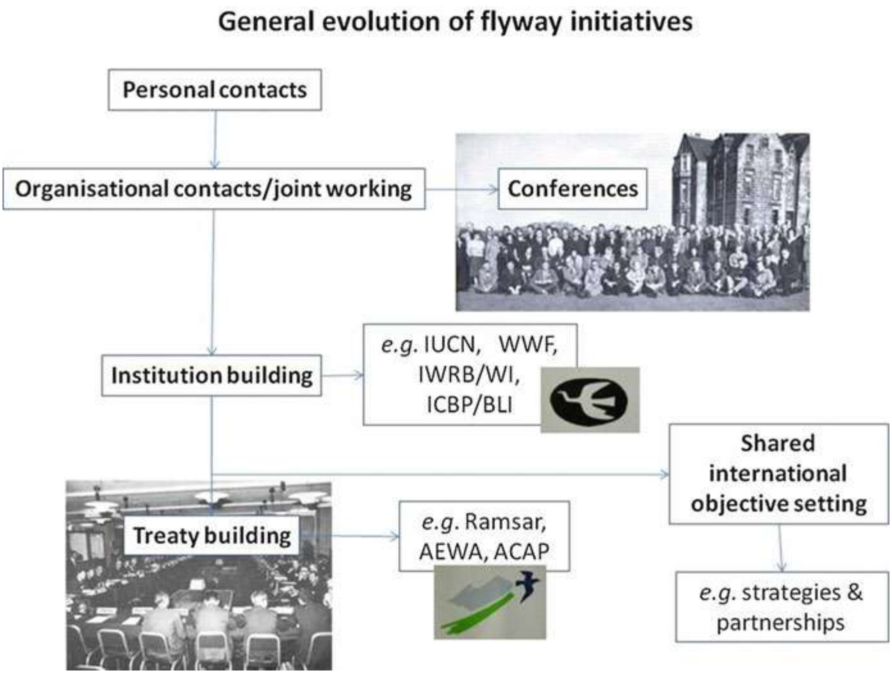

Fig. 3 The general evolution of waterbird flyway initiatives (From: Davidson and Stroud in Global Interflyway Network (2012))

- *Facilitating* (e.g., encouraging formal transboundary cooperation)
- *Encouraging* (reminding governments of the obligations they have assumed)
- *Joint working with governments* (e.g., Wetlands International's International Waterbird Census (IWC))
- *Monitoring and assessing sites* (e.g., Important Bird Areas, Ramsar Sites) *and species* (e.g., IUCN Red List)

Operationalizing Waterbird Flyway Conservation

There are many continuing challenges facing achieving successful migratory waterbird conservation, given that many waterbird species and populations continue to be in decline (see, e.g., Wetlands International 2010). These challenges, and recommendations to address them, are summarized and provided in Boere et al. (2006) and Global Interflyway Network (2012).

One major challenge is capacity and resourcing for implementation, which for the African-Eurasian migratory waterbird region was supported through a recent major project: "Wings over Wetlands" (WOW) supported by the Global Environment Facility (GEF) and many other donors. WOW provides a good example of the

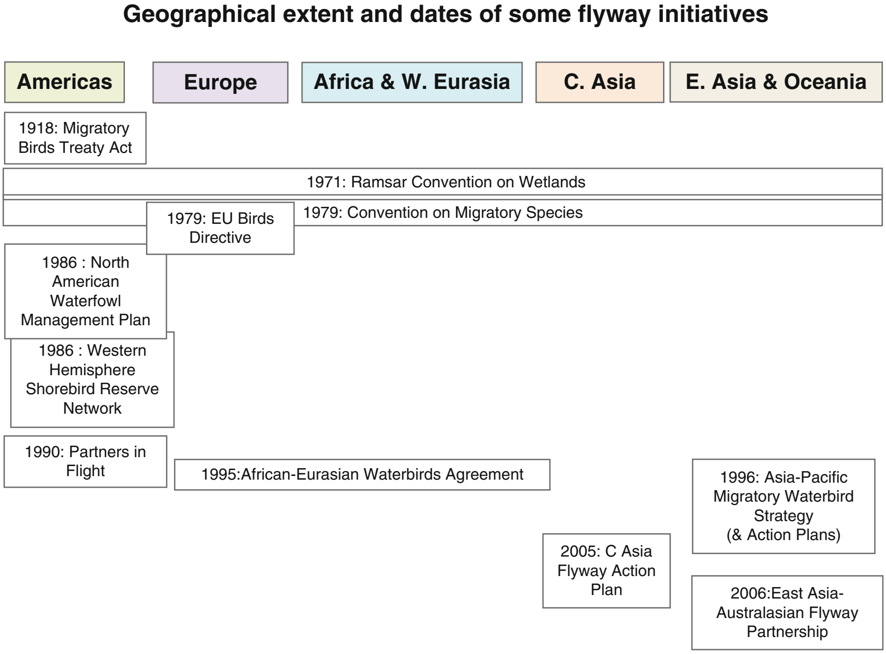

Fig. 4 The dates and geographical scope of different waterbird flyway initiatives (From: Davidson and Stroud in Global Interflyway Network (2012))

need and importance of collaborative partnerships for flyway conservation: it was a highly success collaboration between two multilateral environmental agreements (AEWA and the Ramsar Convention) and two international NGOs (BirdLife International and Wetlands International).

As well as supporting practical, on-the-ground, waterbird conservation implementation in a range of demonstration wetlands throughout the flyway, and supporting regional training centers in delivering flyway conservation training for practitioners, WOW produced two major support tools for waterbird flyway conservation. These are an online GIS-based "Critical Site Network" (CSN) tool, and a Flyways Training Kit. The approach of the CSN tool, and much of the content of the Flyways Training Kit, is highly relevant to implementing waterbird conservation on other flyways. Further information about WOW and access to these two support tools is on: http://www.wingsoverwetlands.org/.

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African-Eurasian Waterbird Agreement (AEWA) and Wetland Management

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Robert J. McInnes

Contents

Abstract

The Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA) is an intergovernmental treaty, which targets the conservation of migratory waterbirds and their habitats across Africa, Europe, the Middle East, Central Asia, Greenland, and the Canadian Archipelago. The agreement was developed under the framework of the Convention on Migratory Species (CMS) and is administered by the United Nations Environment Programme (UNEP). AEWA brings together countries and the wider international conservation community in an effort to establish coordinated conservation and management of migratory waterbirds throughout their entire migratory range.

Keywords

International framework · Waterbirds · Flyway · Migration

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Introduction

The Agreement on the Conservation of African-Eurasian Waterbird Agreement (AEWA) is an intergovernmental treaty which targets the conservation of migratory waterbirds and their habitats across Africa, Europe, the Middle East, Central Asia, Greenland, and the Canadian Archipelago. The Agreement was developed under the framework of the Convention on Migratory Species (CMS) and is administered by the United Nations Environment Programme (UNEP). AEWA brings together countries and the wider international conservation community in an effort to establish coordinated conservation and management of migratory waterbirds throughout their entire migratory range.

Species Covered by AEWA

AEWA covers 255 species of birds which are dependent on wetlands for at least part of their annual cycle. The Agreement therefore addresses many species of divers, grebes, pelicans, cormorants, herons, storks, rails, ibises, spoonbills, flamingos, ducks, swans, geese, cranes, waders, gulls, terns, tropic birds, auks, frigate birds, and even the South African penguin.

All AEWA species traverse international boundaries during their migrations and are dependent on high-quality habitats for breeding as well as a network of suitable feeding and roosting sites along their flyway. International cooperation across their entire migratory range of these wetland-dependent species is therefore essential for the conservation and management of migratory waterbird populations and the habitats on which they depend.

History of AEWA

The Convention on Migratory Species (CMS) of Wild Animals came into force in 1983, with the specific goal to provide for the conservation of migratory terrestrial, marine, and avian species throughout their range. At the first Conference of Parties (COP1) in 1988, CMS decided to prepare an Agreement for the Western Palearctic Anatidae. The Dutch government developed a draft Western Palearctic Waterfowl Agreement as part of its Western Palearctic Flyway conservation program; however, during the drafting process, the name of the Agreement was changed into the African-Eurasian Waterbird Agreement (AEWA), in order to emphasize the importance of Africa for migratory birds.

In June 1991, the final negotiation meeting on AEWA took place in The Hague, the Netherlands. The meeting adopted the Agreement by consensus and also accepted the offer from the Government of the Kingdom of the Netherlands to act as depositary and to provide an interim secretariat at its own expense until 1 January 1999. On 1 January 1996, the Dutch government established the interim secretariat, and from 15 August the same year, the Agreement became open for signature. By

November 1999, the required number of at least 14 range states, comprising at least 7 from Africa and 7 from Eurasia, had formally signed and the Agreement entered into force.

In 2000 a permanent secretariat was established and colocated with the convention secretariat in Bonn and administered by UNEP. A more detailed review of the history of AEWA is provided by Boere (2010).

Parties and Range States

Currently 73 countries and the European Union (EU) have become a contracting party to AEWA (as of 1 October 2014). A total of 39 contracting parties are from Eurasia and 33 from Africa.

Bodies of the Agreement

The Agreement has three main bodies: the Meeting of the Parties (MOP), which is the governing body of AEWA, and the Standing and Technical Committees, respectively, responsible for steering the operations between sessions of the MOP and for providing scientific advice. The UNEP/AEWA secretariat supports the parties and services the bodies of the Agreement.

Meeting of the Parties (MOP)

Established in accordance with Article VI of the Agreement, the Meeting of the Parties (MOP) is the principal decision-making body for the Agreement. The agreement secretariat convenes ordinary sessions of the MOP at intervals of not more than 3 years, unless the Meeting of the Parties decides otherwise.

The MOP is attended by representatives from contracting parties. Observers of non-party range states as well as any agencies or bodies technically qualified in relevant conservation matters or in research on migratory waterbirds can also be represented at sessions of the MOP. The MOP may make recommendations to the parties, the secretariat, the other AEWA subsidiary bodies, and stakeholders by adopting relevant resolutions and taking decisions on substantive or administrative matters.

The Standing Committee

The Standing Committee (StC) is a subsidiary body to the Agreement which provides policy and administrative guidance on behalf of the MOP to the agreement secretariat between sessions of the (MOP). Its main activities are to:

- Provide guidance on the implementation of the Agreement and management of the secretariat's programs
- Oversee the development and execution of the secretariat's budget and also all aspects of fundraising undertaken by the secretariat
- Liaise with parties and promote the flow of information to parties and vice versa
- Maintain regular contact to non-party range states and promote their accession to AEWA
- Report to the MOP on the activities that have been carried out between ordinary sessions of the MOP

The StC comprises:

- Representatives from five contracting parties, based on the principle of balanced geographical distribution (two from the Europe and Central Asia region, one from the Middle East and Northern Africa region, one from the Western and Central Africa region, and one from the Eastern and Southern Africa region)
- A representative of the host country for the next session of the meeting of the parties
- A representative from the depositary

The Technical Committee

The Technical Committee (TC) is a subsidiary body to the Agreement which has the following general functions:

- It provides scientific and technical advice and information to the MOP and, through the agreement secretariat, to parties.
- It makes recommendations to the MOP concerning the action plan, implementation of the Agreement, and further research to be carried out.
- It prepares a report on its activities for each ordinary session of the MOP.
- It carries out any other tasks referred to it by the MOP.

The TC membership comprises:

- Nine experts representing the different regions of the agreement area (Northern and Southwestern Europe, Central Europe, Eastern Europe, Southwestern Asia, Northern Africa, Central Africa, Western Africa, Eastern Africa, and Southern Africa) elected among all the parties of the region in question
- One representative appointed by each of the following organizations: the International Union for Conservation of Nature (IUCN), the Wetlands International, the International Council for Game and Wildlife Conservation (CIC)
- One thematic expert from each of the following fields: rural economics, game management, and environmental law, elected by the parties

Observers of non-party range states and the chairperson of the AEWA StC may be invited to the meetings of the TC. A maximum of four observers from specialized international intergovernmental and nongovernmental organizations can also be admitted, and in addition invited experts may be asked to contribute to specific topics.

The Secretariat

The secretariat is the Agreement's coordinating body. Its functions include:

- The organization and servicing of sessions of the MOP and meetings of the TC and the StC
- The implementation of tasks referred to it by the MOP
- The promotion and supervision of research and conservation projects
- The encouragement of exchange of information between the parties
- The liaison with international governmental and nongovernmental organizations

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North American Waterfowl Management Plan (NAWMP)

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C. Max Finlayson

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Abstract
The North American Waterfowl Management Plan (NAWMP) is an international plan established in 1986 by Canda and the United States with Mexico joining later. The purpose of the partnership established by the NAWMP is to conserve and protect wetland and upland habitats and associated waterfowl populations by connecting people with nature. It was established in response to the increasingly recognized continental scale decline of wetlands and waterfowl across North America, particularly in agricultural areas. It provides a policy framework that describes the scope and goals, identifies problems facing waterfowl populations, and sets general guidelines for addressing the problems.

Keywords
Waterfowl · Conservation · Wetlands · Partnerships

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Introduction

The North American Waterfowl Management Plan (NAWMP) is an international plan established in 1986 by Canada and the United States. The purpose of the partnership established by the NAWMP is to conserve and protect wetland and upland habitats and associated waterfowl populations by connecting people with nature. It was established in response to the increasingly recognized continental scale decline of wetlands and waterfowl across North America, particularly in agricultural areas (e.g., see Tiner 1984). It provides a policy framework that describes the scope and goals, identifies problems facing waterfowl populations, and sets general guidelines for addressing the problems.

Patterson (1995) described the effort to develop the NAWMP, starting in 1979 with agreement that a plan that was continental in scope was needed to resolve the ecological factors behind effective waterfowl conservation that went well beyond the capability of any single agency or country. This approach was developed as a consequence of widening recognition that traditional approaches to waterfowl conservation, encompassing regulation of hunting and protecting important and critical habitat were not successful. The private sector in particular Ducks Unlimited encouraged the Canadian and US Governments to develop a joint approach that would provide more effective conservation.

The plan contained a strategy for regaining the waterfowl population levels that were common in the 1970s and called for \$1.5 billion to be spent over 15 years on habitat conservation with \$1 billion of this being spent in Canada. Each country needed to raise funds from a variety of public and private sources. Patterson (1995) considered this funding arrangement as one of the best thing to happen to the plan as it forced those involved to market the concept and to seek funding from beyond the traditional wildlife agencies. Mexico joined in 1988 and became a signatory to the conservation action plan in 1994. Hence, the NAWMP was extended across all of North America, working at national and regional levels on waterfowl and habitat management issues.

Patterson (1995) also noted that for the NAWMP to be successful it needed to engage with the agricultural sector and address policies and practices that led to the loss and degradation of wetlands and waterfowl populations. This necessitated a focus on common solutions both nationally and locally and enabled pooled money to be used across borders (Hollis et al. 1992).

Implementation of the NAWMP

The North American Wetlands Conservation Council (Canada) has provided leadership and a national mechanism for implementation of the plan in Canada. Information on the plan and the Canadian approach to its implementation is available at http://nawmp.wetlandnetwork.ca/index.php. In the United States, the NAWMP was authorized by the North American Wetlands Conservation Act of 1989(P.L. 101–233) and is administered by the Fish and Wildlife Service, with

USDA agencies participating as appropriate. Information on the plan and the United States approach to its implementation is available atwww.fws.gov/birdhabitat/NAWMP/index.shtm. Information from these websites has been used to provide an overview of the implementation of the NAWMP.

Leadership for the NAWMP is provided by the North American Waterfowl Management Plan Committee which provides a forum for discussion of major, long-term international waterfowl issues and makes recommendations to directors of the three countries' national wildlife agencies. It responsible for updating the plan, considering new scientific information and national and international policy developments, and for identifying the need to expand or diminish activities carried out on behalf of the plan. A NAWMP Science Support Team or "NSST" was established in 2000 as a technical advisory body and consists of three national representatives appointed by the Plan Committee Co-Chairs, one technical representative from each of the joint ventures and flyway councils, representatives from working groups or subcommittees of the NSST, and ad hoc members as required. The specific objectives of the NSST are to:

1. Foster continuous improvement in the effectiveness of plan partners' actions through the establishment of iterative cycles of planning, implementing, and evaluating conservation programs at both the continental and joint venture levels.
2. Conduct large-scale studies of landscape variation and waterfowl demography.
3. Report annually to the Plan Committee and Plan partners on the status of the biological foundations of the plan, evaluation results, and their implications for future conservation activities.

The NAWMP comprises a set of guiding principles and objectives for waterfowl populations and wetland conservation, including restoration, and operated as a joint venture with support from other sectors, including NGOs, in particular Ducks Unlimited (Patterson 1995). The NGOs were seen as critical for maintaining the momentum and supporting the planning and early initiatives, including engaging with the farming communities in areas containing important habitat for waterfowl. At a governmental level, The North American Wetlands Conservation Act 1989 and the funding commitment of the Canadian Government was critical to the initial viability of the plan.

The plan has been updated on a regular basis with input from the waterfowl conservation community. The updates in 1994, 1998, and 2004 supported the goal of securing abundant waterfowl populations through science-based habitat conservation. A major revision to the plan was completed in 2012 with a renewed call to action for the waterfowl conservation community with a vision of "People conserving waterfowl and wetlands" based on adaptive management with clear goals and integrated measurable objectives for populations, habitat, and people (see documents at http://www.nawmprevision.org). It also resulted in the following three goals being agreed:

Goal 1: Maintain long-term average populations of breeding ducks [1955–2014 in traditional survey area (TSA) and 1990–2014 in eastern survey area (ESA)] and periodically, 40 million or more total breeding ducks and 2.7 million or more breeding ducks in the TSA and ESA, respectively.
Goal 2: Increase waterfowl conservation support among various constituencies to at least the levels experienced during the last two decades.
Goal 3: Conserve a habitat system with the capacity to maintain long-term average waterfowl population levels, to periodically support abundant populations, and to consistently support resource users at objective levels.

The revised plan was accompanied by an Action Plan that reflected broad input from the waterfowl management community and provided further details, guidance, and technical direction for the implementation of recommendations and actions. These Actions are expected to evolve as the waterfowl community works towards achieving the above stated conservation goals. The detailed roles of technical committees and administrative bodies in the delivery of the Action Plan are also outlined, with three recommendations being highlighted as priorities for immediate action:

1. Building support for conservation – the need to communicate the value of waterfowl and wetland conservation to the public has emerged as a top tier issue.
2. Development of new NAWMP objectives for waterfowl populations, habitat, and people – the objectives should be clear and measurable, based on the best available science and modeling, and include management actions with associated outcomes, assumptions, and uncertainties.
3. Integrated Waterfowl Management – to ensure the suite of recommendations and existing waterfowl management activities into a coherent and adaptive framework.

The success of the NAWMP depends on the strength of partnerships, called “joint ventures,” involving federal, state, provincial, tribal, and local governments, businesses, conservation organizations, and individual citizens. Joint ventures typically develop implementation plans that focus on areas of concern identified in the plan. There are 18 habitat joint ventures in the United States and 4 in Canada. Two of these have international status, one with boundaries stretching across the Canadian-United States border and one that encompasses areas of the United States and Mexico. Three species joint ventures have also been formed to address monitoring and research needs of specific species or species groups. The species joint ventures are also international in scope.

Despite the successes of the plan, there are still many challenges for waterfowl management, including urbanization and increasing demands for energy and water that influence land use decisions with direct and indirect consequences for waterfowl and other wildlife. Declining numbers of waterfowl hunters, a traditional funding source, is also of concern for maintenance of habitat management and restoration programs. The revision of the NAWMP in 2012 addressed the continuing loss of habitat, management of waterfowl populations, and engaging people who value waterfowl and wetlands.

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Transboundary Wetland Management

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Dave Pritchard

Contents

Abstract

Throughout history, watercourses have offered a convenient geographical demarcation line for defining frontiers between nations. It is no surprise therefore to find numerous examples around the world of wetland systems which straddle these frontiers. Conservation, management and use of these systems thus depends heavily on transboundary cooperation. The Convention on Wetlands (Ramsar Convention) has led much of the attention to this at global level: some of the Convention's key provisions are highlighted in this chapter, along with examples of several other policy, legal and institutional mechanisms of relevance.

Keywords

Ramsar convention · Connectivity · Shared wetlands · Coordination · Migratory species · Transboundary mechanisms

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Introduction

Throughout history, watercourses have offered a convenient geographical demarcation line for defining frontiers between nations. It is no surprise therefore to find numerous examples around the world of wetland systems which straddle these frontiers, each in an ecological sense "belonging" to more than one country. Conservation, management, and use of these systems thus depend heavily on transboundary cooperation. (The term "transboundary" can apply to administrative boundaries within countries as well as between them; but the discussion here refers to its use in the international sense only.)

The mobility of water and migratory animals in particular extends the "ecological" logic of cooperation beyond the borders themselves to other forms of connectedness, such as hydrological catchments and international networks of sites. Further still, the services provided by wetlands (including food, tourism, and carbon storage) are traded in global markets, while flows of knowledge and financial assistance for wetland management criss-cross the world in a growing web of linkages.

The Convention on Wetlands (Ramsar Convention) [*cross-reference*] has been the pre-eminent intergovernmental framework for enabling organized attention to this at global level, and some of its key provisions are highlighted below. Several other policy, legal, or institutional mechanisms are relevant too, and examples of these are mentioned.

International Cooperation is a Fundamental "Pillar" of the Ramsar Convention

The very existence of the Ramsar Convention, as a legal agreement concluded by governments to express their shared desire to achieve common aims for wetlands, is itself a strong manifestation of international cooperation. The text of the treaty, adopted in 1971, refers to the fact that "waterfowl in their seasonal migrations may transcend frontiers and so should be regarded as an international resource" and that "the conservation of wetlands and their flora and fauna can be ensured by combining far-sighted national policies with coordinated international action."

This philosophy is translated into an obligation, accepted by all the Contracting Party governments, to "consult with each other about implementing obligations arising from the Convention especially in the case of a wetland extending over the territories of more than one Contracting Party or where a water system is shared by Contracting Parties" and to "endeavour to coordinate and support present and future policies and regulations concerning the conservation of wetlands and their flora and fauna" (Article 5, one of the so-called "three pillars" of the Convention).

The principal guidance adopted by the conference of parties to assist in implementing Article 5 (annexed to Resolution VII.19, 1999) explains that it is assumed that the requirement to consult refers to all obligations arising from the convention text; that the reference to "wetlands extending over the territories of more than one Contracting Party" refers to wetlands which cross international borders, whether Wetlands of International Importance (as defined and designated under the convention) or not; and that the second clause refers to cooperation “in areas such as shared wetland-dependent species, bilateral and multilateral assistance, trade in wetland-derived plant and animal products, and foreign investment practices”.

Resolution VII.19 calls upon Contracting Parties to give special attention to a number of areas of activity in support of Article 5, such as “identifying shared wetlands, river basins, and wetland-dependent species and supporting initiatives directed at the management of these in cooperation with other Contracting Parties and organizations”.

Shared Wetlands

Ramsar Parties are encouraged to cooperate in the management of all wetlands crossing international boundaries, including those in the coastal zone. Cooperation may take the form of data sharing, coordinated implementation and development of management plans, or formal institutional arrangements for joint oversight.

In the particular case of wetlands designated as Wetlands of International Importance or Ramsar Sites, formal joint arrangements may concern Ramsar Site designation in a coordinated way on both sides (or on all sides, if more than two countries are involved) to address the ecologically unified area in a coherent way.

To date the European region has led the way with this. Around 30 European Ramsar Parties have identified all transboundary wetland systems in their countries, and beginning with a first example in 2001, 15 areas have so far been formally notified as “Transboundary Ramsar Sites.” This title signifies a cooperative management arrangement and helps to raise awareness of the true ecological (rather than geopolitical) nature of the site: it is not a distinct legal status for the Ramsar sites involved. Regular exchanges of experience at regional level on operating these arrangements also take place. Only one non-European example exists at present, in Africa. Transboundary Ramsar Sites may consist of national components that are designated simultaneously and share the same name; or they may be designated at different times with different names: it is coherence of management that is key, not the method of labeling.

Shared River Basins

The principles described above for shared sites are also applicable at the level of river basins and coastal systems that cross international boundaries. According to the United Nations, 145 countries include territory within international basins, and 21 lie entirely within international basins. Cooperation may be necessary to achieve integrated management of the water resources (including groundwater) of an entire catchment area. There may also be situations where a wetland in one country is in a water catchment lying partly in another country, and where upstream actions may have downstream consequences on the other side of a national border. Coastal sites similarly may be affected by adjacent countries’ management of, for example, marine pollution.

Guidance under the Ramsar Convention on Integrated River Basin Management and Integrated Coastal Zone Management urges Contracting Parties to work together in addressing these situations. This may take place at a non-statutory level, with cooperation on technical matters such as monitoring and information exchange. A more formal mechanism used by many countries is the establishment of a joint management authority or commission for a specific river basin, usually with regulatory powers delegated by its member countries. In other cases, cooperation is framed by bilateral or multilateral intergovernmental legal agreements on transboundary watercourses, and there are now more than 300 such agreements in existence around the world.

Shared Species Populations

Wetland animals and plants often also need to be addressed in a transboundary context. The natural range of a species requiring special conservation measures may straddle an international boundary, and any population management actions may therefore need to be coordinated between the countries concerned. Cooperation in respect of border controls may be important in restricting the spread of invasive species or disease-carrying organisms that pose a threat to wetlands.

The most familiar case is probably that of migratory wetland species (including fish, turtles, and aquatic mammals as well as waterbirds). The ecological conservation requirements of these species typically depend on the maintenance of networks of critical wetland habitat distributed throughout their migratory range and on the limiting of any harvesting or hunting to levels that are sustainable for the population as a whole. Setting objectives at a site-by-site level is therefore not enough, and a strategic approach is essential. Many international partnerships, plans and programs now exist for the conservation of shared waterbirds and their habitats, organized according to individual defined migratory routes or “flyways.” The Ramsar Convention Parties in 2008 adopted a comprehensive decision (Resolution X.22) on promoting such flyway cooperation.

Exchange of Know-How and Financial Support

A significant aspect of Article 5 of the Ramsar Convention is its relevance to the role of international flows of development funding and other assistance. In Resolution VII.19, Parties have agreed to raise the level and effectiveness of international development assistance programs directed at the conservation and sustainable use of wetlands and also to ensure that environmental assessments and other safeguards are an integral component of all development projects that affect wetlands, including foreign investments.

Parties are also called upon to share expertise and information and provide training for people involved with wetland conservation and wise use activities. The concept of “twinning” and other forms of networking between Ramsar Sites in different Contracting Parties is encouraged as a way of promoting dialogue and sharing experience.

Ramsar Regional Initiatives

Beginning in the Mediterranean in the early 1990s, and since spreading to around 20 other areas of the world, a mechanism of international cooperation to provide support for improved implementation of the Ramsar Convention has evolved at the regional (sub-global) scale, in the shape of the convention's formal "Regional Initiatives." These are developed by groups of Contracting Parties with a common geographical focus, or in some cases with a common thematic goal, who apply for endorsement from Ramsar's governing bodies to operate as a collaborative entity within the framework of the convention. The initiatives typically comprise either a network for capacity building and cooperation or a "center" for training and capacity building. Each is governed independently but must meet strict requirements for accountability to the convention. Some have received financial assistance from the Ramsar budget, for start-up periods of up to 3 years.

Other Relevant Legal Frameworks

Ramsar is the convention particularly dedicated to wetlands; but other international agreements are also relevant, and Ramsar has formal cooperation arrangements with many of them (this can be thought of as "meta-level" international cooperation).

Reference has been made above to the numerous instruments addressing transboundary watercourses. Two of particular note are administered under the United Nations system: the 1997 UN Convention on the Law of the Non-Navigational Uses of International Watercourses and the 1992 UN Economic Commission for Europe Convention on the Protection and Use of Transboundary Watercourses and International Lakes, also known as the Helsinki Convention or the UNECE Water Convention. Although the latter was initially for European countries, it has since been amended to allow participation by any UN Member State.

In the UNECE region, more than 150 major rivers and 50 large lakes run along or straddle the border between two or more countries. Twenty European countries depend for more than 10% of their water resources on neighboring countries, and for five of these the figure is as high as 75%. The Helsinki Convention, among other things, covers ecologically sustainable management of transboundary surface waters and groundwaters, and its commitment to integrated water resources management replaces an earlier focus on localized sources of pollution and management of separate components of the ecosystem.

In the marine environment, over 140 countries are involved in 13 Regional Seas Programmes established under the UN Environment Programme, and in most cases, these are governed by a specific regional convention established for the purpose. Migratory species conservation is addressed by the Convention on Migratory

Species (CMS), with which Ramsar implements a Joint Work Plan; and the Convention on International Trade in Endangered Species (CITES) specifically combats threats from unsustainable international trade.

Future Challenges

Key future challenges include the following:

- A continuing need to identify and document the transboundary/shared wetland systems of the world
- A need to designate more qualifying wetlands as Transboundary Ramsar Sites, especially outside the European region
- An increasing need to provide financial support from developed countries and international agencies to developing countries to assist in their implementation of the conservation and wise use of wetlands
- A need to negotiate further transboundary agreements for integrated catchment-scale management of wetland resources and strengthen existing agreements, to address inter alia the increasing challenges of water availability in light of human population increases, development pressures, and the impacts of global climate change

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Transboundary Ramsar Site Management: Lake Chad

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Robert J. McInnes

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Abstract

Located between latitude 6° and 24° north and longitude 8° and 24° east, Lake Chad is the fourth largest lake in Africa and represents a vast expanse of freshwater bordering the southern Sahara. The lake is located in the largest closed or endorheic basin on the African continent and covers almost 2.5 million km^2. Lake Chad supports wetlands that are vital for the livelihoods of over 30 million people living within the basin. The successful management of this area depends on transboundary cooperation among seven different countries.

Keywords

Transboundary · Ramsar Site · Livelihoods · International cooperation

Introduction

Located between latitude 6° and 24° north and longitude 8° and 24° east, Lake Chad is the fourth largest lake in Africa and represents a vast expanse of freshwater bordering the southern Sahara. The lake is located in the largest closed or endorheic basin on the African continent and covers almost 2.5 million km^2. However, the majority of this basin is a palaeo-feature corresponding to historical flow regimes.

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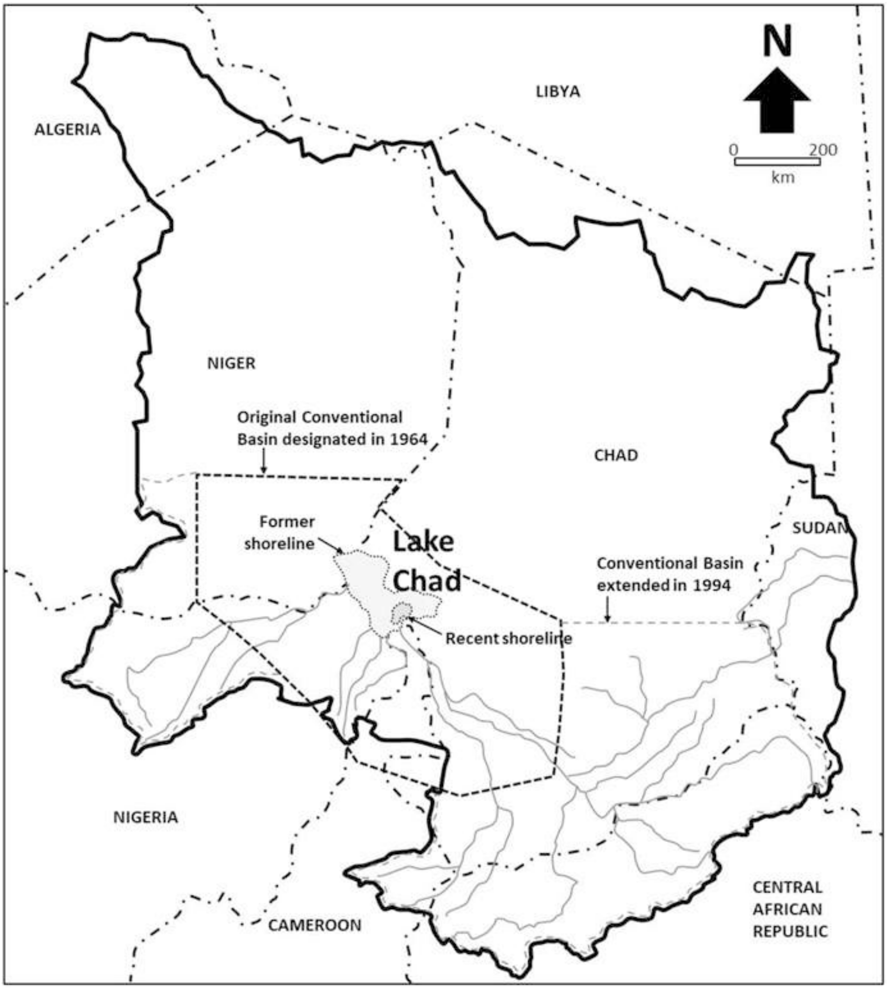

Fig. 1 The Lake Chad drainage and "conventional" basins

The overall Lake Chad basin extends into Algeria, Cameroon, the Central African Republic (CAR), Chad, Libya, Niger, Nigeria, and Sudan (Fig. 1).

The overall drainage basin comprises several subbasins. The main subbasins are those of the Lake Chad itself, Logone (between Cameroon and Chad), a tributary of the Chari (between the Central African Republic (CAR), Chad, and Sudan), Komadugu-Yobe (between Niger and Nigeria), the Borno Drainages (three rivers in Nigeria), and the Northern Diagnostic Basin which covers the largest area but makes a negligible contribution to the hydrological functioning of the Lake.

A study by IUCN (2008) described Lake Chad and its basin wetlands as a unique ecosystem in the region and a reserve of biodiversity of global significance. Like other wetlands, the ecosystem is very productive, performs a variety of functions, provides many products, and delivers various services to society. As examples, the annual contribution of the Lake Chad ecosystems to the economy has been estimated by IUCN (2008) as follows:

- Fishing: US$45.1 million
- Culture's cons season: US$26.6 million
- Breeding: US$14.7 million
- Small-scale irrigation: US$10.8 million
- Large irrigation: US$9.4 million

For millennia, periods of recharge have followed periods of drought, including a total drying out. In 1963, its most recent peak recharge, the lake surface covered approximately 25,000 km^2, but in 1985, it did not exceed 2,500 km^2 reducing to less than 2,000 km^2 in 2010. According to the United Nations' Global Resources Information Database for the environment, the extent of the lake has decreased by almost 95% between 1963 and 1998.

Over the past 30 years, the average annual rainfall within the basin has decreased by approximately 100 mm, and the evidence has been inconclusive as to whether this is a temporary climatic change or a long-term trend. Even without the presence of anthropogenic pressures, there are concerns that the hydrology of the lake could be permanently altered.

Until 1960, Lake Chad was the sixth largest lake in the world. In 1973, it covered an area of 23,000 km^2 (Grove 1996). The volume and area of the Lake decreased between the 1960s and 1990s. The chronological change in the extent of Lake Chad is clearly observed in time-lapse sequence of maps shown in Fig. 2, generated from satellite imagery, illustrating the evolution of the Lake from 1963 to 2001 (Lemoalle 1991; USGS 2001). This is summarized below:

- 1963: Open water is approximately 23,000 km^2 and each basin is connected by open water.
- 1973: The beginning of the effects of the 1972–1974 Sahelian droughts separated the northern pool from the southern by exposure of the Basin's inner ridge, the great barrier which separated an "average" Lake Chad to a "small Lake Chad" (Leblanc et al. 2011). During this period the northern pool suffered a general drying-out period and a change from an open-water lake environment to one of an unstable marshy appearance dominated by aquatic vegetation. The open water had totally dried out by 1975, and during the same period, the open-water areas of the southern pool reduced by 90% (FAO 1986 in Neiland and Béné 2004).
- 1987: The effects of both the 1972–1974 and 1982 Sahelian droughts have resulted in open water being restricted to the southern pool only.
- 1997: There was little variation over the previous decade with open water estimated to vary interannually in area between 1,500 and 2,000 km^2 plus a larger surrounding area of mixed permanent and temporary swampland of between 2,000 and 4,000 km^2.

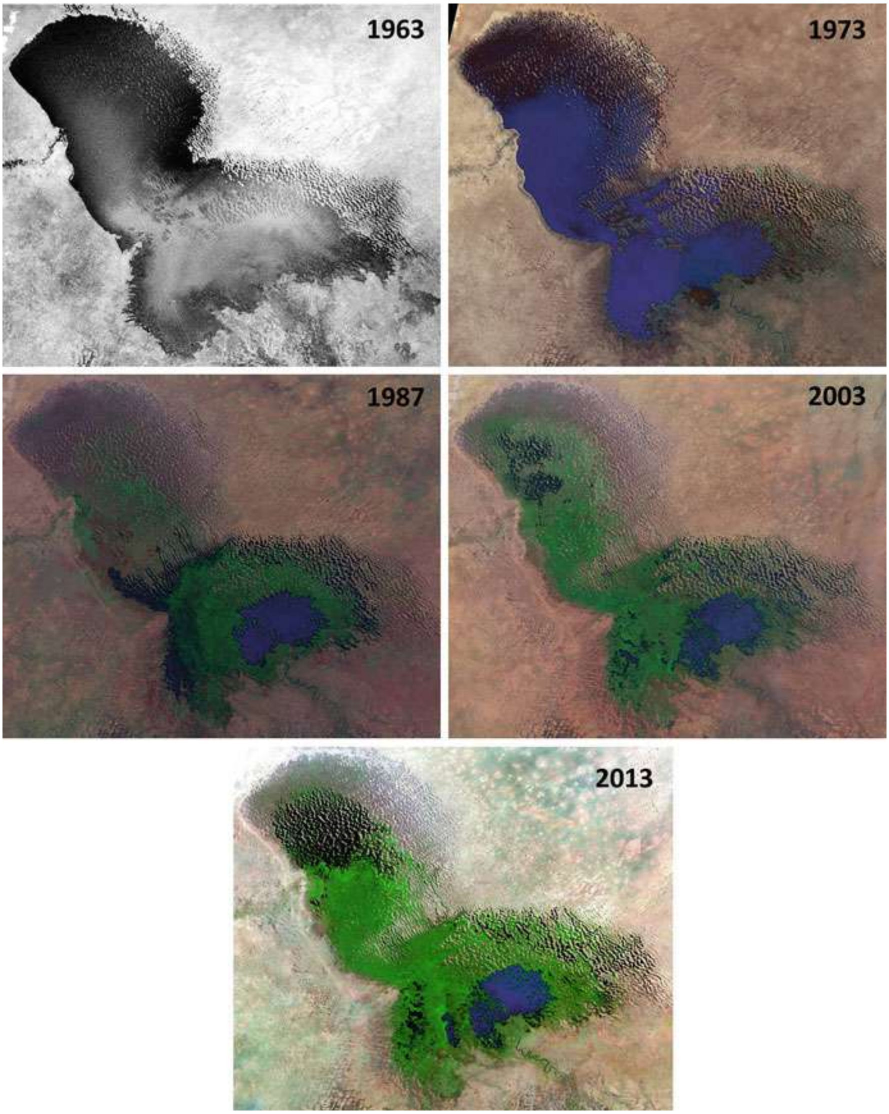

Fig. 2 Aerial images demonstrated the chronological change in the extent and distribution of surface water in Lake Chad (1987-2013) (Source of images: http://earthshots.usgs.gov/earthshots/Lake-Chad-West-Africa#ad-image-4, U.S. Geological Survey)

- 2001: The state of open-water flooding was estimated to be 1,350 km^2 with around 4,000 km^2 of swampland (Neiland and Béné 2004). The expansion in the Lake is masked by the proliferation of floating vegetation.
- 2013: The open water areas in the northern basin has continued to expand and floating vegetation covers extensive areas of surface water across the two basins.

Long-term rainfall records indicate that wetter and drier periods have been experienced in the region since the end of the nineteenth century, with the Sahelian drought being preceded by more than a decade of relatively wet years. Scientific evidence published since 2001 has indicated that river flows, rainfall, and the water levels have shown a small degree of recovery and that the lake may not be disappearing as previously feared. However, the potential implications of this information have had limited impact on the wider consciousness or decision-making processes. Similarly, attempts to model and understand the future trajectory of hydrological change are highly limited.

Given the concerns regarding the potential change to the hydrology of Lake Chad, many interventions and initiatives have been developed to mitigate the degradation of the natural resources of the basin not only to protect the livelihoods of 30 million people who depend on the Lake but also to mitigate the potential source of conflicts among the various stakeholders.

Lake Chad Basin Commission

At the regional level, the Lake Chad Basin Commission (LCBC) is the institution charged with the responsibility for sustainable development and management of water and related resources of the Lake Chad Basin. The aims of the Commission are to regulate and control the use of water and other natural resources in the basin and to initiate, promote, and coordinate natural resource development projects and research. The Commission also promotes mechanisms for settling disputes and enhancing regional cooperation (Bdliya and Bloxom 2007). The LCBC enjoys the status of an international body. The commissioners of the LCBC and the executive secretary enjoy certain diplomatic privileges and immunities accorded to officials of equivalent status of the Organization of African Unity (OAU), now the African Union (AU).

The transboundary management of the lake is the responsibility of the Lake Chad Basin Commission (LCBC), which was established in May 1964 by the Fort Lamy (now N'Djamena) Convention and Statutes by the heads of governments from four riparian countries (Chad, Niger, Nigeria, and Cameroon). The original Conventional Basin covered some 427,300 km^2. In 1994, CAR was admitted as the fifth member state bringing the size of the Conventional Basin to 966,955 km^2. Finally, in July 2000, Sudan was admitted as a member of the Convention increasing the Conventional Basin to 1,035,000 km^2 and effectively including the entire active hydrological basin.

The primary responsibilities of the LCBC are to regulate and control the utilization of water and other natural resources within the basin; to initiate, coordinate, and promote natural resource development projects and research; to examine complaints; and to promote settlement of disputes, thereby contributing to regional conflict resolution and cooperation.

Developing from work conducted by FAO and the World Bank, during the 1980s, the LCBC oversaw diagnostic studies of the Lake Chad Basin and the development of a master plan which was ratified by member states in 1994. Diagnostic studies have continued into the 2000s.

Building on the Africa Water Vision, which was endorsed at an extraordinary summit of the African Union, the LCBC has set out a vision for the Lake Chad Basin in Vision Document 2025. This states that:

> The Lake Chad Region would like to see by the year 2025 the Lake Chad – common heritage – and other wetlands maintained at sustainable levels to ensure the economic security of the freshwater ecosystem resources, sustained biodiversity and aquatic resources of the basin, the use of which should be equitable to serve the needs of the population of the basin thereby reducing the poverty level.

In June 2008 the LCBC member states of Cameroon, CAR, Chad, Niger, and Nigeria agreed to a *Strategic Action Programme* (*SAP*) *for the Lake Chad Basin* which lays down the principles of environmental management and cooperation and establishes an agenda for the delivery of the long-term vision for the sustainable development and environmental stewardship of the Lake Chad Basin.

The Fort Lamy Convention recognizes the sovereign rights of member states over the water resources in the basin while forbidding any unilateral negative exploitation of natural resources where this would impact upon the interest of other states. Additionally, the member states should consult with LCBC on the planning and implementation of projects and should refrain from undertaking activities likely to alter the water balance of the Lake or its water quality or biological resources. The LCBC, through a participatory process, now developed a Water Charter for the basin. The Charter elaborates on commitments and responsibilities of the member states toward the sustainable and equitable management of the water resources of the basin.

Future Challenges

The understanding of the dynamics of Lake Chad is fraught with complications and complexities. Lake Chad is a dynamic system driven by climatic events and modified by human impacts across the watershed. The system is not static but demonstrates variability from year to year and decade to decade.

While the hydrology has been subject to assessment for more than half a century, little attempt has been made to place the current hydrological functioning within the limits of acceptable environmental or social change or within the limits of expected future climatic variability or to fully characterize the ecosystem services provided to human society. The need to understand and manage this sensitive transboundary system will accelerate as growing human demands place more stress of the wetland environment.

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Danube River Basin Regional Management Agreement

65

Robert J. McInnes

Contents

Abstract

Covering more than 800,000 km^2 extending across 19 countries, the Danube River Basin can be considered as the most international river basin in the world. The basin is home to more than 81 million people of multiple countries, cultures, and languages. The International Commission for the Protection of the Danube River (ICPDR) is a transnational body, which has been established to implement the Danube River Protection Convention.

Keywords

Danube · Transboundary · International agreement

Introduction

Covering more than 800,000 km^2 extending across 19 countries, the Danube River Basin can be considered as the most international river basin in the world. The basin is home to more than 81 million people of multiple countries, cultures, and languages.

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The International Commission for the Protection of the Danube River (ICPDR) is a transnational body, which has been established to implement the Danube River Protection Convention. The legal basis of ICPDR is the Convention on Cooperation for the Protection and Sustainable Use of the Danube River (or more informally the Danube River Protection Convention or DRPC). This convention commits the contracting parties to join their efforts in sustainable water management, including conservation of surface and groundwater, reducing pollution, and the prevention and control of floods, accidents, and ice hazards. The convention was signed in Sofia in 1994 and came into force in October 1998.

History of Cooperation Across the Danube River Basin

Before World War II, the European Commission of the Danube, which dated back to the 1856 Treaty of Paris, comprising representatives from each of the riparian countries, was responsible for administration of the Danube River (Sommerwerk et al. 2009). Its primary goal was to ensure free navigation along the Danube for all European countries. The new political alliances manifested as a result of World War II gave rise to a new approach to management of the river basin. At a 1948 conference in Belgrade, the states from the Eastern Bloc shifted control over navigation to the exclusive control of each riparian country. A further outcome of the Belgrade conference was to bestow upon the Commission semi-legislative powers regarding navigation and inspection.

By the mid-1980s, it had become clear that issues other than navigation were gaining in importance within the Danube basin and most notably were problems regarding water quality. Recognizing the increasing degradation of water quality, in 1985 the (at the time) eight riparian states of the Danube signed the "Declaration of the Danube Countries to Cooperate on Questions Concerning the Water Management of the Danube" (or informally, the Bucharest Declaration).

The Bucharest Declaration reinforced the principle that the environmental quality of the river was intrinsically linked to the overall environment of the basin and as such committed the countries to adopt an integrated approach to water management. One of the first steps toward implementation of this integrated approach was the establishment of a transnational unified monitoring network. The changing geopolitics of the region post-1989 had implications for the Danube River Basin (Linnerooth-Bayer and Murcott 1996). Basin-wide coordination was strengthened at meetings in Sofia, Bulgaria, in September 1991. The countries and interested international institutions established an initiative to support and reinforce national actions for the restoration and protection of the Danube River. Through this initiative, the Environmental Programme for the Danube River Basin (EPDRB), the participants agreed that each riparian state would:

- Adopt the same monitoring systems for assessing environmental impacts
- Address the issue of liability for cross-border pollution
- Define rules for the protection of wetland habitats

- Define guidelines for development to conserve areas of ecological importance or aesthetic value

An interim task force was established to coordinate the efforts. Its work was supported by a Programme Coordination Unit (PCU) based in Vienna, Austria. One of the major tasks of the EPDRB was the development of the Strategic Action Plan (SAP). In order to transition from planning to implementation, the proposed SAP addressed the following concerns:

- Measures must be "concrete" and achieve results in short term.
- Major environmental threats must be addressed with realistic costed actions and constraints to problem-solving.
- The SAP should be updated regularly.
- Wide consultation during its preparation is desirable, in particular with parties who would be responsible for its implementation.

As a commitment within the SAP, during late 1993 and early 1994, the riparian countries developed the Convention on Cooperation for the Protection and Sustainable Use of the River Danube. The resultant "Danube River Protection Convention" was finally signed in Sofia on June 29, 1994. The International Commission for the Protection of the Danube River (ICPDR) became active in 1998. Since then, it has grown into one of the largest and most active international bodies of river basin management expertise in Europe dealing with not just the main Danube River but also with the whole Danube River Basin, which includes more than 300 tributaries and the groundwater resources.

How Is the Commission Managed?

The ICPDR is an international organization with the following 15 contracting parties:

- Austria
- Bosnia and Herzegovina
- Bulgaria
- Croatia
- Czech Republic
- Germany
- Hungary
- Moldova
- Montenegro
- Romania
- Slovakia
- Slovenia

- Serbia
- Ukraine
- European Union

The ICPDR meets twice a year: The Ordinary Meeting is held at the Permanent Secretariat in Vienna in December; another meeting of Heads of Delegations is held in June in the home country of the serving president. The meetings comprise of delegations of contracting parties and observer organizations. Every contracting party has one Head of Delegation representing the country. Consensus is sought for all decisions. The meetings are chaired by the ICPDR president; ICPDR presidency is passed on from one country to another in an alphabetical order every year.

In addition, much of the work of the ICPDR is done by expert groups (EGs), which comprise panels of specialists from the ICPDR contracting parties and observers, often civil servants of the relevant ministries, in some cases employees of nongovernmental organizations (NGOs) or contracted agencies. There are seven permanent expert groups and one ad hoc EG. The EGs all have terms of reference and mandates adopted by the Commission. The EGs usually meet up to three times a year.

The ICPDR is primarily not a project-implementing body, but it can take responsibility for some projects in order to achieve its core objectives. However, the ICPDR can act as an implementing entity for some project, and for others it acts as an advisor or as a facilitator for the implementation of different components through its EGs. The ICPDR also acts as a forum for the development of projects that are submitted and carried out by others (but benefiting the cooperation under the ICPDR).

The ICPDR is financed through the contributions from the contracting parties. According to the DRPC, the contracting parties (except for the EU) shall contribute an equal share, unless unanimously decided otherwise by the ICPDR. Some exceptions have been applied for transitional periods. The total annual budget of the ICPDR is in the region of one million Euros. Much of the ICPDR's work is undertaken directly by member countries, and as such these are considered in-kind contributions and are not shown in the ICPDR budget.

By acceding to the convention, contracting parties commit under international law to specific actions and to uphold certain principles. In past conflicts, the ICPDR contributed to the harmonization of efforts by providing a forum for discussion. The president or the staff at the permanent secretariat can initiate dialogue on specific issues and contribute to building consensus among contracting parties. The convention also provides a more formal dispute settlement mechanism, but in practice this has not been needed so far, as the countries concerned have achieved consensus on issues of conflict through dialogue and discussion.

In 2000, the contracting parties nominated the ICPDR as the platform for the implementation of all transboundary aspects of the European Union (EU) Water Framework Directive (WFD). The work for the successful implementation of the EU WFD is therefore high on the political agenda of the countries of the Danube River Basin District. In 2007, the ICPDR also took responsibility for coordinating the implementation of the EU Floods Directive in the Danube River Basin.

Table 1 Basic information on the countries in the Danube River Basin (DRB) (data from Danube Basin Analysis, 2005)

Country	Coverage in DRB (km^2)	Percentage of DRB (%)	Percentage of DRB in country (%)	Population in DRB (mio.)
Albania	126	<0.1	0.01	<0.01
Austria[a]	80,423	10.0	96.1	7.7
Bosnia and Herzegovina[a]	36,636	4.6	74.9	2.9
Bulgaria[a]	47,413	5.9	43.0	3.5
Croatia[a]	34,965	4.4	62.5	3.1
Czech Republic[a]	21,688	2.9	27.5	2.8
Germany[a]	56,184	7.0	16.8	9.4
Hungary[a]	93,030	11.6	100.0	10.1
Italy	565	<0.1	0.2	0.02
Macedonia	109	<0.1	0.2	<0.01
Moldova[a]	12,834	1.6	35.6	1.1
Montenegro[a]	7,075	0.9	51.2	0.2
Poland	430	<0.1	0.1	0.04
Romania[a]	232,193	29.0	97.4	21.7
Serbia[a]	81,560	10.2	92.3	7.5
Slovak Republic[a]	47,084	5.9	96.0	5.2
Slovenia[a]	16,422	2.0	81.0	1.7
Switzerland	1,809	0.2	4.3	0.02
Ukraine[a]	30,520	3.8	5.4	2.7
Total	801,463	100		81.00

(NB: [a]ICPDR contracting parties)

The Danube River Basin

The Danube River Basin covers a total area of 801,463 km^2 and is the second largest watershed in Europe. The Danube River stretches 2,780 km from Germany's Black Forest to the Danube Delta in the Black Sea. Characteristically, the basin is divided into three areas: the upper, middle, and lower basins. The upper basin extends from the source of the Danube in Germany to Bratislava, Slovakia. The middle basin is the largest of the three areas, extending to the dams of the Iron Gate gorge on the border between Serbia and Romania. The lowlands, plateaus and mountains of Romania and Bulgaria form the lower basin. Finally, the river divides into the three main branches of the Danube Delta, which covers an area of about 6,750 km^2, before entering the Black Sea.

The Danube River flows through 19 countries and is a truly transboundary, international river system. Table 1 provides further information on the countries of the Danube River Basin.

Future Challenges

The transboundary nature of the Danube River Basin management is challenging due to a lack of on-site expert knowledge, high administrative and socioeconomic complexity, the need to balance various stakeholder interests, and difficulties enforcing international and national law. Therefore, in the future, the need for an efficient "science–policy interface" is vital for the successful development and implementation of ambitions of the ICPDR (Sommerwerk et al. 2010). The various human activities continue to place environmental pressures on the Danube and its network of tributaries. Industry, agriculture, hydropower, and tourism are all economically important and depend on the Danube as a resource, yet at the same time they also threaten river ecosystems. The ICPDR continues to seek to address a range of issues and challenges through collective working and the implementation of best practice in transnational river basin management.

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Indus Waters Treaty

66

Nick C. Davidson

Contents

Abstract

The Indus River is among the longest rivers in Asia and the transboundary Indus River Basin is one of the largest in the region. Rising in the Tibetan Plateau of the Himalayas, the Indus flows through Afghanistan, China, India, and Pakistan, discharging through the Indus Delta into the Arabian Sea near Karachi. The largest areas of the Basin lie in Pakistan and India. The basin faces severe water stress. Increasing conflict in the 1950s over sharing the Basin's water s between India and Pakistan led to the two countries signing the Indus Waters Treaty in 1960. The Treaty is recognized as one of the most successful transboundary water-sharing mechanisms in the world.

Keywords

Indus · River basin · Treaty

The Indus River Basin

The Indus River is among the longest rivers in Asia, at approximately 3,180 km long, and the transboundary Indus River Basin is one of the largest in the region, with a catchment area of approximately 1.12 million km^2. Rising in the Tibetan Plateau of

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the Himalayas, the Indus flows through Afghanistan, China, India, and Pakistan, discharging through the Indus Delta into the Arabian Sea near Karachi (Fig. 1). The largest areas of the Basin lie in Pakistan (520,000 km^2; 65% of Pakistan's land area) and India (440,000 km^2; 14% of India's land area) (FAO 2011).

The lower reaches of the Basin are densely populated, and up to 300 million people have been estimated to live in the Basin, many of whom depend on the river and its resources. The water resources of the Indus are vital for agriculture and food production in Pakistan. Water infrastructure developments in the twentieth century have created an extensive network of dams, reservoirs, and irrigation canals which now deliver water to about 110,000 km^2 of irrigated agricultural land – the largest irrigated area of any river basin (FAO 2011).

However, the Indus Basin has been recognized as facing increasingly severe water stress. Wong et al. (2007) ranked the Indus among the world's top ten rivers at risk. The highest risk comes from climate change since the river flow largely

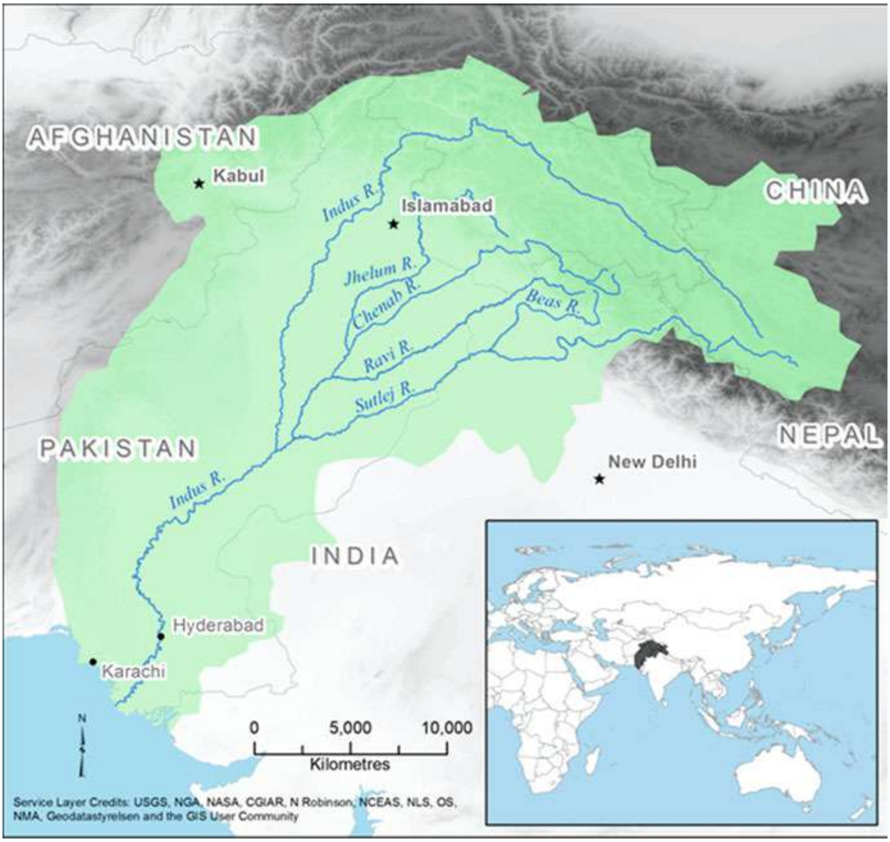

Fig. 1 Location of the Indus River Basin (© Spatial Data Analysis Network, Charles Sturt University)

comes from glacial meltwater, but also with water abstraction, agricultural pollution, and water infrastructure, including dam construction, contributing to the pressure on the river system. Similarly, the World Resources Institute (Gassert et al. 2013) assessed the Indus River Basin as having an extremely high “baseline water stress” (the ratio of total annual water withdrawals to total available renewable water supply) and with a stress score of 4.3 out of 5.0, the sixth highest risk score among the world’s major river basins.

The Indus Waters Treaty

Following the partition of the British Indian Empire in 1947, which meant that the upper tributaries of the Indus were then in India with the downstream river in Pakistan, there were increasing concerns and conflict during the 1950s over the sharing of the Indus’ waters between the two countries. There was an initial, interim agreement on allocations of Indus water – the Inter-Dominion Agreement of 4 May 1948 (http://www.internationalwaterlaw.org/documents/regionaldocs/punjab-canal.html) – through which India would release water downstream to Pakistan in return for annual payments by Pakistan. Negotiations between India and Pakistan for a more permanent agreement then stalled (World Bank undated).

Interventions by the International Bank for Reconstruction and Development (now the World Bank) led to the resumption of negotiations, on both technical and political aspects of such an agreement, and eventually led to the signing by the Indian prime minister and Pakistan president in 1960 of the Indus Waters Treaty on the sharing by the two countries of Indus water (World Bank undated). This agreement has stood the test of time and is recognized as one of the more successful transboundary water-sharing mechanisms in the world.

Despite its title, the treaty focuses on the partitioning between India and Pakistan on use of the tributary rivers of the Indus, rather than the sharing of Indus water per se. Under the treaty, the eastern rivers (Ravi, Beas, and Sutlej) are allocated to exclusive use by India, and the western rivers (Jhelum, Chenab, and Indus) allocated to exclusive use by Pakistan. An initial transition period of 10 years was agreed for use of water from the eastern rivers, during which India was required to supply water to Pakistan, pending Pakistan’s completion of canal infrastructure for the western rivers, and during which Pakistan received a one-off financial compensation payment for the loss of access to water from the eastern rivers. The full text of the treaty is available on: http://siteresources.worldbank.org/INTSOUTHASIA/Resources/223497-1105737253588/IndusWatersTreaty1960.pdf.

The treaty established a Permanent Indus Commission, with a commissioner appointed from each of India and Pakistan, and through which the agreed exchange of data and cooperation is facilitated. The commission provides the mechanisms for consultation and conflict resolution concerning the Indus. Under the treaty, the commission meets to discuss potential disputes and to facilitate cooperation on any future development of the water infrastructure of the Indus Basin. Each party (India, Pakistan) to the treaty must notify the other of any water engineering work plans which could affect the other party. When there is disagreement over such plans, the commission can bring in an independent expert to mediate and arbitrate on the issue.

Despite the successes of the Indus Rivers Treaty over the past 45 years, recent concerns have been expressed by Pakistan over the impact on downstream water resources of dam construction in the Indian portion of the Basin, and it is recognized that the treaty does not yet adequately address issues of the impacts of the changing climate on water availability in the Basin; this issue is recognized by Wong et al. (2007) as the major pressure on the Basin. A more extreme and volatile monsoon season, which led to devastating Indus Basin floods in 2010 and 2011, has also been attributed to the changing climate.

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Mekong River Basin Regional Legal Framework

67

Huynh Tien Dung

Contents

Abstract

The Mekong River is a transboundary river in Southeast Asia draining a basin covering approximately 800,000 km^2. At an estimated length of 4,350 km, it is the world's 12th longest river and the 7th longest in Asia. The river rises on the Tibetan Plateau and runs through China's Yunnan province, Myanmar, Lao People's Democratic Republic (PDR), Thailand, Cambodia, and Vietnam where it discharges into the South China Sea. The origins of transnational and transboundary management of the main channel and its tributaries date from the late 1950s, culminating in 1995 with the signing of the Mekong Agreement by Thailand, Lao PDR, Vietnam, and Cambodia. The agreement codified principles of regional cooperation and established the Mekong River Commission (MRC). Under the agreement, MRC member countries agree to cooperate in all fields of sustainable development and in the utilization, management, and conservation of water and related resources in the Mekong River Basin. The MRC has a three-tiered institutional structure comprising a Ministerial Council, a Joint Committee, and a Secretariat.

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Keywords
Mekong River · International agreement · Transboundary management · Transnational cooperation

Introduction

The Mekong River is a transboundary river in Southeast Asia draining a basin covering approximately 800,000 km^2. At an estimated length of 4,350 km, it is the world's 12th-longest river and the 7th longest in Asia (Mekong River Commission 2003). The river rises on the Tibetan Plateau and runs through China's Yunnan province, Myanmar, Lao People's Democratic Republic (PDR), Thailand, Cambodia, and Vietnam where it discharges into the South China Sea. The basin is characteristically divided into the Lower and Upper Basins. The Upper Basin comprises approximately 24% of the total basin area and contributes 15–20% of the water that flows into the Mekong River. The tributary river systems in the Upper Basin are small; however this changes as the floodplain becomes wider and the tributaries drain larger subbasins in the Lower Basin. The Mekong drainage network is extremely complex, and across the entire basin, its management affects the lives of approximately 60 million people (Fig. 1).

A Brief History of Transboundary Management

Going back to the 1950s, the historic governance of the Mekong has been dominated by the desire to collect a variety of hydrologic data in order to manage the Mekong into a "working" river (Sneddon and Fox 2006). Three principal goals were pursued in the 1950s, namely, the generation of hydroelectricity to develop industrial development, the storage of water in order to facilitate the expansion of irrigated agriculture, and the control of annual flood risks.

The origin of the transnational and transboundary management of the main channel and its tributaries can be traced to the late 1950s. The Committee for the Coordination of Investigations in the Lower Mekong Basin, or "Mekong Committee," was created in 1957 under the guidance of the United Nations Development Programme (UNDP). The Committee's 1957 statute mandated that it "promote, coordinate, supervise, and control water resource development projects in the lower Mekong basin." Following its inception, and over the next 15 years or so, the Mekong Committee established a network for the collection of hydrologic data. The primary use of these data was to assess the feasibility of constructing large hydroelectric generating dams on the main river and the overseeing of tributary projects within the national territories of the Lower Basin's states (Lao PDR, Thailand, Cambodia, and Vietnam).

The Mekong Committee continued to negotiate over transnational water management issues throughout the period of the Vietnam-America war, during which time South Vietnam and Lao PDR maintained a generally cooperative relationship with Thailand. However, the war effectively placed wider development of initiatives on hold, and many of the planned projects were never put into effect.

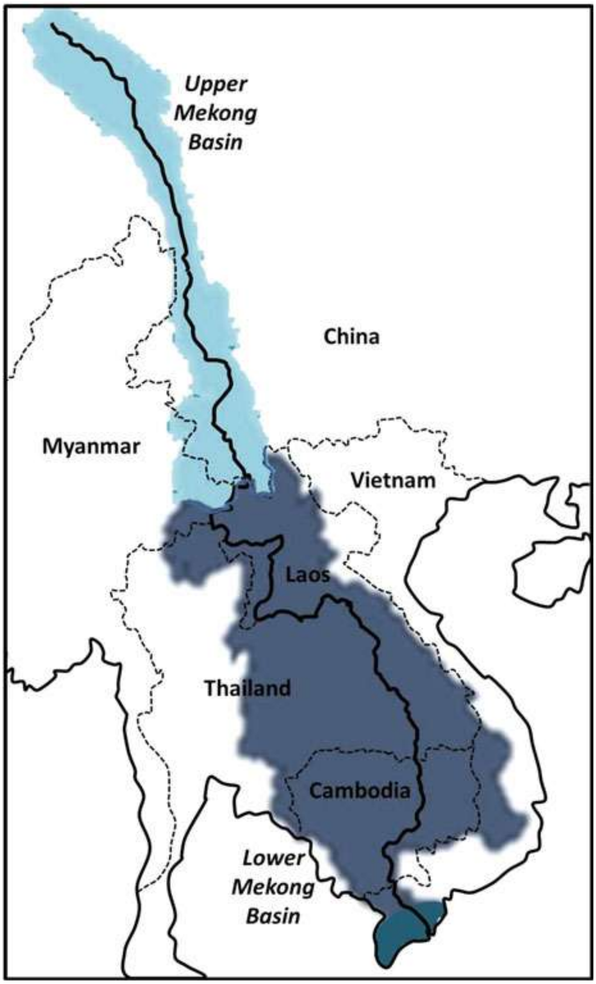

Fig. 1 The Mekong River Basin

In 1975, under the auspices of the Mekong Committee, the “Joint Declaration of Principles for the Utilization of the Waters of the Lower Mekong Basin” was signed. This Joint Declaration stressed the “resources of common interest” and effectively gave the riparian states a veto power over plans by other nations to divert water from the main river. However, implementation of the 1975 Declaration was impeded by ongoing conflict and political reform within the riparian states. Vietnam, Cambodia, and Lao PDR all underwent changes in political regime following the end of the Vietnam-America war. As a result of the prevailing geopolitics in the Basin, the Mekong Committee failed to meet in 1976 and 1977, and it appeared that the initiative would fail.

However, in 1978, Thailand, Lao PDR, and Vietnam established an interim body (called the Interim Mekong Committee) to encourage and facilitate the continuation of dialogue. Cambodia, which by now was under the rule of the Khmer Rouge, failed to participate in the Interim Committee, thus limiting the capacity of the committee to fulfill planning and management objectives at a basin-wide scale.

Through the Interim Mekong Committee, partial collaboration continued until 1991 when Cambodia resumed negotiations with the other Lower Basin countries. However, difficulties were encountered in reaching agreement on a draft Mekong Committee Declaration. These difficulties were primarily the result of a disagreement between Vietnam and Thailand over the diversion of water by the upstream state as part of the planned Khong-Chi-Mun interbasin transfer project in Northeast Thailand. Thailand demanded a review of the 1975 Joint Declaration, and the principles of which had guided cooperation in the Interim Committee to date.

In the early 1990s, the United Nations Development Programme (UNDP) intervened, establishing a working group to investigate future collaboration in the Lower Basin. Working group meetings involving the riparian countries continued through 1993 and 1994. The outcome of these meetings was the draft *Agreement on the Cooperation for the Sustainable Development of the Mekong River Basin* (the Mekong Agreement).

The 1995 Mekong Agreement and the Mekong River Commission

On 5 April 1995, the Mekong Agreement was signed by Thailand, Lao PDR, Vietnam, and Cambodia. The Agreement codified principles of regional cooperation and established the Mekong River Commission (MRC) (http://www.mrcmekong.org/). Under the Agreement, MRC member countries agree to cooperate in all fields of sustainable development and in the utilization, management, and conservation of water and related resources in the Mekong River Basin, for example, navigation, flood control, fisheries, agriculture, hydropower, and environmental protection. This approach to transboundary management has been lauded as the most progressive of institutional frameworks for the governance of an international watercourse but also encounters many challenges.

Under the auspices of the Mekong Agreement, the MRC has a three-tiered institutional structure comprising a Ministerial Council, a Joint Committee, and a Secretariat. The Council is the highest body within the organization and is responsible for overseeing MRC activities and directing MRC policies. The Joint Committee is responsible for implementing Council initiatives and supervising the activities of the Secretariat.

The Secretariat, which is based in Vientiane, Lao PDR, is responsible for the day-to-day administration of MRC affairs and for the development and implementation of a variety of programs. In order to assure neutrality, the Chief Executive Officer of the Secretariat is a citizen of a non-riparian state. Representative parity for each riparian state is practiced within the secretariat in order to employ equal numbers of staff from each member state. A key MRC policy, which enables ongoing capacity building in river basin management, is to limit staff tenure to a maximum of 6 years.

Future Challenges

While many elements of the transboundary management of the Mekong River have been successful, the future is not without challenges. At the Mekong Summit in 2010, the prime ministers from the four MRC countries decreed that the organization would need to move from a position of almost total dependency on external funding to a position of self-financing by 2030. A suggestion was to move toward decentralization and a greater level of national implementation (MRC 2012). Challenges also remain regarding the hydropolitics of environmental conflicts. It has been argued that the major challenges facing the transboundary management of the Mekong River are not necessarily among riparian states, but rather they will involve conflicts between state agents (such as dam-building agencies, power generation departments, and irrigation organizations) and non-state actors (such as civil society groups and local communities) over specific interventions and alterations of the river environment (Sneddon and Fox 2006).

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Murray-Darling Basin: Conservation and Law

68

Jamie Pittock

Contents

Abstract

Better wetland conservation law can be informed by lessons from Australia's Murray–Darling Basin. The legal character of water entitlements is critical for ensuring that water is available to adequately sustain wetlands. Better management has been informed by national harmonization of water data collection and providing public access to this information. An independent statutory manager of environmental water in the Federal Government has ensured that environmental water is protected are used to conserve wetlands. Domestic law has been considerably strengthened by drawing on international treaties, especially the Ramsar Convention on Wetlands. Overlapping roles of federal and state governments have hindered some conservation initiatives but have also ensured some level of wetland conservation continues at one level of government when the other level of government does not do so. As direct government action has become more financially and politically constrained, businesses, community organizations, and Indigenous peoples have been enabled to play greater roles in conservation of

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wetlands. This broader approach to wetlands governance has generated more innovative approaches and stakeholder support for wetland conservation.

Keywords
Australia · Business sector · Conservation reserves · Environmental law · Federal government · Indigenous peoples · Murray-Darling Basin · Non-government organizations · Ramsar Convention on Wetlands · River basin management · Water entitlements · Water law · water markets · Wetlands conservation

Introduction

Conservation of wetlands in the Murray-Darling Basin has been advanced by the evolution of environmental and water laws that have been increasingly influenced by international environmental agreements. The basin occupies a seventh of the Australian continent, nearly a tenth of this area; some 5.7 million hectares are regarded as wetlands (Kingsford et al. 2004).

The basin largely lies in the Mediterranean temperate zone and is especially prone to water scarcity, extreme events, and climate change, requiring the development of institutions for managing great hydrological variability that may hold lessons for other areas of the world (Grafton et al. 2012). As Australia has a federal system of government, lessons for wetlands and river basin governance in the basin are likely to be particularly applicable in the 28 federations globally (Garrick et al. 2013). In this chapter the history of water-related laws is outlined before focusing on recent wetland conservation initiatives, including the influence of international agreements.

Indigenous Laws and Institutions

Prior to European occupation, the wetlands of the basin were focal points in the territories and livelihoods of several dozen Aboriginal nations. One example is the Willandra Lakes World Heritage Area that is shared by three Indigenous nations and holds some of the oldest remains of human occupation of Australia (Australian Government 2002). Institutions existed that governed the extensive and sustainable use of wetland resources that are indicated by such sites as a canal to enable eel migration across catchment and permanent fish trap infrastructure (Bandler 1995; Lintermans 2004).

While the Indigenous nations were substantially dispossessed in the basin, the 1992 decision to recognize that native title remains where unbroken cultural and economic use of lands can be demonstrated, and the increasing return to and purchase of land by Aboriginal communities means that a modest but growing portion of the basin's wetlands is under Indigenous management (as detailed below). In 2004 the federal and state governments agreed in the National Water Initiative to recognize the Indigenous people's rights to water for cultural and economic purposes (Commonwealth of Australia et al. 2004; Jackson and Morrison 2007). Although a First People's Water Engagement Council was formed to advise on implementation (First People's Water Engagement Council 2012), the definitions of cultural water needs and mechanisms to give these effects remain contested (Weir 2011).

Early European Water Laws and Institutions

The early European occupiers of Australia quickly realized that the extreme hydrological variability of the basin required new kinds of water laws. Deliberately rejecting the Western United States prior appropriation model, the Australian colonies – later states – largely adopted water use entitlement that is an annual share of the available resource (Connell 2007). When Australia federated, the management of natural resources was largely left as the responsibilities of the states, despite an ambiguous clause in the constitution preventing the Commonwealth Government from unreasonably abridging the rights of the states to the conservation (utilization) of water. The debates over sustainable management of the River Murray's waters in the federation conventions continue to this day (Connell 2007).

In 1915 the three lower basin states and Commonwealth Government formed a River Murray Commission to undertake the development of infrastructure for shipping – even then largely superseded – and irrigated agriculture. Decades later the increasing degradation of the basin's rivers and other wetlands due to water diversions was expressed through rising salinity levels, and in 1991, a 2,000 km long, poisonous cyanobacteria bloom (Bowling and Baker 1996). This was the catalyst for the Murray-Darling Basin Agreement, formation of the Basin Commission and subsidiary natural resource management programs, institutions that were based on consensus between all six states, territory, and federal governments (Connell 2007). However these initiatives failed to stem the ecological decline of the basin's wetlands, and severe drought prompted further reform in 2007–2008 (Grafton et al. 2014).

Wetland Conservation Laws and Institutions

Following the 1983 dispute over the Tasmanian state government's plans to dam the Franklin River, the federal government began using its constitutional powers to legislate to implement international agreements and to regulate trading corporations to conserve the environment (Fisher 2003). While Australia ratified the 1971 Ramsar Convention on Wetlands and nominated many sites in the basin to the Register of Wetlands of International Importance, initially they had no effective protection in domestic law. Similarly a number of migratory species agreements signaled intent but not domestic legal protection. The 1992 Convention on Biological Diversity was translated into laws at the federal level and a number of states that enabled the evaluation and listing of threatened species and also ecological communities, which then required the preparation of recovery plans (State of the Environment Committee 2011). Many aquatic species and some ecological communities have been listed for legal protection and conservation. Threatening processes can also be nominated and listed under federal law, requiring the protection of threat abatement plans, although this mechanism has been ineffectual to date.

More effective has been the National Reserve System and Indigenous Protected Areas programs (Ross et al. 2009; State of the Environment Committee 2011), which from the late 1990s saw large areas of wetlands acquired and designated for conservation, for instance, in the Paroo River catchment. The National Reserve System program provided federal government matching funding to state government agencies and nongovernment conservation land trust groups to acquire lands containing underrepresented ecosystems for conservation reserves. For instance, Bush Heritage Australia acquired the 14,400 ha property Naree Station to conserve significant wetlands (BHA 2014).

In 1999 the Commonwealth Environment Protection and Biodiversity Conservation Act (EPBC Act) was adopted, which now names nine “matters of national environmental significance” where the federal government requires proponents of “new actions” that may significantly impact on one or more of these matters to undergo environmental assessment and approval (Australian Government 1999). This ensures a more objective assessment of proposed actions compared with those of the pro-development state governments. The EPBC Act codified many aspects of the Ramsar Convention on Wetlands in domestic law. The requirement for environmental impact assessment of new developments extends beyond the Ramsar site boundary to include actions that may significantly impact upon its ecological character, such as any new, upstream water diversions. These legal ramifications prompted the Commonwealth Government to fund proper mapping of Ramsar site boundaries, defining the ecological character of each site, as well as preparing management plans. Despite at least one outdated critique (Farrier and Tucker 2000; Blasco 2001), Ramsar listing has resulted in a renewed focus on providing modest environmental flows to the designated wetlands in the basin, including in the Basin Plan (Gardner 2012).

However, the EPBC Act also generated risk-adverse responses from the governments that has diminished potential wetland conservation activities (Pittock et al. 2010). Concern over federal government regulation of Ramsar sites has seen state governments designate fewer wetlands, exhibiting a misplaced fear since migratory and threatened species, among other matters, already enable federal regulation of new developments in most significant wetlands in Australia. After early conflicts with the states and nongovernment litigators, the Commonwealth Government administrators have also ceased to designate new Ramsar sites unless a proposal has a state government agreement, a surveyed site boundary, an ecological character description, a management plan, and an environmental flow agreement (SEWPAC 2012). In the absence of strong public support for site conservation, these exhaustive bureaucratic requirements have stalled new wetland conservation measures even though the Commonwealth Government could choose to override any state government opposition.

The EPBC Act did not legislate to implement a number of other national obligations under the Ramsar Convention, for example, establishment of a national wetlands advisory committee, nor an indicative list of sites that should be considered for Ramsar designation. Despite the EPBC and Water acts, the ecological character of a great many of the listed wetlands continues to decline (Pittock et al. 2010). While the Australian Government has reported a few of these cases to the Ramsar Secretariat as required by Article 3.2, many have not been reported (Pittock et al. 2010; Ramsar 2009). The Australian Government has also unilaterally decided not to report changes in ecological character due to climate change while at the same time funding an extensive program of "environmental works and measures" – major infrastructure intended to conserve wetland biodiversity with less water that has been questioned by academics (DEWHA 2009; Pittock et al. 2012).

The Water Act and the Basin Plan

In 2007–2008 at the peak of the Millennium Drought, the Commonwealth Government gained grudging consent from the state governments for it to regulate water management in the basin. The resulting Water Act requires the Commonwealth Government to set "sustainable diversion limits" based on best available science in a Basin Plan that is to be revised at least every 10 years (Commonwealth of Australia 2008). In large part the Water Act derives its constitutional mandate from implementation of the Ramsar Convention, migratory species agreements, and the Convention on Biological Diversity (Pittock et al. 2010). This quickly sparked a political debate as to whether the Water Act required environmental sustainability first and then optimization of economic and social welfare (a reasonable reading of the law) or whether the three objectives should be balanced, as interpreted by subsequent governments. Obtaining the consent of all the state governments to the Water Act has required the Commonwealth Government to grant more than ten billion Australian dollars, often spent in ways that economists regard as poorly targeted (Grafton 2011), as well as delaying full implementation of the Basin Plan to 2019 (Pittock 2013).

The new Murray-Darling Basin Authority has undertaken an analysis of water required to conserve wetlands throughout the system, and while it is proposed to reallocate up to 27% of consumptive water to wetland conservation in the Basin Plan (Commonwealth of Australia 2012), it is questionable where this is a sufficient volume of water. It is notable that a number of key wetland conservation issues are yet to be resolved, including the representativeness of the wetlands being watered, allocation of water to wetlands in dry years under existing state government operating rules and adaptation to climate change (Grafton et al. 2014; Pittock 2013).

The Water Act has other benefits for wetland conservation, including better data and environmental water management. Different state water accounting systems have been harmonized under the national Bureau of Meteorology, providing comparable data for better management (Commonwealth of Australia 2008; BoM and ABS 2011). The Water Act also establishes a Commonwealth Environmental Water

Holder (CEWH) to hold and manage the water entitlements acquired for the environment by the federal government through purchase and funding water efficiency savings (Connell 2011). In the past, environmental ("rule-based") water was that left after consumptive entitlements had been allocated and this diminished significantly in dry years. With the purchase of entitlements, the water held by the CEWH ("water entitlements ") – which may eventually amount to a quarter of the environmental water – has the same legal characteristics as the water entitlements held by farmers and is less easily politically manipulated.

Increasing Influence of Nongovernment Organizations

Budget cuts and the greater politicization of water management and wetland conservation have seen governments designate fewer wetland reserves in the past decade. Instead there has been a rise in involvement of business, environmental, and Indigenous organizations in wetland governance. Businesses wanting to demonstrate their sustainability practices have undertaken some innovative wetland restoration and conservation programs, including the major wineries Banrock Station and Chateau Tahbilk. After a protracted process of gaining government concurrence, five floodplain graziers and Banrock Station had portions of their wetlands designated as Ramsar sites, partly in an effort to pressure the governments to restore adequate environmental flows (NSW Ramsar Managers Network 2010; DoE 2011).

Nongovernment environmental organizations are moving beyond acquisition of land to become involved in trading, ownership, and management of water entitlements in order to restore wetland health. The Murray Wetlands Working Group is one example of a community organization selling un-needed water and acquiring water when required, often using the irrigation canal system to water wetlands (MWWG 2014). Numerous legal and operational constraints to using water for wetlands rather than irrigated agriculture are gradually being overcome.

Indigenous communities are also reclaiming their lands as these two New South Wales examples illustrate. The property Toogimbie was acquired by the Indigenous Land Corporation (a quasi-government agency that acquires land for dispossessed communities) and returned to the Nari Nari Tribal Council. These traditional owners have designated the floodplain of the Murrumbidgee River as a 4,600 ha Indigenous protected area (DoE 2013). In 2010 the New South Wales Government accepted advice from its Natural Resources Commission for around 20,000 ha of the Werai floodplain forests and Taroo lake to be owned and managed by their traditional owners as Indigenous protected areas (NRC 2009).

Conclusions

The evolution of wetland conservation law in the Murray-Darling Basin holds a number of lessons for other places. Allocating water entitlements as a share of the available resource is key to ensuring that some water is available to wetlands in dry years. Harmonizing data collection and public access is vital for providing information needed for better management. The establishment of a separate, statutory manager of environmental water ensures that this resource is deployed to best conserve wetlands. Bringing into domestic law provisions from international agreements like the Ramsar Convention can be used to increase wetland conservation measures. Overlapping powers between different levels of government (in this case, federal and state governments) may delay action but can also ensure additional consideration of wetland conservation where there is a pro-development government. Finally, enabling businesses, community organizations, and Indigenous peoples to play roles in wetland governance achieves more than governments will alone, enables innovation, and creates new constituencies for wetland conservation.

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The Okavango Delta Legal Framework

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Lars Ramberg

Contents

Abstract

The political and legal framework regulating the management of the Okavango Delta in Northern Botswana, its water, land, and natural resources is extremely complex. On a local level this is caused by frictions among traditional users, initially hunter-gatherers and latterly agriculturalists, professional hunters, and tourist operators. These uses are managed by systems of wildlife management areas, game reserves, and national parks with a trend in time away from informal local management to national governance. The Delta receives all its water, apart from direct rainfall, from the upstream riparian countries of Angola and Namibia. To manage this has resulted in the formation of a joint management structure, the OKACOM. In particular, the biological productivity, biodiversity, and beauty of the Delta makes it unique, and it has therefore been gazetted both a "Wetland of International Importance" under the Ramsar Convention and a "World Heritage Site" under the UNESCO. However, the uniqueness of the Delta is the product of the inflowing water: the quantity water, chemistry, and sediment transport. None of the protective measures address this fact, and as the pressure from the two

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upstream countries to use the river water for irrigation, diversion, and hydroelectric power production now is escalating, the integrity of the Delta is under serious threat. An improved agreement between the three riparian states is needed to address the standards of the inflowing river water, maintenance of water quantity and hydro-period, nutrients and bicarbonate concentration, as well as sediment transport. This will in turn require agreements about developments in the whole drainage basin. Such ambitions are reflected in the joint OKACOM documents, while the national plans on the contrary express the urgent ambitions to enter into river water exploitations. The risk that such actions will result in the destruction of the Delta environments is considered to be very high.

Keywords
International river basin · Irrigation · Joint management · OKACOM · Okavango Delta · Ramsar · Southern Africa · UNESCO · Water diversion

History

The Okavango Delta has been maintained in a fairly pristine state mainly due to deterrent factors. The population density in the whole drainage basin (Ramberg 2014) is very low due to low agricultural potential, a result of nutrient-poor Kalahari sands and low erratic rainfalls. In addition, all inflowing water to the Delta emanates from Angola that until 1975 was ruled as a sleepy colony by Portugal. Added to this, there have been several wars and conflicts. After independence in 1975, Angola was thrown into a civil war that, to a large extent, took place in the river basin. This conflict ceased in 2001. Namibia, through which the middle part of the river flows (Fig. 1), was a colony under South Africa until 1990. Up until 1990 a liberation war took place which effectively transformed northern Namibia into a war zone. Under such unruly conditions, no economic development initiatives could be taken (Wolski et al. 2010).

The Delta where the Okavango River ends lies in the northwestern part of Botswana, a very remote and inaccessible area in the middle of the Kalahari. Livestock rearing which is the main economic activity was not possible in the Delta due to wildlife-vectored diseases, in particular foot-and-mouth, and the tsetse fly which carries sleeping sickness to both livestock and humans. The fly was not erased until 2002.

With the independence of Botswana in 1964, a number of water development projects – such as irrigation and channeling of water usually financed by foreign aid – were initiated in the period 1960–1990. All of these initiatives failed due to a lack of knowledge of the nature of the Delta and/or lack of understanding of social and cultural factors.

Due to this cluster of deterring factors, there has been little need for any conservation measures until the last two to three decades. Need has arisen due to

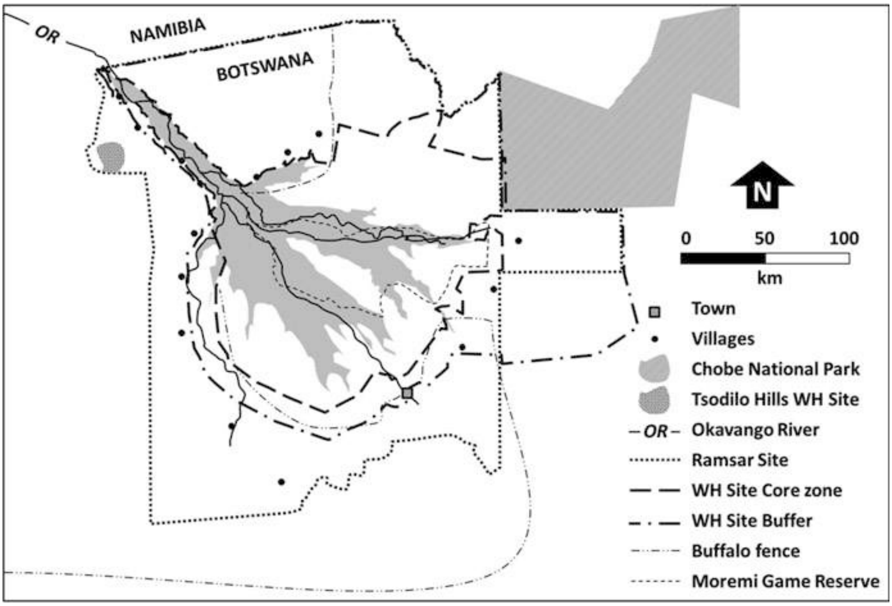

Fig. 1 The administration and legal boundaries within and around the Okavango Delta. Note the location of villages and one town at the fringe of the Delta but beyond the areas reserved for wildlife

increased pressure in all the three riparian states to use the water of the Okavango River for various economic development purposes. While Botswana has modest plans for local domestic use the two upstream countries have large plans for consumptive use of the river water for in particular irrigation but also for diversion and hydro-electric power production (OKACOM 2011b, c, d). Counter to this, strong forces have emerged that want to preserve the Delta.

The reasons for the outstanding biology of the Delta, in contrast to its surroundings, have not been fully understood until the last two decades, but are due to two unique – or very unusual – biogeochemical processes of flood switching and desalination that are vital for the very existence and maintenance of the biological system. These two processes in turn are caused by a very rare combination of (at least) five geographic, geologic, and hydrologic features and processes of which only one takes place in the Delta itself. Firstly, the Delta is located on an extension of the Great East African Rift System, a sand-filled graben more than 100 m deep and slowly sinking. The other four features are those of the inflowing Okavango River that has a pronounced seasonal flood pulse causing the flooding of the Delta, is a carrier of fine sediments, has low nutrient concentrations, and has a very high bicarbonate concentration. These are the features that need to be protected for an effective conservation of the Delta.

The Permanent Okavango River Basin Water Commission (OKACOM) 1994-

In 1994 the three riparian countries of the Okavango River system signed an agreement to work for joint planning, management, and information sharing regarding the river basin and to form a joint commission for that purpose: the Permanent Okavango River Basin Commission (OKACOM 1994). The two upstream countries Angola and Namibia were predominantly interested in water withdrawals for domestic use and irrigation as well as to construct hydroelectric power plants, where 17 suitable sites had been identified in Angola and one in Namibia. Botswana on the other hand, as the end user, emphasized the need for water for conservation of the Okavango Delta.

The OKACOM agreement concerns joint monitoring, data sharing, and planning of resource use; prior reporting on plans that might affect the water resources of the basin; and consensus between the three states before any actions are taken. A major aim of OKACOM was to develop a joint development and management plan for the river basin with emphasis on sharing of the water. This work progressed very slow partly due to the civil war in Angola and partly due to lack of trust between the parties. This caused a serious crisis in 1996.

After several years of drought, in 1995 Namibia started to plan for a forced construction of a pipeline from the Okavango River up to their highland channel system that would deliver the water to the capital Windhoek (Ramberg 1997). This was done as an “emergency situation” with reference to the Helsinki rules and without prior consultation with Angola and Botswana. It caused very strong reactions in Botswana, and one of the countermeasures became to unilaterally ratify the Ramsar Convention in 1996 and declare the Okavango Delta a Ramsar site: a wetland of international importance, which was endorsed by the convention in 1997.

Fortunately, the drought ended with good rains in the beginning of 1997, and the construction of the pipeline was put on hold.

In spite of problems, the OKACOM continued to meet, and from the beginning of the new millennium, it has made considerable progress and trust between the three parties has improved. Existing knowledge on the whole river basin regarding its hydrology, geography, biology, and social and economic factors has been collected and reported (OKACOM 2011a) in a transboundary diagnostic analysis (TDA) document. A permanent secretariat has been established in Maun that handles communication, information, documentation, and organizational matters. The commission meets at least once a year, and a number of joint technical subcommittees are established.

After the TDA report three National Strategic Action Plans (NAP) have been developed (OKACOM 2011b, 2011c, 2011d) and based on all four documents a Strategic Action Plan (SAP) for the whole system (OKACOM 2011e) has been produced.

Southern African Development Community (SADC) 2000

The democratic transition in South Africa between 1990 and 1994 vitalized the collaborative work in the Southern African Development Community (SADC) that organizes the 15 southernmost states in Africa, including the three riparian countries sharing the Okavango Basin. A number of binding protocols has been developed, among them the Revised Protocol on Shared Watercourses in the Southern African Development Community (SADC 2000), of which there are several within the region. The organization, communication, and mode of operation of shared river commissions within the SADC is outlined. Importantly, it is specific on prior reporting about planned impacts, how these should be done, and how disputes between parties sharing a water resource should be resolved. Consensus between the parties is a cornerstone principle. It thus overlaps and overarches the OKACOM agreement.

The Ramsar Convention 1997

After the gazetting by the Ramsar Convention of the Okavango Delta as a "Wetland of International Importance" in 1997, the government of Botswana started to develop – as a condition of the Ramsar Convention – an Okavango Delta Management Plan (ODMP). This involved all relevant government departments and the Tawana Land Board (see below) under the leadership of a temporary project management structure within the newly formed Department of Environmental Affairs. It was a huge effort with several hundred meetings between government departments, various technical subcommittees, and consultants on one hand and on the other hand more than 150 consultations with stakeholders including local communities, tourism operators, etc. The planning project was finalized in 2008 (DEA 2008) and the planning structures dissolved.

Formally, it then became the responsibility of the Department of Environmental Affairs to implement the ODMP, but each of 11 subdepartments is in charge of its normal area of responsibility such as departments of water affairs, wildlife and national parks, tourism, livestock, and animal production. The jurisdiction of the land itself however is under the Tawana Land Board that is a traditional tribal authority that allocates land for a limited period to different users, and no land within the whole North-Western District can be bought or sold.

The gazetting and demarcation of the Ramsar site did therefore not change the traditional power sharing between these authorities; it just placed a larger frame around older legal and physical ones (Fig. 1). An important element is the buffalo fence – for which the Department of Livestock and Animal Production is responsible – that surrounds the eastern and southern parts of what is normally considered as the Okavango Delta. Its aim is to prevent wildlife-vectored diseases to spread into livestock. No permanent settlements are allowed either. There are, however, two old villages, and a number of "service villages" are emerging adjacent to tourist camps.

The Moremi Game Reserve of 5000 km^2 (including a small part of Chobe National Park, Fig. 1) in the east and central parts of the Delta was gazetted by the Tawana tribe during the 1940s. This is now the core of the Ramsar site and targeted for wildlife protection. Game-viewing tourism is however allowed and about 16 tented camps have been erected. This area is managed by the Department of Wildlife and National Parks and entrance fees are charged.

Outside this area and roughly up to the buffalo fence (Fig. 1), the Delta is subdivided into 29 Wildlife Management Areas (WMAs) meant as a buffer zone of 13,000 km^2, where sustainable use of wildlife resources is allowed, but no permanent settlements are permitted. Of the WMAs, 13 are given to local communities; the rest are tendered for and leased out to tourist operators. All are used either for scenic tourism or for safari hunting. The hunting quotas are set by the Department of Wildlife and National Parks, and traditional harvesting by local villagers of natural resources such as wood, reeds, thatching grass, and fish is allowed, which is mainly supervised by the Tawana Land Board.

The total area of the Okavango Delta Ramsar site is 68,640 km^2 of which about 26% – most of the Okavango Delta – is devoted to wildlife conservation. The rest, mostly dry Kalahari savanna, is used for small-scale agriculture or low-intensity livestock ranching. Notably the Ramsar site has borders in common with national parks in Botswana and Namibia and one small Ramsar site at the Okavango River inlet (Fig. 1) adding another 19,000 km^2 to a common wetland conservation area, connecting the hydrologic- and biologic-related Okavango and Zambezi river systems.

UNESCO World Heritage Site

In 2014 the Okavango Delta was inscribed as a UNESCO World Heritage Site on the initiative of Botswana. The designated area of 20,000 km^2 and an additional 23,000 km^2 as a buffer zone, is considerable (Figure 1) but smaller than that of the Ramsar Site, mainly because it avoids areas that could be exploited (such as by mining). All the three riparian countries are members of UNESCO and the effective protective power of both the World Heritage status as well as the designation as a Ramsar Site demonstrate their willingness to be good and proactive members of the world community.

Conclusions

On the national level Botswana has put in place a set of policies and laws that define the user status of wildlife management areas, game reserves, and national parks that encompass practically the whole Delta (Fig. 1). In any of the user categories the emphasis is on conservation. The two international agreements, the Ramsar

Convention and the UNESCO World Heritage Convention put emphasis on conservation but are restricted to the Delta itself (Fig. 1) and address only in passing the need to maintain the integrity of the drainage basin.

Although the Delta itself is shielded by an almost overwhelming blanket of protective policies, the properties of the inflowing river water, its quantity and quality, are not protected by any policy. It is however the unique features of this inflow that create and maintain the exceptional features of Okavango Delta (Ramberg and Wolski 2008). The two collaborative agreements of a regional character – OKACOM and SADC – put emphasis on *water use and water sharing* in the whole river basin. These are the structures where politics and policies are formed that decide the fate of the Okavango Delta and the wider river basin.

In the Transboundary Diagnostic Analysis (OKACOM 2011a; King and Chonguica 2016), a hydrologic-economic analysis has been made for three levels of water utilization in the drainage basin based on national plans for irrigation, hydroelectric dams, and water diversions. All three scenarios predict net economic losses and severely so for the medium- and high-level alternatives. Reduced flooding of the Delta and loss of tourism revenue in Botswana – its second largest income earner – are the major reasons, but economic gains in the upstream exploiting countries are likely to be negative due to poor soils, poor infrastructure, and remoteness. In spite of this the national strategic action plans (NAP) published after the TDA report, the plans for Angola and Namibia put emphasis on the development of irrigation and water diversion. Thereafter a joint SAP for the whole river basin has been produced by OKACOM (2011e) but was not made public until the end of 2016. The emphasis of this agreement is to continue to strengthen administrative and management structures, to start environmental monitoring in the whole drainage basin, to improve forecasts and models, and to develop a joint vision for the future of the Okavango system (King and Chonguica 2016). No exploitive measures, such as irrigation or water diversions, are considered or even mentioned. However, in the same document it is clearly stated that the ambition to pursue the exploitive projects as planned in the National Action Plans (NAPs) remains unchanged, "The implementation of the NAPs moves forward and independently of the SAP process. . . ." In these national plans there is also an ambition to move fast and interventions within the drainage basin might have already happened as there is no definitive information about the status since 2010.

Both the OKACOM TDA and the SAP documents reflect a lack of understanding of the uniqueness of the Delta, in particular the vital importance of flood switching and desalination. The unpredictability of flooding is described as a major problem that ought to be controlled, while its vital role for the biologic functioning of the Delta is not emphasized. The absolute critical function of desalination is not mentioned at all, although the magnitude of its importance in maintaining the fresh water nature of the Delta is immense. The sensitivity or fragility of this process is now impossible to predict with any certainty as there are limited in-depth studies and, due to the uniqueness of the Delta, no comparable systems.

It must be emphasized that the planned, proposed, or already ongoing intrusions if fully implemented will significantly reduce river flow and cause a number of changes in the water quality as well, which is likely to cause a sudden collapse and a complete change of the nature of the Delta (Ramberg 2016; Ramberg and Wolski 2008).

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European Union Natura 2000

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Robert J. McInnes

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Abstract

Natura 2000 is the centerpiece of European Union (EU) nature and biodiversity policy. It is an EU-wide network of nature protection areas established under the 1992 Habitats Directive. The aim of the network is to assure the long-term survival of Europe's most valuable and threatened species and habitats. It is comprised of Special Areas of Conservation (SACs), designated by member states under the Habitats Directive, and also incorporates Special Protection Areas (SPAs) which they designate under the 1979 Birds Directive. The establishment of the Natura 2000 network of protected areas also fulfills a European community obligation under the United Nations Convention on Biological Diversity (CBD). Today, the network comprises over 25,000 sites representing 18% of the area of the member states of the European Union.

Keywords

International agreements · Habitats directive · Birds directive · Transboundary agreements

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Background

Natura 2000 is the centerpiece of European Union (EU) nature and biodiversity policy. It is an EU-wide network of nature protection areas established under the 1992 Habitats Directive (EC 1992). The aim of the network is to assure the long-term survival of Europe's most valuable and threatened species and habitats. It is comprised of Special Areas of Conservation (SACs), designated by member states under the Habitats Directive, and also incorporates Special Protection Areas (SPAs) which they designate under the 1979 Birds Directive (EC 1979). The establishment of the Natura 2000 network of protected areas also fulfills a European community obligation under the United Nations Convention on Biological Diversity (CBD). Today the network comprises over 25,000 sites representing 18% of the area of the member states of the European Union.

Natura 2000 is not a system of strict nature reserves where all human activities are excluded. The network includes many nature reserves; however most of the land is privately owned, and the emphasis is on ensuring that future management is sustainable, both ecologically and economically.

Natura 2000 applies to bird sites (SPAs) and to habitat sites (SACs), which are divided into biogeographical regions. It also applies to the marine environment.

Bird Sites

The identification and delimitation of SPAs is based on scientific criteria such as "1% of the population of listed vulnerable species" or "wetlands of international importance for migratory waterfowl." Hence, there is considerable overlap with designated Ramsar sites. Member states have a margin of discretion in determining the most appropriate criteria. However, they must then fully apply those criteria in a way that ensures that all the "most suitable territories," both in number and surface area, are designated. On the basis of information provided by the member states, the European Commission determines if the designated sites are sufficient to form a coherent network for the protection of the vulnerable and migratory species.

Habitat Sites

SACs are established with the aim to protect the 220 habitats and approximately 1,000 species listed in Annex I and II of the directive which are considered to be of European interest following criteria given in the directive (EC 1992). Under the Habitats Directive, there are three stages in the selection of SACs:

1. The responsibility for proposing sites under the Habitats Directive lies with the member states. Each state carries out comprehensive assessments of each of the habitat types and species present on their territory. The choice of sites is a purely

scientific process, based on standard selection criteria specified in the directive. Site-specific data are communicated to the Commission by using Standard Data Forms.

2. On the basis of the proposed national lists, the Commission, in agreement with the member states, must adopt the lists of "Sites of Community Importance." Scientific seminars are then convened by the Commission for each biogeographical region in order to analyze the member states' proposals in a transparent way. They are open to the member states concerned and to experts representing relevant stakeholder interests, including owners, users, and environmental NGOs. These seminars are supported by the European Environment Agency, assisted by the European Topic Centre on Biological Diversity which is based in Paris, France. These expert seminars aim to establish if sufficient high-quality sites have been proposed by each member state to ensure the favorable conservation status of each habitat type and species throughout their range in the EU. The objective is to establish a list of "Sites of Community Importance" for each of the regions determined by the Habitats Directive, applying a consistent approach across the member states.
3. Once the lists of "Sites of Community Importance" have been adopted, it is for the member states to designate all of these sites as "Special Areas of Conservation," as required by the Habitats Directive, as soon as possible and within 6 years at the most. They should give priority to those sites that are most threatened and/or that are of most importance in conservation terms. During this period, member states must take the necessary management or restoration measures to ensure the favorable conservation status of those sites.

Biogeographical Regions

The Habitats Directive divides the EU into nine ecologically coherent "biogeographical" regions – the **Atlantic**, **Continental**, **Alpine** (which includes the Pyrenees, the Alps, the Carpathian Mountains, and parts of Scandinavia), **Mediterranean**, **Boreal** (Finland, Sweden, Estonia, Latvia, and part of Lithuania), the **Macaronesian** (Madeira, Azores, and Canary Islands), the **Pannonian** (essentially Hungary and parts of the Czech Republic, Romania, and Slovakia), the **Steppic,** and the **Black Sea** Region (parts of Bulgaria and Romania). Natura 2000 sites are selected according to each biogeographical region. Working on this level makes it easier to conserve species and habitat types under similar natural conditions across a suite of countries.

Future Challenges

The development of the Natura 2000 network has not been without considerably scientific and political challenges (Keulartz and Leistra 2008); however it is now moving toward the delivery of a coherent and robust network of protected sites.

But designation remains the starting point for sites, rather than the end. Sites still require protection from loss and damage and monitoring to assess the effectiveness of protection (Evans 2012). A serious future challenge will be the response of member states to a changing climate and especially the squeeze on coastal sites as a result of the dual pressures of a rising sea level and development pressures.

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European Union Water Framework Directive and Wetlands

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Robert J. McInnes

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Abstract

Across Europe, rivers, lakes, and coastal waters are vital natural resources which provide drinking water for humans, provide habitats for many different species of wildlife, and are an important resource for inter alia agriculture, industry, and recreation. Through generations of misuse, a significant proportion of these wetlands have become degraded or threatened. The future protection and improvement of the water environment has been recognized at the European Union (EU) level as being an important component of achieving sustainable development and vital for the long-term health, well-being, and prosperity of European citizens. In October 2000 the Directive 2000/60/EC of the European Parliament and Council established a framework for Community action in the

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field of water policy, commonly referred to as the Water Framework Directive or WFD. The purpose of the Directive is to establish a framework for the protection of inland surface waters (rivers and lakes), transitional waters (estuaries), coastal waters, and groundwater. It aims to ensure that all aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands meet “good status” by 2015.

Keywords
European union · Legislation · Policy · International cooperation

Background

Across Europe, rivers, lakes, and coastal waters are vital natural resources which provide drinking water for humans, provide habitats for many different species of wildlife, and are an important resource for inter alia agriculture, industry, and recreation. Through generations of misuse, a significant proportion of these wetlands have become degraded or threatened. The future protection and improvement of the water environment has been recognized at the European Union (EU) level as being an important component of achieving sustainable development and vital for the long-term health, well-being, and prosperity of European citizens.

In October 2000 the “Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy” (European Commission 2000) (or in short the *Water Framework Directive* or WFD) was adopted and came into force in December 2000. The purpose of the Directive is to establish a framework for the protection of inland surface waters (rivers and lakes), transitional waters (estuaries), coastal waters, and groundwater. It aims to ensure that all aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands meet “good status” by 2015.

The WFD improved on earlier, piecemeal EU water legislation and expanded the scope of water protection to all waters (Chave 2001). The WFD also adopted the most appropriate model for a single system of water management, namely, the river basin. River basins are the natural geographical and hydrological unit and therefore the logical basis for water management, as opposed to working with administrative or political boundaries. In the development of the WFD, initiatives developed by the states concerned for the Maas, Scheldt, or Rhine river basins have served as positive examples of this approach, with their cooperation and joint objective setting across member state borders or in the case of the Rhine even beyond the EU territory. While several member states were already adopting this approach, it was recognized that this was not universal across the EU.

For each river basin district, several of which traverse national frontiers, the Directive requires a cyclical process where river basin management plans are prepared, implemented, and reviewed every 6 years. The first cycle covers the period

2009–2015. Within the river basin management process, there are four distinct elements: the characterization and assessment of impacts on river basin districts, the environmental monitoring, the setting of environmental objectives, and the design and implementation of the program of measures needed to achieve them.

Horizontal Guidance on the Role of Wetlands in the WFD

The WFD clearly identifies the protection, restoration, and enhancement of the water needs of wetlands as part of its purpose. Article $1_{(a)}$ of the WFD states that:

> The purpose of this Directive is to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater which:
> (a) prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems.

However, the Directive failed to provide any specific definition for wetlands. Similarly, the Directive does not state clearly or explicitly the extent to which wetlands should be used for the achievement of environmental objectives. Member states and stakeholders felt that it would be helpful to explore and clarify the role of wetlands in implementing the Water Framework Directive. Therefore a Common Implementation Strategy guidance document was developed, in which the pressures on wetlands were acknowledged, and their potentially important roles in river basin management, and particularly in helping to achieve WFD environmental objectives, were clarified. This resulted in the preparation and publication of *horizontal guidance on the role of wetlands in the WFD* (European Union 2003).

Aspects of implementation are clearly described in the horizontal guidance document including where the appropriate use of wetland protection and restoration could help to fulfill environmental objectives in a cost-effective and sustainable manner. The document also acknowledged that wetlands are a crosscutting issue and elaborated a common understanding of the WFD requirements regarding wetlands and their role in its implementation. In some cases, where additional effort could lead to considerably enhanced results, the horizontal guidance goes one step further and illustrates best practices beyond the legal requirements of the WFD while recognizing that member states have the flexibility to establish stricter environmental protection according to their particular national concerns and situations.

The WFD and the associated horizontal guidance do not seek to establish a new international definition of wetlands for the purposes of the Directive. Rather, the guidance explains the relevance of wetlands to the achievement of the Directive's environmental objectives. Key within this approach is the attention given to key relationships among significant elements of the hydrological network and the role of wetlands with respect to this approach.

It could be argued that the Directive is lacking in two principal areas in relation to wetlands. Firstly, while the Directive refers to wetlands (Recitals 8 and 23, Article $1_{(a)}$, and Annex $VI_{(vii)}$), it does not provide a definition for them or provide a size range to indicate their dimension. Secondly, the Directive does not set specific obligations or recommendations for wetlands or other terrestrial ecosystems per se. However, the environmental objectives of the WFD are to be applied to, and monitored through, "water bodies"; therefore it is important for member states to have a clear understanding of the relationship between water both surface and groundwater bodies and wetlands, in order to understand how these systems might be encompassed within the cycle of river basin planning (European Union 2003).

The Directive sets out the following environmental objectives for water bodies:

- Preventing deterioration in status
- Achieving good surface water status or, for artificial or heavily modified surface water bodies, good ecological potential and good surface water chemical status
- Good groundwater status
- Any less stringent objective applicable under Article 4.5

The focus of the WFD on water bodies, and particularly their relationships within a river basin, helps to emphasize the functional role of wetland systems within the wider hydrological cycle. This is reflected in the Directive by means of a complex set of provisions which are illustrated in Fig. 1.

Figure 1 illustrates the various different ecosystems that may be present in a river basin district and which may be relevant to the achievement of the Directive's objectives. The relative sizes and overlaps of the ellipses depend on the sorts of ecosystems present within each river basin district. The central ellipse represents the "universe" of wetlands and demonstrates its reach into the various component ecosystems. The following sections of the guidance describe the role of these different ecosystems in the river basin management planning process.

The nature of the role of each single category of wetland typology identified in Fig. 1 is described in more details below.

Surface Water Bodies (River, Lake, Transitional, and Coastal Waters)

These can be divided into two systems:

(a) **Wetland ecosystems identified as water bodies**
Many wetland ecosystems are mosaics of surface water, permanently and temporarily inundated or waterlogged land, such as lowland mire systems or floodplain wetlands. WFD provisions in relation to surface waters directly help to protect and enhance these wetland ecosystems through their designation as water bodies, and by setting objectives for them, where they fall within the WFD categories of rivers, lakes, and transitional or coastal waters.

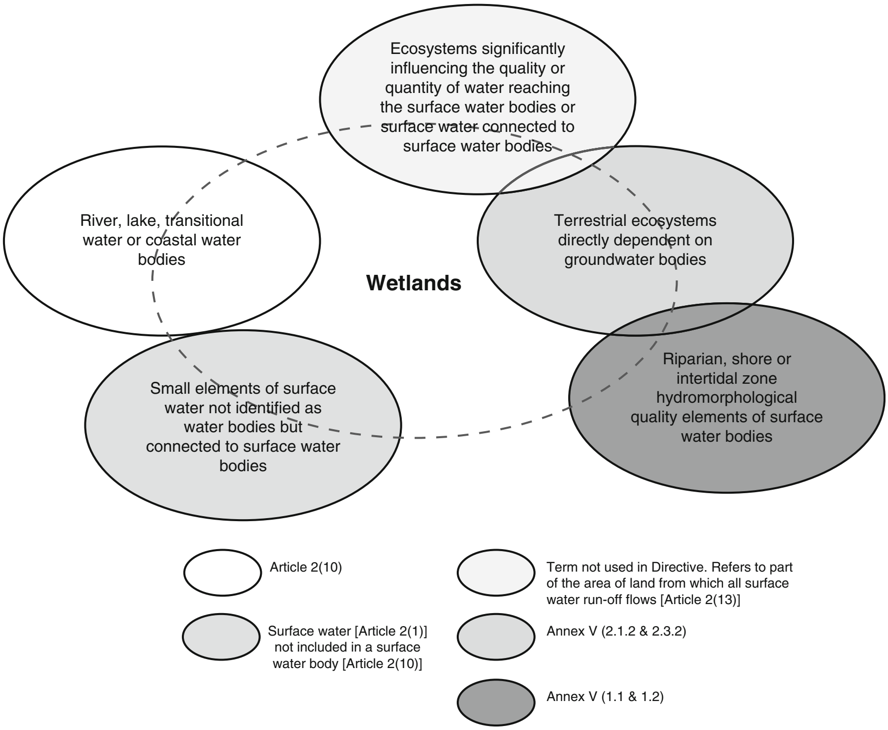

Fig. 1 Ecosystems relevant to the achievement of the Directive's objectives

(b) **Riparian, shore, and intertidal zone quality elements of surface water bodies**
The hydromorphological quality elements of surface water bodies include the structure and condition of the riparian zone of rivers, the shore zone of lakes, and the intertidal zones of transitional and coastal waters. These zones can include ecosystems which are widely regarded as wetlands, where the structure and condition of such wetlands is relevant to the achievement of the objectives for a surface water body.

Ecosystems Significantly Influencing the Quality and Quantity of Water Reaching Surface Water Bodies or Surface Waters Connected to Surface Water Bodies

Ecosystems which are adjacent to, or contiguous with, water bodies and which may influence the status of those water bodies should be encompassed within the riparian, lakeshore, or intertidal zones (above), in order to ensure effective operation of WFD environmental objectives. However, there may be other wetland ecosystems within river basins which are not adjacent to water bodies and therefore do not form part of the riparian, shore, or intertidal zones, which may influence the quality and quantity of water reaching those bodies or reaching small elements of surface waters connected to those bodies. Member states need to ensure that the quality and quantity of water entering surface water bodies via these disparate wetland ecosystems is such as to ensure the achievement of the relevant objectives for the water bodies. In doing so, member states may determine where appropriate or necessary to manage relevant activities on, protect, enhance, restore, or even artificially create these wetlands.

Terrestrial Ecosystems Directly Depending on Groundwater Bodies

The Directive's objectives of achieving good groundwater quantitative status and good groundwater chemical status require inter alia that the groundwater needs of terrestrial ecosystems that are directly dependent on bodies of groundwater are protected and, where necessary, restored to avoid or remedy significant damage to such ecosystems. The terrestrial ecosystems that depend directly on a body of groundwater will include types of systems that occur in areas where the water table is at or near the surface of the ground, or, in other words, wetlands.

Small Elements of Surface Water Connected to Water Bodies but Not Identified as Water Bodies

There are major practicality issues in identifying every element of surface water in a river basin district as a water body or part of a water body. Member states have to decide within the river basin management planning process which elements of surface water are not sufficiently discrete and/or significant to be identified as water bodies. Many of the elements of surface water that are not identified will nevertheless be important elements in wider hydrological cycles and potentially connected to surface water bodies. Such elements need to be protected or, in some cases, enhanced and restored to ensure that anthropogenic impacts do not compromise the achievement of the environmental objectives of the water bodies to which they are connected. In some cases, member states may even choose to artificially create such surface waters where they determine that this is an appropriate or necessary means of achieving the objectives of the Directive for surface water bodies. An example of this would be the creation of constructed wetlands to mitigate the impacts of urban or agricultural runoff on river water bodies.

Future Challenges

Considerable progress has been made on the implementation of the WFD (European Union 2010). However, significant challenges remain to fully achieve the ambitious objectives of the Directive. Some of these challenges revolve around the degree of fit within institutional structures and practices (Moss 2004), while others relate both directly and indirectly to wetland management issues.

For instance, the concept of environmental flows is well established in wetland science and literature. However, the WFD does not explicitly use the term environmental flows but requires member states to achieve good ecological status in all water bodies through an assessment of aquatic biology. Nevertheless, it is accepted that to meet ecologic targets requires appropriate hydrological regimes to be in place. Therefore the implementation of environmental flows will be a key measure for restoring and managing river ecosystems and delivering on the WFD objectives (Acreman and Ferguson 2010).

Similarly, wetlands, as small elements of surface water bodies, can play a key role in mitigating the impacts of agricultural pollution (Harrington and McInnes 2009). However, results of economic modeling from the Upper Ems river basin in northwestern Germany have shown that drastic measures, which include wetland protection, are unrealistic from a socioeconomic point of view. To achieve the water quality target in the basin would require not only an increase in the area of protected wetlands from 0% to 9%, but accompanied land-use changes such as a reduction of arable land from 77.2% to 46%, an increase of pasture from 4% to 15%, and afforestation from 10% to 21% (Volk et al. 2009). Furthermore, there are additional barriers to the integration of wetlands into programs of measures through the failure of traditional environmental accounting. A study from the River Elbe demonstrates that riparian wetlands provide significant benefits that should be considered in river basin management decisions; however these are often neglected leading to biased cost–benefit analysis results and therefore misguided the decision-making processes (Meyerhoff and Dehnhardt 2007).

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North America Transnational Legal Frameworks

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Abstract

There are a number of legal frameworks that directly or indirectly affect wetlands management in North America. All are designed to promote transnational cooperation to protect, conserve, and manage wetlands. These legal frameworks include regional agreements that encompass the three countries of North America (Canada, Mexico, and the United States), as well as bilateral agreements between Canada and the United States and the United States and Mexico in relation to transboundary waters.

Keywords

Legal frameworks · Transnational · Multilateral · Bilateral · North America

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Introduction

There are a number of legal frameworks that directly or indirectly affect wetlands management in North America. All are designed to promote transnational cooperation to protect, conserve, and manage wetlands. These legal frameworks include regional agreements that encompass the three countries of North America (Canada, Mexico, and the United States) as well as bilateral agreements, between Canada and the United States and the United States and Mexico in relation to transboundary waters.

Regional Agreements

1988 U.S.-Mexico-Canada Tripartite Agreement on the Conservation of Wetlands (Tripartite Agreement)

The historical underpinnings of the Tripartite Agreement are found in the Convention for the Protection of Migratory Birds signed between the United States and Great Britain (representing Canada) in 1916, which was created to counteract the decline in migratory bird populations due to unregulated harvesting for meat and plumage, among other uses. International cooperation to protect migratory birds was expanded to include Mexico in 1936 as well as Japan in 1972 and the Soviet Union in 1978. In 1986, a desire to protect declining populations of migratory waterfowl led to the North American Waterfowl Management Plan signed between Canada and the United States. Mexico joined this collaborative effort in 1988 via the Tripartite Agreement, which was signed by the directors of the three federal wildlife agencies in order to create a management plan to restore and maintain not only waterfowl populations but also the wetlands habitat that was critical for waterfowl as well as other wetland-dependent wildlife. The North American Waterfowl Management Plan envisioned collaborative conservation in the form of Joint Ventures whereby government agencies, nonprofit organizations, corporations, tribes, and/or individuals would plan, fund, and implement projects to conserve high priority bird habitat, including wetlands. Since the first Joint Venture was created in 1987, a total of 24 Joint Ventures have been created that geographically cover nearly all of Canada and the United States and much of Mexico (USFWS, Joint Venture Map 2014a). A number of these Joint Ventures have focused on conservation of wetlands that stretch across borders, including the Pacific Coast Joint Venture, the Rio Grande Joint Venture, and the Sonoran Joint Venture (USFWS, Joint Venture Directory 2014b; Pacific Coast Joint Venture 2014; Rio Grande Joint Venture 2014; Sonoran Joint Venture 2014).

North American Agreement on Environmental Cooperation (NAAEC)

The NAAEC was negotiated in conjunction with the North American Free Trade Agreement (NAFTA) and came into force at the same time on January 1, 1994 (CEC, About the CEC 2014a). As noted in the listed objectives in Article 1 of the NAAEC, one of the goals was to "increase cooperation between the Parties to better conserve, protect, and enhance the environment, including wild flora and fauna," (USTR, NAAEC 1994). The NAAEC established the Commission for Environmental Cooperation (CEC), an international organization to support cooperation among the NAFTA Parties to address environmental issues across North America. Over the past 20 years, the environmental ministries of the three countries have funded the work of the CEC, which in turn has coordinated a cooperative program of environmental work. A number of the cooperative programs have touched on wetlands management. For example, a collaborative study was done to examine the extent and dynamics of carbon sequestration of blue carbon ecosystems in North America, including coastal wetlands (CEC, North America Blue Carbon Scoping Study 2014b). As blue carbon ecosystems, including coastal wetlands, have been found to store two to four times more carbon in their vegetation and soils than mature tropical forests, these ecosystems create the potential for carbon offset payments when they are conserved or restored in the context of developing carbon markets (NOAA 2014).

Bilateral Agreements

Canada and the United States

Canada and the United States have entered into bilateral agreements related to transboundary waters, such as the Boundary Waters Treaty (BWT), which dates back to 1909. The BWT sets forth general principles to be followed by the two nations in relation to commerce, navigation, water use, diversion, and obstruction with the aim of preventing and/or resolving issues related to shared waters. The BWT established the International Joint Commission (IJC), which consists of three appointed independent experts from each country who regulate shared water usage, investigate transboundary issues, and recommend solutions (IJC, BWT 1909). The IJC's guiding principles include a focus on sustainable development, an ecosystem approach, elimination of persistent toxic substances, sound science and, where necessary, a precautionary approach (IJC, Guiding Principles 2014a). The IJC's regulation of shared water uses, including decisions regarding water level and flow across the boundary, has a direct affect on wetlands management. In addition, the IJC has reporting requirements under the Great Lakes Water Quality Agreement (GLWQA), which was signed by Canada and the United States in 1972 and later amended several times to address additional environmental concerns related to the waters in and around the Great Lakes, including the surrounding wetlands (IJC, Mission and Mandates 2014b). Article 3 of the 2012 Protocol to the GLWQA includes the objective of "support[ing] healthy and productive wetlands" (IJC, GLWQA 2012). In the 15th Biennial Report on Great Lakes Water Quality, the IJC focused on the nearshore zones of the Great Lakes and made a number of recommendations to address threats to nearshore water quality, including the use of

Lakewide Water Management Plans that would encompass wetlands (IJC, 15th Biennial Report 2011).

Mexico and the United States

There are several treaties between Mexico and the United States dating back to 1848 that focus on delineating the boundary between the two countries in relation to the river systems along the border (IBWC, Treaties Between the United States and Mexico 2014a). The 1944 Treaty, entitled "Utilization of Waters of the Colorado and Tijuana Rivers and of the Rio Grande," forms the basis for regulating shared waters. Article 3 of the 1944 Treaty entrusts the International Boundary and Water Commission (IBWC) with the regulation of joint use of the waters, including for "beneficial uses which may be determined by the Commission" (IBWC, 1944 Treaty). The IBWC is an international body composed of a United States Section (staffed by the U.S. Department of State) and a Mexican Section (staffed by the Secretariat of Foreign Relations of Mexico). Each section is headed by an Engineer-Commissioner appointed by his/her respective President. The IBWC implements the boundary and water treaties, including the 1944 Treaty, through cooperative projects and attempts to resolve any disputes that may arise under the various treaties (IBWC, About Us 2014b). By way of example, in March 2014, thanks to a landmark agreement between the United States and Mexico under the 1944 Treaty known as "Minute 319" and the related cooperative project coordinated by the IBWC and a binational coalition of nongovernmental organizations, a pulse flow of approximately 105,000 acre-feet of water from the Colorado River will mimic the natural spring floods and inundate the wetlands in the delta region for the first time since 1960 (IBWC, Minute 319 November 2013; National Geographic, Water Currents 2014).

Future Challenges

While the decline in wetlands acreage in North America is less precipitous than the period prior to the 1970s, the decline continues in many areas due to many pressures including urban development and farming (EPA 2014). As wetlands supply vital ecosystems services such as filtering out pollutants, absorbing excess rainwater to prevent flooding, and providing vital wildlife habitat, legal frameworks for collaborative efforts in North America will need to address how to value these ecosystem services such that the continued existence of the wetland is more valuable than draining and filling for development.

Climate change poses a number of significant impacts on wetlands, including altered hydrology, such as flooding and drought, erosion, spread of invasive species, and extended range for pest and disease vectors (Erwin 2008). In light of this, current collaborative efforts will need to devote more focus and funding on climate adaptation in order to conserve and restore the wetlands of North America.

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Climate Change and Wetlands

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C. Max Finlayson

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Abstract

Projected changes in the climate are expected to increase temperatures, modify precipitation, raise sea levels, and increase extreme climate events with large impacts on wetlands, including their occurrence and extent, vegetation structure, ecological processes and functions, as well as the livelihoods and wellbeing of the people that depend on the wetlands and their ecosystem services. Some changes in wetland species have already been observed with both freshwater and marine species shifting their geographic ranges, seasonal activities, and migration patterns. At the same time it has been recognized that some wetlands store and

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emit greenhouse gases with their degradation and destruction contributing to greater releases of such gases and negative implications for atmospheric temperatures.

Keywords
Climate change · Vulnerability · Adaptation · Mitigation

Introduction

The Intergovernmental Panel for Climate Change (IPCC) has provided regular assessments on the consequences of climate change on ecosystems based on thorough assessments of published information. The Fifth Assessment Report (IPCC 2014) has documented information about the impact of change in climate on natural systems including wetlands, both inland and coastal/marine, as well as on the quality and quantity of the hydrological systems that are so important for wetlands. Projected changes in the global climate are expected to increase temperatures, modify precipitation, raise sea levels, and increase extreme climate events with large impacts on wetlands, including their occurrence and extent, vegetation structure, ecological processes and functions (Junk et al. 2013), as well as the livelihoods and wellbeing of the people that depend on the wetlands and their ecosystem services (Finlayson et al. 2006; Horwitz et al. 2011). Some changes in wetland species have already been observed with both freshwater and marine species shifting their geographic ranges, seasonal activities, and migration patterns (Finlayson et al. 2006). At the same time it has been recognized that some wetlands store and emit greenhouse gases with their degradation and destruction contributing to greater releases of such gases and negative implications for atmospheric temperatures (Lloyd et al. 2013).

In response to these scenarios, Junk et al. (2013) undertook a continental/regional analysis of the likely impacts of climate change on wetlands. This was based on individual analyses undertaken for North and Central America (Mitsch and Hernandez 2013), South America (Junk 2013), Africa (Mitchell 2013), East and High Asia (An et al. 2013), tropical Asia (Gopal 2013), Europe (Čížková et al. 2013), Russia (Robarts et al. 2013), and Australia (Finlayson et al. 2013). In doing this it was noted that the current distribution and extent of wetlands no longer coincided closely with that which previously existed as a consequence of conversion and loss (MEA 2005; Davidson 2014) and the alteration or replacement of others, such as the construction of rice paddy in many parts of Asia in particular. In this respect, climate change was seen as an additional pressure on wetlands that were already under severe pressure (MEA 2005). Further, the extent of wetlands in many regions is still poorly known with inadequate inventory (Finlayson et al. 1999; Finlayson 2012), including within large basins such as the Amazon (Junk 2013), and limited understanding of the important role that wetlands have played in sustaining human health and wellbeing (Horwitz et al. 2011).

The following text addresses the vulnerability of wetlands to climate change, the likely effects at a regional/continental scale, the more specific effects on migratory and nomadic waterbirds given their reliance on multiple wetlands, and finally a discussion on adaptation or responses to such effects.

Mitigation of Climate Change

Mitigation in the context of wetlands and climate change describes any action that prevents, reduces, or slows climate change and can be achieved either by reducing greenhouse gas emissions from wetlands or by enhancing the capacity of wetlands to act as greenhouse gas sinks. The importance of carbon storage and emissions from wetlands, in particular from temperate or tropical peatlands, and coastal wetlands, such as salt marshes, mangroves, and seagrass beds, is increasingly being recognized. It is well known that many wetlands contain large stores of carbon, much of it laid down over centuries, and if these were drained or otherwise converted, large amounts of carbon in the form of greenhouse gases will be released to the atmosphere and contribute to further climate change. This is well known for peatlands, both forested and nonforested. Tropical forested peatlands in particular have received a great deal of attention, with more recent and thorough investigations reducing some of the uncertainties associated with previous estimates (Page et al. 2011). Under existing land use practices it is expected that subsidence and carbon loss from tropical peatland is inevitable in oil palm plantations and could end in flooding and loss of agricultural production as well as contributing to the continuing destruction of valuable ecosystems and increased carbon emissions.

The Food and Agriculture Organization of the United Nations and Wetlands International have produced guidance for the management of peatlands, including the identification of finance options, to achieve reductions in emissions and enhance the maintenance of other ecosystem services from peatlands (Joosten et al. 2012). The guidance provides a decision support tree as a guide to opportunities for both cultivated and uncultivated peatlands and outlines methods and data available for quantifying emissions from peatlands and other organic soils to achieve reductions in emissions and enhance the maintenance of other ecosystem services from peatlands.

There has recently been increased attention to the storage of carbon in coastal ecosystems, notably mangroves, tidal saltmarshes, and seagrass beds, which can store large amounts of carbon in the underlying sediment (Herr et al. 2011). There is growing evidence that the management of these “blue carbon” wetlands has the potential to transform global carbon management, contribute to avoiding further loss and degradation of these ecosystems, and provide further incentives for their restoration and sustainable use. Efforts are underway to develop measures, including policies, to encourage the maintenance of the carbon storage function of these wetlands, many of which have been degraded or lost over recent decades.

A range of policies and practices have been introduced to support mitigation activities, including the development of an international fund for Reducing Emissions from Deforestation and Forest Degradation (REDD+) in developing countries. REDD+ includes policies and incentives for developing countries and includes the role of conservation, sustainable management of forests, and enhancement of forest carbon stocks. It also has immense potential for encouraging the management and restoration of wetlands now that it has been expanded to include activities that improve ecosystem services and provide co-benefits to communities and biodiversity (Alexander et al. 2011).

The IPCC is also paying increasing attention to wetlands in relation to greenhouse gases in response to an invitation by the UNFCCC. A Task Force on National Greenhouse Gas Inventories has prepared supplementary guidance to address, as far as possible, the gaps identified in earlier guidelines. These focus on those human activities and management that give rise to anthropogenic emissions or removals by wetlands and includes cross-cutting guidance on organic soils, the rewetting and restoration of peatlands, other freshwater wetlands, coastal wetlands, and constructed wetlands used for wastewater treatment. In 2013 the IPCC produced a supplement to the 2006 Guidelines for National Greenhouse Gas Inventories (Hiraishi et al. 2014).

Increasingly, coastal ecosystems have also been recognized for their role in carbon sequestration and, when degraded, their potential to become sources of carbon emissions. Progress has been made to include these systems in international and national policy and finance mechanisms. The full integration of coastal management activities as part of countries' portfolio of solutions to mitigate climate change has however not yet been fully achieved due to a lack of technical guidance leading to comparable information and scientific results. In response a manual containing standardized methods for field sampling, laboratory measurements, and analysis of blue carbon stocks and flux in coastal ecosystems has been developed.

Vulnerability of Wetlands to Climate Change

The Ramsar Convention on Wetlands has considered the vulnerability of wetlands to climate change and provided a framework for vulnerability assessment (Gitay et al. 2011) as part of its Integrated Framework for Wetland Inventory, Assessment and Monitoring (IF-WIAM). In line with the convention's focus on maintaining the ecological character of wetlands vulnerability assessment was presented "as an approach that can provide information and guidance for maintaining the ecological character of wetlands which are subject to adverse change as a consequence of climate change (including sea level rise), whilst recognising that climate change will interact with the many other anthropocentric pressures on wetlands." Vulnerability was then defined as "*the degree to which a wetland is sensitive to and unable to adapt to or moderate the consequences of climate change and other (anthropocentric) pressures on its ecological character*" and comprised two major components, namely the sensitivity of the system and its adaptive capacity or resilience. Sensitivity was defined as "*the degree to which a wetland is affected, either adversely or* *beneficially, by climate-related stimuli*" and adaptive capacity as "*the ability of a wetland to adjust to climate change, to take advantage of opportunities, or to cope with or moderate the consequences.*"

By bringing together various methods and approaches, a general framework for wetland vulnerability assessment was developed, comprising the following elements: (i) establishing the status of the biophysical and social components of the wetland, including present and recent pressures; (ii) determining the sensitivity and adaptive capacity of the ecological and social components of the wetland to multiple pressures and developing plausible futures; (iii) developing responses to ensure these futures can be achieved; and (iv) monitoring and adaptive management to reach the desired outcomes. Obtaining the necessary information to undertake an effective assessment may not be straightforward given the paucity of spatial and temporal data, at appropriate scales, as a time series to determine the present condition and trends in the condition of a wetland and its sensitivity and adaptive capacity. It is expected that in many instances the assessment will be an iterative process based on qualitative or subjective information and updated or improved as more information becomes available.

Impact of Climate Change on Wetlands

Climate change is likely to affect wetlands and their biota directly (for example, through increased temperatures) and indirectly (for example, through affecting the water regime). The anticipated impacts are expected to be more negative than positive and include (Finlayson et al. 2006): (i) an initial increase in biological production in some mid-latitude regions and a reduction in the tropics and subtropics; (ii) adverse effects on coastal wetlands, including freshwater lagoons, coral reefs, and mangroves; (iii) decreased water availability in many arid- and semiarid regions; (iv) adverse impacts on migratory and nomadic biota that depend on specific ecosystems for feeding and breeding; (v) immediate effects on wetlands in high latitudes and high altitudes as temperatures increase and the permafrost and glaciers, respectively, melt; and (vi) an increased threat of extinction for currently rare or vulnerable species and biota restricted to islands, peninsulas, or coastal areas. The ecological composition of most wetlands is likely to change, depending on the relative ability of the biota to disperse and establish in new locations. The extent and type of other pressures on wetlands is expected to greatly affect the rate of change in response to changes in the climate.

The following regional summary of the expected impact of climate change on wetlands is taken from Junk et al. (2013) and sources therein. More detailed information can be obtained for individual wetlands from the increasing amount of information that has been published over the past few decades, including analyses of the likely impact of sea level rise and larger storm events, and melting of the permafrost and glaciers.

North and Central America (Mitsch and Hernandez 2013)

Melting of the permafrost in the tundra is expected to result in major changes to many wetlands given increased temperature and changed rainfall and surface runoff and groundwater flows. The extent of change will determine the impact on wetlands with a decline in precipitation or an increase in evapotranspiration causing less-frequent flooding of existing wetlands but not necessarily the loss of wetlands. Increased precipitation would increase the length and depth of flooding of inland wetlands. Most susceptible are the many small wetlands, such as those in the prairie region that have very small watersheds and are located between the arid and the moist climate zones and which have seen extensive restoration given their importance for waterfowl breeding. Coastal wetlands may well be lost given limits on their ability to migrate inland, or adjacent salt marsh may be replaced by invading mangroves. The impact of hydrological change on the water regime and biogeochemical cycling in wetlands is likely to be large, but with uncertain outcomes, and affected also by multiple existing pressures. Changes in both boreal and tropical wetlands will affect the storage of carbon and release of carbon gases.

Europe (Čížková et al. 2013) and Russia (Robarts et al. 2013)

Annual precipitation is expected to increase in northern Europe especially in the winter months but decrease in most of southern Europe. Increased evapotranspiration may offset the increased water inputs. Summer drought is likely to be important for inland wetlands with the frequency and intensity most likely to increase from the north to the south. Sea level rise will likely have decisive outcomes for coastal wetlands especially along the Atlantic coast, with an expected increase in extreme sea level events along the continental North Sea coast. A generally wetter and warmer climate is expected for Russia with the highest increase in annual and winter mean precipitation in boreal Asia resulting in an increase in annual runoff in the major Siberian rivers. Annual precipitation is expected to increase the most in northern Russia with increases in both summer and winter. West Siberia is warming faster than the Arctic as a whole and it also contains the world's largest stores of peat with expected large increases in dissolved organic carbon in rivers and significant impacts on aquatic biogeochemical cycles and biological productivity.

South America (Junk 2013)

Temperature increases are expected to be greatest in the continental regions, including the central Amazon, but there is insufficient data or information about the effect on wetland species. It is expected that the species composition of many wetlands may change, but given the wide range of many species this is unlikely to result in mass extinctions. Temperate and boreal species may shift to higher latitudes and to higher altitudes in mountainous regions, and mangroves may move further to the north and south. However, while the species composition may change with increased temperature, the tropical and subtropical wetlands are expected to remain at the same locality as long as hydrological conditions are suitable. A major threat will be the increase of fires, especially in periodically flooded savannahs such as the Pantanal and the Araguaia River floodplain where flood adapted tree species are sensitive to fire. Further fires in the grassland that have resulted as a consequence of land clearing may extend the impact into adjacent wetlands. There are contradictory views on the effects on the Amazon rainforests, about one third of which is periodically flooded or waterlogged. There are predictions that the Amazon rain forest may change to a semi-deciduous forest with a dense network of evergreen riparian and floodplain forests. Accelerated melting of snow and ice, including from glaciers in Patagonia and in the Andes, will reduce the amount of water stored in these areas. Accelerated snowmelt in the Andes could change the wet-dry periodicity in large rivers with large changes in the discharge and flood pulses. Rising sea levels will affect coastal wetlands that in many areas have limited room to migrate given the extent of human infrastructure and development. The biota in the lower reaches of many large rivers will likely be affected by sea level rise and an extension of tidal movement upstream. Stronger El Nino and La Nina events will strongly affect the rainwater-fed floodplains and swamps.

Africa (Mitchell 2013)

Precipitation is projected to increase over the equatorial region of Africa, but in the mid-latitude regions, over the deserts, it is projected to decrease and result in less surface water being available. The Mediterranean coast of North Africa will become drier. A band running west to east, which includes parts of the Sahel and areas to the south of it, is predicted to have increasing water resources. Under a scenario of moderate climate change, severe future threats to both freshwater ecosystem functioning as well as water availability for humans may be expected, as for instance for south-western part of Africa, e.g., the Namib and Kalahari deserts. This will be exacerbated by the accompanying changes in the frequencies and magnitudes of floods and droughts. The eastern part of South Africa, particularly east of the escarpment, will receive increased precipitation while the western coast will become drier. But, there are considerable disagreements between the predictions of different models. It is possible that the resilience of many ecosystems may be exceeded this century by an unprecedented combination of climate change and associated disturbances, such as floods, drought, and other global change drivers like land-use change, pollution, fragmentation of natural systems, and overexploitation of resources. Another predicted effect of increasing temperature is that the distribution of tropical diseases such as malaria and bilharzia will change and include areas currently free of these diseases.

Asia (An et al. 2013; Gopal 2013)

The expected rise in temperature across Asia is expected to be greater in South and Central Asia than in Southeast Asia with warmer winters also occurring in the south. Temperature increases in the Himalaya will accelerate the melting of glaciers with changes to the water regimes of the rivers and associated wetlands. Precipitation from the monsoon is expected to increase annually but with more temporal and spatial variability, with fewer rainy days and higher intensity of extreme events. Heavy rainfall events are projected to increase in the western and eastern parts of south Asia, and summer precipitation to decline over the semi-arid western central Asia but increase in South and Southeast Asia. The impact of sea level rise is expected to have a major impact on the mangroves and coastal freshwater or brackish wetlands throughout the region, including those in the deltas of large rivers. Warming has also reduced the water-storage capacity of the Qinghai-Tibet Plateau with consequences for the flow of water into some of the major rivers of the world, including the Yangtze, Mekong, and Ganges. The loss of glaciers and wetlands in high altitude areas is expected to continue, but further changes in precipitation and water flows confounds efforts to predict what changes are likely.

Australia (Finlayson et al. 2013)

Rainfall is expected to decrease across many parts of the continent with more dry days but more intense storms when it does rain, including for tropical cyclones. Drought and evaporation rates are expected to increase, as is the frequency of fire. There is an expectation that across much of southern Australia it will be slightly warmer with lower and more variable rainfall. Such changes could exacerbate the extent of rainfall variability already experienced. Based on the experience of a recent decade long drought this could cause major changes in water flows and inundation of wetlands and affect the dispersal and breeding of many migratory or nomadic species. Prolonged drying of riverine and floodplain environments could lead to the development of acidic soils, as experienced during the drought years. The vulnerability of coastal wetlands in northern Australia to climate change and sea level rise has been assessed and is expected to result in substantive change to the biophysical character of the wetlands. Sea level rise, shoreline erosion, and saltwater intrusion could change both saline and freshwater wetlands and enable mangroves to invade coastal marshes. Coral reefs are at risk from further bleaching and acidification of the oceans.

Impact on Waterbirds

The general nature of the impacts of climate change on waterbirds is described below based on a summary provided by Finlayson et al. (2006) who noted that although some changes can be identified the exact extent, intensity, and time frames of such changes are difficult to predict. The most severe effects of climate change on waterbirds come about because of: (i) the loss and degradation of inter-tidal habitats; (ii) increased salinization of coastal freshwater wetlands; (iii) a reduction in the extent of wetlands and duration of flooding in arid and semi-arid areas; and (iv) the loss of wetland breeding habitats in the arctic and subarctic. Many of these changes are likely to occur relatively quickly as sea levels rise and temperatures increase.

The extent of loss and degradation of coastal wetlands and its effects on coastal waterbirds will depend on the ability of coastal environments to move inland as sea level rises. Where this is possible the area of inter-tidal wetlands might even expand; however, where coastal infrastructure is well established and where sea defenses are strengthened the area of inter-tidal habitats is likely to be reduced. Further, the length of time that the remaining inter-tidal areas are exposed during each tidal cycle may be reduced, affecting the time available for shorebirds to feed. The time shorebirds spend feeding during low tide varies between species and depends on their body size and ambient temperatures and increases when they are accumulating reserves prior to migration. As smaller species spend almost all of their time feeding they could potentially be the most seriously affected. Some inter-tidal areas will probably become unusable by the smaller species. Expected reductions in the metabolic rates of some species as a consequence of rising temperatures are unlikely to offset the loss of feeding time. Reduced feeding could also result in reduced survival during winter and during migration. Further, reduced body condition on arrival at the breeding grounds in the Arctic might reduce the breeding success of migratory species. Rising temperatures will also cause changes in the abundance and growth of the inter-tidal invertebrate populations that constitute the prey of shorebirds.

The effects of rising temperatures on plant communities in the Arctic may be substantial and, among other changes, lead to an expansion of boreal forest into current tundra areas that are important for breeding waterbirds, such as many goose species and a large proportion of calidrid sandpipers. While an increase in temperatures could increase productivity, which could potentially compensate to some extent for habitat loss, it may be offset by a likely increase in loss of nests and chicks to predation as predators, such as the red fox, expand their range. This example points to the complex ecological interactions that may occur and affect the survival of breeding populations in specific areas.

Climate change is also expected to result in changes to rainfall patterns, with some areas experiencing increases and others decreases or more intense events. Many currently dry areas, such as in parts of Australia, Asia, and Africa, may experience more rainfall variability and drier conditions. As many wetlands and waterbird populations in these areas are already highly stressed from habitat loss due to agriculture, including reduced water flows caused by water extraction for irrigation, pollution, and increasing salinization, these changes may exacerbate current pressures. This may impact on waterbird populations in drier environments that are characterized by unpredictable and relatively infrequent breeding, high adult survival, and high mobility. Successful breeding in such areas often requires exceptional flooding events that last long enough for completion of the breeding cycle. Reduced rainfall could increase the intervals between flooding events as well as shorten their duration. Reduced breeding success and recruitment should be expected as a first response to such changes, followed by reduced survival of adults and then widespread population declines. There will almost certainly be a great deal of regional variation, with some areas experiencing increases in waterbird populations and others, decreases, although the current concern has focused on the latter.

A recent analysis of 52 breeding migratory shorebird taxa in North America found that 45 were predicted to be more at risk of extinction when climate change was added to existing risks (Galbraith et al. 2014). The increased risk was due to loss of (i) breeding habitat; (ii) coastal and inland migration stopover habitats due to sea level rise and drought; and (iii) loss of coastal wintering habitat due to sea level rise. For migratory bird species, and in particular long-distance migrants, further effort is needed to determine: (i) the effects of habitat alteration at stopover points and wintering areas; (ii) the amounts and locations of major habitat types, as well as the carrying capacities of these habitats; (iii) the behavior, home range, and resource needs of species most likely to be vulnerable to global warming; and (iv) suitable policies and legislation at a continental scale, or better, at the flyway scale where possible to respond to adverse changes.

Adaptation

The IPCC has defined adaptation as "initiatives and measures to reduce the vulnerability of natural and human systems against actual or expected climate change effects" although others have extended this to also include the ability of a system to cope with environmental stress, including that from climate change. In this respect, adaptation fits comfortably with the Ramsar Convention's wise use approach that underpins global efforts to maintain or restore the ecological character of wetlands. Specifically, wise use of wetlands is defined as "the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development." (Finlayson et al. 2011).

While the convention provides guidance for the wise use of wetlands through the Wise Use Handbooks (Ramsar Secretariat 2010), it does not provide specific guidance on responding to the ecological consequences of climate change. In response to this situation, Finlayson et al. (2013) suggested areas where the convention could develop guidance applicable to climate change as part of its wise use approach, including: (i) assessing changes in the distribution of species and whether these constitute a change in the ecological character of the wetland; (ii) assessing the usefulness of models of wetland response to climate change; (iii) assessing the value in allocating water to protected sites where restoration would be contingent on reallocation of larger volumes of water; (iv) assessing the efficacy of engineering responses with the potential to deliver more water-efficient environmental outcomes for wetlands; and (v) determining if the description of the ecological character of a Ramsar site at the time of listing is a suitable reference for management purposes.

Other approaches to adaptation have considered the adequacy of current management approaches and assessed these to determine if they were appropriate under climate change or represented maladaptation. This approach takes advantage of existing knowledge and understanding of wetland management and places it within climate scenario as a way of ascertaining what may or may not be appropriate responses given local circumstances and opportunities. Lukasiewicz et al. (2013) investigated the potential of a number of existing management approaches, including the use of environmental flows and engineering works and measures, for adaptation and concluded that there were many activities underway that, if extended and linked, would comprise a substantial ecosystem-based approach to adaptation.

While adaptation measures have the potential to reduce the adverse effects of climate change it is unlikely that they can prevent all impacts. They can promote conservation and the sustainable use of biodiversity and reduce the impact of changes in climate and climatic extremes on biodiversity. Further, the effectiveness of adaptation activities can be enhanced when they are integrated within broader strategies designed to make development more sustainable, although the adoption of an ecosystem-based approach would likely be constrained by institutional complexity and socioeconomic issues.

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74 Climate Change: United Nations Framework Convention on Climate Change (UNFCCC) and Intergovernmental Panel for Climate Change (IPCC)

C. Max Finlayson

Contents

Abstract

This chapter provides a general outline of the features of the United Nations Framework Convention on Climate Change (UNFCCC) and the Intergovernmental Panel for Climate Change (IPCC) and their relevance to wetlands. General features of the impact of climate change on wetlands are not considered.

Keywords

Climate · UNFCC · IPCC · Adaptation · Mitigation

Introduction

This chapter contains a general outline of the features of the United Nations Framework Convention on Climate Change (UNFCCC) and the Intergovernmental Panel for Climate Change (IPCC) and their relevance to wetlands. General features of the impact of climate change on wetlands are not considered.

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United Nations Framework Convention on Climate Change (UNFCCC)

The UNFCCC is one of three intergovernmental conventions adopted at the Rio Earth Summit in 1992, along with the UN Convention on Biological Diversity and the Convention to Combat Desertification. It came into force on 21st March 1994 and now has 195 contracting parties. The general information on the convention that is presented below largely comes from the material presented on the website https://unfccc.int/2860.php.

The general purpose of the convention is to cooperatively consider what could be done to limit average global temperature increases and the resulting climate change and to cope with whatever impacts may occur. The ultimate aim is to prevent "dangerous" human interference with the climate system and to stabilize greenhouse gas concentrations "*... at a level that would prevent dangerous anthropogenic (human induced) interference with the climate system*" (https://unfccc.int/). The convention further anticipates that "*... such a level should be achieved within a time-frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened, and to enable economic development to proceed in a sustainable manner.*"

The convention puts the responsibility on developed countries to lead the international effort to meet these aims, given that they have been responsible for most of the past and current greenhouse gas emissions and are therefore expected to do the most to cut emissions. This is, at times, a contentious issue given the increase in emissions from some developing countries. The developed countries are referred to as Annex I countries and belong to the Organization for Economic Cooperation and Development (OECD). They were expected by the year 2000 to reduce emissions to 1990 levels; some of them have succeeded. Developed countries have also agreed to support climate change activities in developing countries through financial assistance to address specific climate change actions and to share appropriate technology.

Annex 1 countries are required to report regularly on their climate change activities and policies, including the measures outlined under the Kyoto Protocol if they have formally accepted it. An inventory of their greenhouse gas emissions, including for 1990 (the baseline for this purpose) and all years since is required annually. The requirements on developing countries are less arduous with reports on their actions to: (i) mitigate climate change and (ii) adapt to the impacts. This reporting is contingent on obtaining funding to prepare the reports, especially for the least developed countries. The Kyoto Protocol is an agreement linked to the UNFCCC which commits the contracting parties to set and meet binding emission reduction targets (https://unfccc.int/kyoto_protocol/items/2830.php). The Protocol was adopted in December 1997 and came into force in February 2005 with detailed rules, known as the Marrakesh Accords, being agreed in 2001. The protocol places a heavier requirement on developed nations to reduce emissions under the principle of common but differentiated responsibilities, with the first commitment period from 2008 to 2012 and the second from 2013 to 2020. The first commitment period included 39 industrialized countries and the European Community that committed to reduce GHG emissions to an average of five percent against 1990 levels. The second period contains a commitment to reduce emissions by at least 18% below 1990 levels. Countries are expected to meet their targets mainly through national measures or through three market-based mechanisms, namely: international emissions trading; clean development mechanism; and joint implementation. Emissions have to be monitored and records kept of all emissions trading.

The convention emphasizes that all countries are vulnerable to the effects of climate change and calls for special efforts to ease the consequences, especially in developing countries. This led to the consideration of approaches for assessing the vulnerability of both socioeconomic and ecological systems to climate change, with the Ramsar Convention on Wetlands providing a method for assessing the vulnerability of wetlands to climate change (Gitay et al. 2011) and requesting further investigation of the socioeconomic vulnerabilities of communities dependent on wetlands for at least part of their livelihood. Further effort has also been afforded to adaptation to climate change, with an emphasis on observation, assessment of climate impacts and vulnerability, planning and implementation, and monitoring and evaluation of adaptation actions. Lukasiewicz et al. (2013) have highlighted that current management practices could enable adaptation to climate change, but under some conditions these could prove to be maladaptive.

The convention also requires contracting parties to mitigate climate change, reduce emissions, and enhance carbon sinks. A range of policies and practices has been introduced including the development of an international fund for Reducing Emissions from Deforestation and Forest Degradation (REDD+) in developing countries. It embraces policies and incentives for developing countries and includes the role of conservation, sustainable management of forests, and enhancement of forest carbon stocks in developing countries. It also has immense potential for encouraging the management and restoration of wetlands now that it has been expanded to include activities that improve ecosystem services and provide co-benefits to communities and biodiversity (Alexander et al. 2011).

Intergovernmental Panel on Climate Change (IPCC)

The Intergovernmental Panel on Climate Change (IPCC) was established in 1988 by the United Nations Environment Programme (UNEP) and the World Meteorological Organization (WMO) to provide a scientific assessment of knowledge about climate change and possible environmental and socioeconomic impacts. Later the same year, the General Assembly of the United Nations endorsed the establishment of the IPCC. The information that follows has been largely derived from the IPCC web pages (http://www.ipcc.ch/).

The IPCC is an intergovernmental body open to all member countries of the United Nations (UN) and WMO. There are currently 195 member countries. The purpose of the IPCC is to review and assess the most recent technical and socioeconomic information about the cause, extent, and consequences of climate change to support the formulation of realistic responses. It does not undertake research nor monitor changes in the climate. Governments participate in the review process and the plenary sessions where decisions are made about the work program and assessment reports. The plenary session also elects the IPCC Bureau Members, including the Chair. A Secretariat is hosted at the WMO in Geneva, coordinating the work of the IPCC and liaising with governments.

A large number of scientists voluntarily contribute to the assessments undertaken by the IPCC. These assessments are based on extensive reviews in order to ensure a complete and objective coverage of current information. The initial task given to the IPCC was the preparation of a comprehensive review and recommendations about "*. . .the state of knowledge of the science of climate change; the social and economic impact of climate change, and possible response strategies and elements for inclusion in a possible future international convention on climate.*" Today its role is "*. . .to assess on a comprehensive, objective, open and transparent basis the scientific, technical and socio-economic information relevant to understanding the scientific basis of risk of human-induced climate change, its potential impacts and options for adaptation and mitigation. IPCC reports should be neutral with respect to policy, although they may need to deal objectively with scientific, technical and socio-economic factors relevant to the application of particular policies.*"

The first IPCC Assessment Report in 1990 underlined the importance of climate change as a challenge requiring international cooperation to tackle its consequences. This played a key role in the formation of the United Nations Framework Convention on Climate Change (UNFCCC). Since then, the IPCC has delivered regular scientific reports about climate change produced worldwide. Five assessment reports have now been produced (http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml#1):

First Assessment Report – FAR – 1990
Second Assessment Report – SAR – 1995
Third Assessment Report – TAR – 2001
Fourth Assessment Report – AR4 – 2007
Fifth Assessment Report – AR5 – 2013/14

Wetlands have been considered in the assessment reports with information spread across many chapters, including regional assessments and those addressing ecosystem services. The information on wetlands from the TAR was extracted and provided as a report to the Ramsar Convention's 7th conference of parties in 2002 in support of a formal resolution on climate change and wetlands (van Dam et al. 2002).

The IPCC also provides scientific and technical information to the UNFCCC, such as methodological reports and guidelines to help parties prepare national greenhouse gas inventories. In 2013, the IPCC produced a supplement to the 2006 Guidelines for National Greenhouse Gas Inventories (Hiraishi et al. 2014). The Second Assessment Report provided information that was used to support the adoption of the Kyoto Protocol in 1997. The protocol is an international agreement associated with the UNFCCC. It came into force in February 2005 and contains detailed rules, known as the Marrakesh Accords, for setting internationally binding emission reduction targets (https://unfccc.int/kyoto_protocol/items/2830.php).

The Ramsar Convention has participated in meetings of the IPCC and has a keen interest in the impact of climate change on wetlands and the potential contribution of wetlands to the mitigation of climate change. In addition to developing a method for assessing the vulnerability of wetlands to climate change, including the impacts of sea level rise, the convention has addressed adaptation to climate change through its wise use principles (Finlayson 2013) with a current interest in ecosystem-based adaptation which is compatible with wise use (Finlayson et al. 2011). It has further taken an interest in the potential of wetlands to sequester carbon to mitigate climate change, including in peatlands and coastal wetlands (Lloyd et al. 2013).

The IPCC and Albert Arnold (Al) Gore Jr. were awarded the Nobel Peace Prize "*...for their efforts to build up and disseminate greater knowledge about human-made climate change, and to lay the foundations for the measures that are needed to counteract such change*" (http://nobelpeaceprize.org/en_GB/laureates/laureates-2007/announce-2007/). By making the award, the Norwegian Nobel Committee sought to sharpen the focus on the decisions and mechanisms needed to reduce the threat of climate change and benefit humankind.

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Reducing Emissions from Deforestation and Forest Degradation

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Sasha Alexander

Contents

Abstract

Reducing emissions from deforestation and forest degradation (REDD) was first introduced in 2005 at the 11th meeting of the Conference of the Parties (COP11) of the UN Framework Convention on Climate Change (UNFCCC) in order to establish an international funding mechanism for reducing carbon emissions and protecting forest ecosystems. At that time, REDD was envisioned as a fund-based and/or market-based approach to assist the developing countries in meeting their emission obligations under the Clean Development Mechanism (CDM), where the contribution of conservation, afforestation, and reforestation (ecological restoration) to reducing emissions was first recognized. In 2007 (UNFCCC COP15), the Bali Action Plan was agreed to, and the REDD mechanism was expanded to include the sustainable management of forests and enhancement of forest carbon stocks and is now known as REDD+.

Keywords

REDD+ · Ecological restoration · Wetland forests · Carbon stocks · Nested governance

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Background

Reducing emissions from deforestation and forest degradation (REDD) was first introduced in 2005 at the 11th meeting of the Conference of the Parties (COP11) of the UN Framework Convention on Climate Change (UNFCCC) in order to establish an international funding mechanism for reducing carbon emissions and protecting forest ecosystems. At that time, REDD was envisioned as a fund-based and/or market-based approach to assist the developing countries in meeting their emission obligations under the Clean Development Mechanism (CDM), where the contribution of conservation, afforestation, and reforestation (ecological restoration) to reducing emissions was first recognized.

The degradation and loss of terrestrial and wetland forests has become a significant challenge for natural resource management policies and strategies that aim to mitigate, and foster adaptation to, climate change impacts. The Intergovernmental Panel on Climate Change (IPCC) concluded that "reducing and/or preventing deforestation and preventing the release of carbon emissions into the atmosphere is the mitigation option with the largest and most immediate carbon stock impact in the short-term per hectare and per year globally" (Metz et al. 2007).

In 2007, at the UNFCCC COP15, the Bali Action Plan was agreed to, and the REDD mechanism was expanded to include the sustainable management of forests and enhancement of forest carbon stocks and is now known as REDD+ (or REDDplus). In response to the processes outlined in the Bali Action Plan, the UN-REDD Programme and the World Bank Forest Carbon Partnership Facility (FCPF) were established in 2008 to financially help the developing countries to build capacity, broaden stakeholder engagement, and provide overall support for their REDD+ readiness efforts.

The Cancun Agreements

In December 2010, at the UNFCCC COP17, the almost unanimous Cancun Agreements crystallized the previously negotiated REDD+ language which now embraces "policy approaches and positive incentives on issues relating to reducing emissions from deforestation and forest degradation in developing countries; and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries" (UNFCCC 2010). Bolivia's dissent in Cancun revolved around their opposition to using a market-based approach (carbon credits) to finance REDD+, thus delaying a final decision and continuing to rely on bilateral and multilateral funding until the UNFCCC COP18.

Furthermore, the Cancun Agreements include the creation of a Green Climate Fund and guidance/safeguards for potential REDD+ donors and recipients during the so-called "fast-start finance" period in preparation for the post-2012 climate change agreements. However, until the financing and monitoring (MRV) mechanisms have been agreed upon, all potential REDD+ projects will continue to remain outside the

UNFCCC. Although the primary goals of REDD+ are to reduce emissions and sequester carbon in forest ecosystems, the Cancun Agreements include activities that also provide multiple co-benefits to biodiversity, ecosystem functions, and local and indigenous communities (Dickson and Osti 2010).

Implications for Forested Wetlands

Forest conservation, management, and restoration can lead to reduced emissions and increased carbon sequestration; however the magnitude and timescale of the response varies by ecosystem type. The sustainable management of and enhancement of carbon stocks in forested wetlands (mangrove, peatland, and bottomland forests) are considered to make relatively large contributions to global emission reductions in the long term. Although these wetland ecosystems represent only a small percentage of the world's forests, they are some of the most productive in terms of carbon storage and other vital ecosystem services. This is largely a result of high aboveground biomass or standing carbon stocks and their capacity to store carbon belowground in soil and sediment.

For example, the average aboveground biomass in mangrove forests is 247.4 t ha^{-1} (similar to that of tropical terrestrial forests), while carbon burial averages 181.3 gC m^{-2} $year^{-1}$ or a total of 29.0 TgC $year^{-1}$ (Alongi 2009). This significant amount of long-term carbon storage suggests that REDD+ funding for conservation, management, and restoration activities in these forested wetland ecosystems could reduce emissions and increase global carbon storage, perhaps even more than upland forests on a per hectare basis (Laffoley and Grimsditch 2009). The tangible co-benefits of revitalized mangrove forests extend to local and indigenous communities that depend on their goods and services (e.g., timber, fisheries, water treatment, and storm/climate protection).

Future Challenges

It is important to recognize that, in practice, the multiple objectives of REDD+ (carbon, biodiversity, ecosystems, and communities) are not always synergistic or complementary and that, in fact, there are often significant and, some may argue, unnecessary trade-offs. The lack of good governance is perhaps the single greatest factor inhibiting these seemingly natural synergies or co-occurrences: government policy and regulation often segregate and skew outcomes by focusing on short-term benefits for a few, creating further inequity for many; social and consumer priorities when translated into action on the ground often fail to recognize the vital connection between biodiversity, ecosystem services, and sustainability; and the marginalization and limited land tenure rights of local and indigenous communities often tend to reduce stakeholder involvement and access to benefits (Skutsch and McCall 2010).

Measuring, **Reporting**, **and Verification** (**MRV**):In order to be eligible for funding, REDD+ activities will need to demonstrate cost-effectiveness, additionality (i.e., would be otherwise unfunded), and the capacity for, and verification of, long-term carbon sequestration. However, there are still a number of important unresolved issues related to carbon leakage, reporting, and verification. For example, a nation can opt out of reporting soil carbon emissions if it is determined that they are not substantial. Although MRV systems are currently being developed to provide the technical support needed to assist developing nations in designing and implementing REDD+ projects, there is a need for greater input on establishing parameters for carbon, biodiversity, ecosystem, and socioeconomic indicators.

Informal Forestry Sector and Subnational Integration: The informal forestry sector is one that evolves in response to scarce resources or seeks to operate outside regulations that often disfavor local and indigenous communities and incentivize unsustainable uses. The informal sector is usually community based and not subject to the same systematic private/public management found within the formal forestry sector (e.g., timber concessions, national parks, and other protected areas). An important challenge for forest restoration will be the inclusion of this sector in REDD+ national strategies or action plans so as to ensure effective stakeholder participation, the integration of traditional and community forestry management practices, and a commitment to low-emission rural development. In addition to the informal sector considerations, it is critical that subnational activities be recognized and effectively incorporated into these national strategies or action plans as many potential REDD+ activities are under the jurisdiction of local and state/provincial authorities.

Benefits to Local and Indigenous Communities: Yet another challenge to overcome is safeguarding the rights and land tenure of local and indigenous communities and acquiring their full participation, when appropriate, as key stakeholders in REDD+ activities. Local and indigenous communities are often the traditional stewards of forest-based ecosystem services that not only benefit them but the rest of the world. Policy approaches to, and positive incentives for, reducing emissions, like REDD+, will undoubtedly increase the value of local community and indigenous lands. Thus, clear tenure rights (land titling) would be a cost-effective component of REDD+ national strategies or action plans to reduce emissions and provide multiple co-benefits to biodiversity, ecosystems, and communities. One idea that seems to hold great potential for the equitable treatment and involvement of local and indigenous communities is that of nested governance whereby there is a hierarchy of decision-making processes and multi-sectoral coordination that accounts for REDD+ activities at all spatial scales, from the global to the local (Sikor et al. 2010).

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Management and Sustainable Development of Wetlands

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Robert J. McInnes

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Abstract

Wetland management has over the past few decades shifted strongly toward a focus on ensuring the sustainable development of individual sites and suites of wetlands across landscapes. This considers the multiple uses of wetlands and the increasing demands for the ecosystem services they provide. The concept of sustainable development has a long history in both the development and environmental literature and can be considered as development that meets the needs of the present without compromising the ability of future generations to to meet their owns needs. The Ramsar Convention on Wetlands recognizes that the conservation and wise use of all wetlands contributes towards achieving sustainable development throughout the world. Therefore, sustainable development is at the heart of the global wise use of wetlands.

Keywords

Sustainable development · Brundtland report · Ramsar convention · Sustainable development goals (SDGs) · Millennium development goals (MDGs)

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Introduction

Wetland management has over the past few decades shifted strongly toward a focus on ensuring the sustainable development of individual sites and suites of wetlands across landscapes. This considers the multiple uses of wetlands and the increasing demands for the ecosystem services they provide, as expressed through the Millennium Ecosystem Assessment (2005) and other publications. The concept of sustainable development has a long history in both the development and environmental literature. However, a key landmark was achieved in 1984, when the United Nations established an independent group of 22 people drawn from developing and developed member states in order to identify long-term environmental strategies for the global community. In 1987, the World Conference on Environment and Development published their report entitled "Our Common Future" (WCED 1987), often referred to as the "Brundtland Report," with respect to the chairperson, the then Norwegian Prime Minister, Gro Harlem Brundtland. The term "sustainable development" was widely used throughout the report and was defined as:

> Development that meets the needs of the present without compromising the ability of future generations to meet their own needs.

The report is commonly considered to represent a watershed in terms of environmental awareness and placed sustainable development into the political arena of international development thinking (Sneddon et al. 2006). Since publication, the report has been translated into more than 20 languages and, despite the fact that there are many different definitions emanating from various disciplines each with different assumptions about the basic relationship between society and nature, its definition of sustainable development continues to be widely used. This reflects a common understanding that humanity has the ability, understanding, and technology to make development sustainable to ensure that the needs of the present generation are met without compromising the ability of future generations to meet their own needs.

The concept of sustainable development implies that there are limitations imposed by the current knowledge, technology, and societal organization capabilities on environmental resources and the capacity of the biosphere to tolerate the negative impacts of human activities. But it has been argued as far back as Thomas Malthus (1872) that technology and social organization can be both managed and improved to facilitate progressive economic growth. However, today sustainable development is being pursued in an increasingly globalized world, which still exhibits a high degree of poverty (Chany and Gertz 2011) and increasing rates of environmental degradation and biodiversity decline (Butchart et al. 2010), including for wetlands (Davidson 2014; Gardner et al. 2015). The global challenge for sustainable development lies in understanding the complex and evolving interdependencies of environment, social and economic development, and the ability to understand the conceptual, contextual, and geopolitical dimensions in order to achieve the future perceived by the Brundtland Report. This is as important for wetlands as it is for other ecosystems especially given the high rates of wetland loss globally (Davidson 2014).

Sustainable Development Challenges

While numerous definitions of sustainable development exist, most encompass the idea that it is comprised of three interdependent pillars, namely, ecological, economic, and social. (In many cases the words ecological and environmental are used interchangeably, although the latter can also be used to encompass ecological, social, and economic components.) The differences and nuances contained in definitions used by different sectors of society are important as they can indicate differences in the way that sustainability is viewed and the means for achieving sustainable development. Difficulties can arise as sustainable development, and *sustainability* are complex and multidimensional concepts, which often seeks to combine efficiency and equity based on social, economic, and environmental elements (Ciegis et al. 2015). These differences can become increasingly important when transferred to the policy development arena. Even the seemingly simple definition in the Brundtland Report, which separates the needs of present and future generations, implies that future needs are known and that any limits that are applied today will meet the demands of tomorrow. Similarly, "needs" may mean many different things to different societies and people, and some future needs may remain unknown. Therefore, ambiguities emerge in terms of the legacy that one generation should leave for its progeny. Often this is considered in terms of natural capital, but it may also embrace cultural, technological, or social elements.

Setting Global Targets for Sustainable Development

The United Nations Conference on Environment and Development, the "Earth Summit," which took place in Rio de Janeiro, Brazil in 1992, marked a significant moment in the course of sustainable development. One of the central aims was to identify the principles of an agenda for future action toward sustainable development with, for the first time, consideration of environmental sustainability and specifically biodiversity. However, tensions were in evidence between developed and developing countries and the different needs for exploitation of resources in order to deliver economic benefits. However, the post-Rio development of international treaties, such as the Convention on Biological Diversity (CBD) and the United Nations Framework Convention on Climate Change (UNFCCC), brought with them the positive ambition to foster the coordination of markets and public policies at the global level while maintaining a commitment to sovereignty in order to deliver a more sustainable future.

In September 2000, to mark the new millennium, 147 heads of state met at the United Nations Headquarters to consider global social and environmental priorities. To elucidate their ambitions and to evaluate progress, a set of numerical targets and deadlines were established. These were called the Millennium Development Goals

(MDGs) (http://www.un.org/millenniumgoals/). All United Nations member states at the time, and a variety of international organizations, committed to help achieve the following MDGs by 2015:

1. To eradicate extreme poverty and hunger
2. To achieve universal primary education
3. To promote gender equality
4. To reduce child mortality
5. To improve maternal health
6. To combat HIV/AIDS, malaria, and other diseases
7. To ensure environmental sustainability
8. To develop a global partnership for development

Each goal was given specific targets and deadlines. Target 7 was divided into the following four components which captured elements of sustainable development including integrating the principles of sustainable development into country policies and programs and reversing the loss of environmental resources and reducing biodiversity loss by 2010 in order to achieve a significant reduction in the rate of loss.

- Target 7.A: Integrate the principles of sustainable development into country policies and programs and reverse the loss of environmental resources.
- Target 7.B: Reduce biodiversity loss, achieving, by 2010, a significant reduction in the rate of loss.
- Target 7.C: Halve, by 2015, the proportion of the population without sustainable access to safe drinking water and basic sanitation.
- Target 7.D: Achieve, by 2020, a significant improvement in the lives of at least 100 million slum dwellers.

In the report on progress toward the MDGs published 1 year before the achievement is due (United Nations 2014), there were some alarming failures with regard to delivery on sustainable development objectives described in Target 7. For instance, emissions of carbon dioxide (CO_2) were observed to continue to rise globally, water scarcity perpetuated in Northern Africa and parts of Western Asia, and starkly the report stated that overall, species populations were declining and distributions retracting and, hence, some were moving faster toward extinction (Ibid.). Of all the earth's ecosystems and species they support, the overall picture was most bleak for wetlands (Butchart et al. 2010). Consequently, question marks are raised regarding the ability of the global community to fully commit to achieving sustainable development and to integrate commitments under different treaties and obligations. A criticism made of implementation of the MGDs in Latin America and the Caribbean suggested that the 2010 Biodiversity Target represented a conservation goal that did not integrate wider sustainable development requirements, and while it catalyzed increased cooperation in the biodiversity cluster of multilateral environmental agreements, it failed to align policies and implementation activities around the CBD's sustainability principles (Gomar 2014).

In June 2012, 20 years after the "Earth Summit," world leaders, along with representatives from the private sectors and nongovernmental organizations came together to discuss "the future we want" at the Rio+20 United Nations Conference on Sustainable Development. The discussions focused on two main themes: how to build a green economy in order to deliver on sustainable development while lifting people out of poverty and how to improve global coordination for sustainable development. Through the outcome document "The Future We Want," the conference set out the mandate to develop the successors to the MDGs, which expire at the end of 2015, termed the Sustainable Development Goals (SDGs).

The United Nations published the zero order draft of the proposed SDGs in June 2015 (https://sustainabledevelopment.un.org/content/documents/7261Post-2015%20Summit%20-%202%20June%202015.pdf). The SDGs are an agenda for people, planet, and prosperity that seek to strengthen universal peace and shift the world on to a sustainable path. The agenda defines a sustainable future where the planet is protected and natural resources, including wetlands, are used sustainably. The zero order draft document acknowledges the progress made through the MDGs but accepts that progress has been uneven and some goals were not achieved. Therefore at the heart of the SDGs is the desire to finish what the MDGs failed to complete.

World leaders have pledged common action and endeavor across a broad policy agenda which has been captured in 17 goals (below) and 169 associated targets in an ambition to achieve a path toward sustainable development. The SDGs will come into effect on 1 January 2016 and will guide decision-making over the ensuing 15 years. The SDGs, along with the associated targets, are intended to be integrated and indivisible, global in nature and universally applicable. They take into account the realities of different national circumstances, capacities, and levels of development and also pay respect to sovereign policies and priorities. Targets are defined as aspirational and global, and it is the responsibility of each government to establish its own national targets based on the overarching global level of ambition.

- **Goal 1**. End poverty in all its forms everywhere.
- **Goal 2**. End hunger, achieve food security and improved nutrition, and promote sustainable agriculture.
- **Goal 3**. Ensure healthy lives and promote well-being for all at all ages.
- **Goal 4**. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all.
- **Goal 5**. Achieve gender equality and empower all women and girls.
- **Goal 6**. Ensure availability and sustainable management of water and sanitation for all.
- **Goal 7**. Ensure access to affordable, reliable, sustainable, and modern energy for all.
- **Goal 8**. Promote sustained, inclusive, and sustainable economic growth, full and productive employment, and decent work for all.
- **Goal 9**. Build resilient infrastructure, promote inclusive and sustainable industrialization, and foster innovation.
- **Goal 10**. Reduce inequality within and among countries.

- **Goal 11**. Make cities and human settlements inclusive, safe, resilient, and sustainable.
- **Goal 12**. Ensure sustainable consumption and production patterns.
- **Goal 13**. Take urgent action to combat climate change and its impacts.
- **Goal 14**. Conserve and sustainably use the oceans, seas, and marine resources for sustainable development.
- **Goal 15**. Protect, restore, and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss.
- **Goal 16**. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all, and build effective, accountable, and inclusive institutions at all levels.
- **Goal 17**. Strengthen the means of implementation and revitalize the global partnership for sustainable development.

Unlike the MDGs, wetlands have a high profile within the SDGs. Goal 6 seeks to ensure availability and sustainable management of water and sanitation for all. Within the goal there is a clear acknowledgment to protect and restore water-related ecosystems which include wetlands. Similarly, within Goal 15, which seeks to protect, restore, and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss, the need to conserve, restore, and sustainably use wetlands in line with obligations under international agreements, such as the Ramsar Convention on Wetlands, is clearly stated. Therefore, information on the management of wetlands will be crucial for the ongoing reporting and evaluation of the delivery against the objectives of the SDGs. Further, information on the implementation of commitments under international agreements will also be needed, noting that Finlayson (2012) reported that with a few exceptions implementation of commitments under the Ramsar Convention had been inadequate.

Wise Use of Wetlands and Sustainable Development

The Ramsar Convention on Wetlands has adopted as its mission: "the conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world" (http://www.ramsar.org/about/the-ramsar-convention-and-its-mission). Therefore, sustainable development is at the heart of the global wise use of wetlands. This commitment has been further endorsed through declarations and resolutions.

"Wise use" can be considered as the longest established intergovernmental example of an approach to conservation and sustainable development (Finlayson et al. 2011). The wise use concept was first defined by the Third Conference of the Parties to the Ramsar Convention (COP3, held in Regina, Canada, 1987) as "their sustainable utilization for the benefit of humankind in a way compatible with the maintenance of the natural properties of the ecosystem." It further defined "sustainable utilization" as "human use of a wetland so that it may yield the greatest continuous benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations" (Ramsar Convention 1987). This was very similar to the definition adopted by the Brundtland Report published in the same year.

At the Ninth Conference of the Parties to the Ramsar Convention (COP9, held in Kampala, Uganda, 2005), further efforts were made to harmonize various terms in use by the Convention, including "wise use" and "ecological character" to take into account more widely used terms and definitions relating to ecosystems and sustainable development. The result was the following updated definition of wise use:

> Wise use of wetlands is the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development.

This definition considered ecosystem approaches – such as the one adopted by the Convention on Biological Diversity which states that the ecosystem approach is a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way – as a basis for conservation and sustainable development of natural resources. The definition also placed such an approach "within the context of sustainable development" with the intention of recognizing that development could be facilitated in sustainable ways but that it should not imply that development is, or will be, the objective for every wetland.

To assist countries to manage their wetlands, the Ramsar Convention has produced a substantial amount of guidance for the wise use of all wetlands. Much of this has been presented in the Ramsar Handbooks for the Wise Use of Wetlands (Ramsar Convention Secretariat 2010) as well as Ramsar Technical Reports and Briefing Notes (Table 1). Further information is available in resolutions adopted by the 11th and 12th Conferences of Ramsar Contracting Parties (COP11 2012; COP12 2015) (see www.ramsar.org).

The linkages between the wise use and the sustainable development of wetlands were strengthened at the Global Forum on Wetlands for the Future, which was held in Tehran, Islamic Republic of Iran, in March 2011 to commemorate the 40th anniversary of the signing of the Ramsar Convention on Wetlands. The forum was attended by ministers, senior officials, and high-level representatives from contracting parties, international and regional organizations, academic institutions, and other partners to the Convention. A significant outcome of the forum was the "Tehran Declaration on Wetlands and Sustainable Development" (http://www.ramsar.org/sites/default/files/documents/pdf/ramsar_40/Tehran_Declaration%202011.pdf) which reiterated the linkages between the wise use of wetlands and sustainable development and called for the contents of the declaration to be conveyed to the World Summit on Sustainable Development.

Table 1 Guidance from the Ramsar Convention on Wetlands for ensuring the wise use of wetlands (Accessible through www.ramsar.org)

Overarching frameworks	Reference sources
Wise use of wetlands	
Concepts and approaches for the wise use of wetlands	Wise Use Handbook 1
An integrated framework for the convention's water-related guidance	Wise Use Handbook 8
An integrated framework for wetland inventory, assessment, and monitoring	Wise Use Handbook 13
A framework for Ramsar data and information needs	Wise Use Handbook 14
Wetland inventory and assessment	
A Ramsar framework for wetland inventory and ecological character description	Wise Use Handbook 15
Guidelines on biodiversity-inclusive environmental impact assessment and strategic environmental assessment	Wise Use Handbook 16
Guidelines for the rapid assessment of inland, coastal, and marine wetland biodiversity	Ramsar Technical Report No. 1 Anon (2006)
Low-cost GIS software and data for wetland inventory, assessment, and monitoring	Ramsar Technical Report No. 2 Lowry (2006)
Guidance for valuing the benefits derived from wetlands ecosystem services	Ramsar Technical Report No. 3 De Groot et al. (2006)
A framework for a wetland inventory metadatabase	Ramsar Technical Report No. 4 Lowry (2010)
A framework for assessing the vulnerability of wetlands to climate change	Ramsar Technical Report No. 5 Gitay et al. (2011)
State of the world's wetlands and their services to people: a compilation of recent analyses	Ramsar Scientific and Technical Briefing Note No. 7; Gardner et al. (2015)
Wetland management	
Frameworks for managing Ramsar sites and other wetlands	Wise Use Handbook 18
Addressing change in the ecological character of Ramsar sites and other wetlands	Wise Use Handbook 19
Establishing and strengthening local communities' and indigenous people's participation in the management of wetlands	Wise Use Handbook 7
Integrating wetland conservation and wise use into river basin management	Wise Use Handbook 9
Guidelines for the allocation and management of water for maintaining the ecological functions of wetlands	Wise Use Handbook 10
Managing groundwater to maintain wetland ecological character	Wise Use Handbook 11
Wetland issues in integrated coastal zone management	Wise Use Handbook 12
Determination and implementation of environmental water requirements for estuaries	Ramsar Technical Report No. 9 Adams (2012)

(*continued*)

Table 1 (continued)

Overarching frameworks	Reference sources
The benefits of wetland restoration	Ramsar Scientific and Technical Briefing Note No. 4; Alexander and McInnes (2012)
Toward the wise use of urban and peri-urban wetlands	Ramsar Scientific and Technical Briefing Note No. 6; McInnes (2013)

Future Challenges

After 40 years of countries implementing the Convention, Finlayson et al. (2011) posed and then answered the question "has Ramsar's wise use approach helped achieve sustainable development for wetlands?" To relate this back to the Brundtland definition, the question could be rephrased as has development taken place in a manner as not to compromise the ability of wetlands to allow future generations to meet their own needs. Drawing on conclusions presented in the Millennium Ecosystem Assessment (2005), which reported that the degradation and loss of wetlands is more rapid than for other ecosystems and the status of both freshwater and coastal wetland species is deteriorating faster than those of other systems, the answer was that overall it seemed that this approach had not helped achieve sustainable development for wetlands. This position appears to have been substantiated by similar work on both species decline (Butchart et al. 2010) and estimates on the rate and extent of wetland loss (Davidson 2014). Finlayson et al. (2011) did add that without the efforts under the Convention that the situation may have been worse, as shown through the successful development of the list of Ramsar wetlands of international importance.

As the global store of wetlands decreases, the needs of future generations continue to be eroded. Therefore, the pressure on those responsible for implementing the SDGs to genuinely protect, restore, and promote sustainable use of ecosystems is greater than ever. Wetlands have been explicitly highlighted within the SDGs, giving them a higher profile than in the MDGs, which hopefully will keep the protection and restoration of wetlands on the global agenda and ultimately deliver sustainable development and the wise use of wetlands.

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77 Sustainable Development Goals

Robert J. McInnes

Contents

Abstract

The concept of sustainable development has been embedded in development and environmental thinking for several decades. In 2000 the world's leaders established the Millennium Development Goals (MDGs) as a collective agenda to improve the lives of people across the globe. In 2015 the Sustainable Development Goals (SDGs) were proposed, as the successors to the MDGs, representing an agenda for people, planet, and prosperity that seek to strengthen universal peace and shift the world on to a sustainable path.

Keywords

Sustainable development · Intergovernmental cooperation

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Introduction

The concept of sustainable development has a long history in both the development and environmental literature. However, there have been several key landmarks over the last 30 years or so. One of the first milestones occurred in 1987 when the World Conference on Environment and Development published their report entitled, "Our Common Future" (WCED 1987), commonly termed the "Brundtland Report," after the chairperson, the then Norwegian Prime Minister, Gro Harlem Brundtland. The term "sustainable development" was widely used throughout the report and was defined as:

> Development that meets the needs of the present without compromising the ability of future generations to meet their own needs.

This definition has shaped considerably the thinking on sustainable development and has been very influential in directing global efforts to secure a sustainable future for all. The concept of sustainable development recognizes that there are limitations imposed on the capacity of the biosphere to tolerate the negative impacts of human activities by the current knowledge, technology, and societal organization capabilities. The current global challenge for sustainable development lies in understanding the complex and evolving interdependencies of environment, social and economic development within an increasingly globalized world, as shown through the analyses undertaken in the Millennium Ecosystem Assessment (2005).

The Millennium Development Goals: Precursors to the Sustainable Development Goals

The United Nations Conference on Environment and Development, known colloquially as the "Earth Summit," took place in Rio de Janeiro, Brazil, in 1992 and marked a significant moment in the history of sustainable development. One of the central aims was to identify the principles of an agenda for action towards sustainable development in the future which for the first time considered the environment and specifically biodiversity. Some 8 years late in September 2000, to mark the new millennium, heads of state met at the United Nations headquarters to review progress on the Earth Summit and consider a way forward in order to deliver a more sustainable future. The meeting adopted the Millennium Declaration which established the Millennium Development Goals (MDGs) which aimed to improve the lives of people around the world and included specific targets to be met by 2015. The MDGs cut across a variety of issues and were to germane to wetland management at many levels including through food production, hunger and poverty eradication, climate change adaptation and mitigation, water extraction, sanitation, and water-borne and vectored disease.

In the report on progress towards the MDGs published 1 year before achievement is due (United Nations 2014) there were some significant failures with regard to delivery. It is clear that some MDGs will meet their objectives by 2015, others will be within reach, while some will remain as urgent priorities for global leaders, in need of attention beyond 2015. It has been argued that progress of the MDGs has been limited by system-level constraints, including poor governance, weak institutions, and a lack of appropriately trained personnel (Horwitz et al. 2014).

In June 2012, 20 years after the “Earth Summit”, world leaders, along with representatives from the private sectors and nongovernmental organizations came together to discuss “the future we want” at the Rio + 20 United Nations Conference on Sustainable Development. The discussions focussed on two main themes: how to build a green economy in order to deliver on sustainable development while lifting people out of poverty and how to improve global co-ordination for sustainable development. Through the outcome document “The Future We Want” the conference set out the mandate to develop the successors to the MDGs, which expire at the end of 2015, termed the Sustainable Development Goals (SDGs).

The aspirations of the above-mentioned initiatives have been paraphrased by McCartney et al. (2014) in terms of seeking a balance between development and the sustainable use of ecosystem services as follows: “The key to sustainable development is achieving a balance between the exploitation of natural resources for economic development and conserving ecosystem services that are critical to everyone’s wellbeing and livelihoods. There is no blueprint for obtaining this balance but it is essential to understand how ecosystem services contribute to livelihoods and who benefits and who loses from changes arising from development interventions.” The further reiterated that the findings from the Millennium Ecosystem Assessment (2005) that emphasized the importance of adopting ecosystem-based approaches to wetland management that enabled communities to consider the trade-offs between different wetland ecosystem services as mechanisms for supporting the MDGs were as relevant for designing the SDGs.

The SDGs and Wetland Management

The United Nations published the zero order draft of the proposed SDGs in June 2015 (https://sustainabledevelopment.un.org/content/documents/7261Post-2015%20Summit%20-%202%20June%202015.pdf). The SDGs are an agenda for people, planet, and prosperity that seek to strengthen universal peace and shift the world on to a sustainable path. The agenda defines a sustainable future where the planet is protected and natural resources, including wetlands, are used sustainably. The “zero draft” document acknowledges the progress made through the MDGs but accepts that progress has been uneven and some goals were not achieved. Therefore, at the heart of the SDGs is the desire to complete what the MDGs failed to finish.

World leaders have pledged common action and endeavour across a broad policy agenda which has been captured in 17 Goals (below) and 169 associated targets in an ambition to achieve a path towards sustainable development (https://sustainabledevelopment.un.org/topics/sustainabledevelopmentgoals). The SDGs will come into effect on 1 January 2016 and will guide decision-making over the ensuing 15 years.

The SDGs, along with the associated targets, are intended to be integrated and indivisible, global in nature, and universally applicable. They take into account the realities of different national circumstances, capacities, and levels of development and also pay respect to sovereign policies and priorities. Targets are defined as aspirational and global, and it is the responsibility of each government to establish its own national targets based on the overarching global level of ambition.

- **Goal 1.** End poverty in all its forms everywhere
- **Goal 2.** End hunger, achieve food security and improved nutrition, and promote sustainable agriculture
- **Goal 3.** Ensure healthy lives and promote wellbeing for all at all ages
- **Goal 4.** Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all
- **Goal 5.** Achieve gender equality and empower all women and girls
- **Goal 6.** Ensure availability and sustainable management of water and sanitation for all
- **Goal 7.** Ensure access to affordable, reliable, sustainable, and modern energy for all
- **Goal 8.** Promote sustained, inclusive, and sustainable economic growth, full and productive employment, and decent work for all
- **Goal 9.** Build resilient infrastructure, promote inclusive and sustainable industrialization, and foster innovation
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- **Goal 16.** Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels
- **Goal 17.** Strengthen the means of implementation and revitalize the global partnership for sustainable development

Wetland managers and policy-makers can contribute to the delivery of SDGs through the close links to a range of wetland issues including *inter alia* human health and disease, the provision of food, water and sanitation, the ability of wetlands to mitigate pollution, the wise use of urban wetlands as important natural infrastructure and the increased resilience of human communities to hazards. Unlike the MDGs, wetlands have a high and specific profile within the SDGs. Goal 6 seeks to ensure availability and sustainable management of water and sanitation for all and has the specific target of protecting and restoring water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers, and lakes by 2020.

Similarly, within Goal 15, which seeks to protect, restore, and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss, wetlands receive a specific mention in the targets. Target 15.1 states that by 2020, ensure the conservation, restoration, and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains, and drylands, in line with obligations under international agreements. Therefore, the need to conserve, restore, and sustainably use wetlands in line with obligations under international agreements, such as the Ramsar Convention on Wetlands, is clear.

Future Challenges

Information on the management of wetlands will be crucial for the ongoing reporting and evaluation of delivery against the objectives of the SDGs. There will be an onus on wetland managers, from the site level, up to the intergovernmental level through the Ramsar Convention to play their part. In the recently adopted Ramsar Strategic Plan 2016–2024 (Ramsar Convention 2015), it is recognized that all wetlands including the Ramsar Site network will have a direct relevance to the achievement of the SDGs and especially those related to poverty eradication, food and nutrition, healthy living, gender equality, water quality and supply, water security, energy supply, reduction of natural disasters, innovation and the development of appropriate infrastructure, sustainable human settlements adaptation to climate change, oceans, seas and marine resources, biodiversity, and sustainable use of ecosystems.

The challenge in meeting the multiplicity of goals outlined through the SDGs has previously been expressed in many publications, as summarized in global assessments (MEA 2005; Molden et al. 2007) and expressed by Falkenmark et al. (2007) when considering how to avoid the costs of going too far when addressing the balances and trade-offs between agriculture, water, and ecosystems. This could be done by reconsidering the web of relationships that occur between people and wetlands through the medium of ecosystem services and by adopting the “settings” approach (Horwitz and Finlayson 2011), wherein wetlands are a setting for achieving the SDGs with commensurate benefits for people. In this manner, the many benefits that accrue for people through maintaining healthy wetlands can be assessed and trade-offs identified and agreed within the relevant socio-political landscape.

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Millennium Development Goals

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Pierre Horwitz

Contents

Abstract

This short paper outlines how intervening in the disruption to wetland ecosystem services will improve human health and help address the Millenium Development Goals. It also discusses systemic consequences, where addressing the MDGs (principally by the interventions imposed by other sectors) will need to be aware of the relationship between human health and wetland ecosystem health.

Keywords

Sectoralism · Ecosystem Services · Poverty · Child Mortality · Primary Education · Gender Equality · Major Diseases · Global Partnerships

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*This article has been adapted from Annex 1 of Resolution XI. 12 of the Ramsar Convention's Conference of Parties held in Bucharest Romania in 2012.

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Introduction

Through the adoption by the United Nations in 2000 of the *Millennium Declaration,* the world's governments established the Millennium Development Goals (MDGs) to improve the lives of people around the world, particularly those most vulnerable and disadvantaged, with specific targets to be reached by 2015.

The MDGs are designed to lift people out of poverty, save lives, ensure adequate childhood education, reduce maternal deaths, and expand opportunities for women and girls through empowerment. Of direct relevance to wetlands and water resource management, they seek to ensure access to clean water and alleviate the burden of deadly and debilitating diseases that many people face. They seek to promote sustainable development and protect the most vulnerable from the devastating effects of multiple crises, whether they be conflicts, natural disasters, or volatility in prices for food and energy (United Nations 2012).

Beyond Sectoralism

Global attempts to achieve these goals and targets have not abated in the second half of the 15-year period: "At the 2010 High-level Plenary Meeting of the General Assembly on the Millennium Development Goals, world leaders reaffirmed their commitment to the MDGs and called for intensified collective action and the expansion of successful approaches" (United Nations 2011, p. 5). Along with this reaffirmation came a concerted effort to look beyond the sectoral boundaries that defined each of the goals. In particular, the "environmental" or "natural resource" sectors recognized that they had a contribution to make to each of the MDGs beyond just Goal 7 that sought to ensure environmental sustainability.

Early efforts to go beyond Goal 7 included UN WWDR (2006), Molden (2007), UNEP (2007), global reports that sought to place water management, agriculture, and global state of the environment reporting in the context of vulnerable peoples, and governance for sustainability. Attentiveness to the links and interplays between biodiversity and poverty and between human health and ecosystem services (Secretariat of the CBD 2010; Horwitz et al. 2012; Pretty et al. 2011) has demonstrated that the complex and "wicked" problems of both ecosystem change, and poverty must be addressed through intersectoral action. This is because many of the social, economic, and demographic driving forces, and possible response options, lie primarily outside the direct control of any one sector, like the environmental sector or the health sector. Instead they are embedded as attitudes, perceptions, and realities in sectors such as sanitation and water supply, education, energy, agriculture, trade, tourism, transport, development, and housing (Horwitz et al. 2015).

Table 1 Ways in which wise use and wetland management can contribute toward the achievement of the Millennium Development Goals (Modified from Horwitz et al. (Ramsar Technical Report No. 6, 2012), which was compiled using the material presented in Molden (2007), UNEP (2007), UN WWDR (2006), and as otherwise indicated)

Millennium Development Goals (MDGs)	How will intervening in disruption to wetland ecosystem services improve human health and help address the MDGs?	Systemic consequences: where will addressing MDGs need to be aware of the relationship between human health and wetland health?
1. Eradicate extreme poverty and hunger	Food security of the poor often depends on healthy ecosystems and the diversity of goods and ecological services they provide. Diverse wetland ecosystems are self-sustaining and provide the essential genetic material for aquaculture and horticulture. Sustainable livelihoods by definition seek to ensure that the core requirements of food and water are provided to those dependent on the provisioning of wetland ecosystems	The challenge for irrigated agriculture is to improve equity, reduce environmental damage, increase ecosystem services, and enhance water and land productivity in existing and new irrigated systems. Improving productivity should not be at the expense of other ecosystem services. If it is, the human, animal, and plant health consequences of ecosystem disruption will occur in full or in part due to a range of both direct and indirect impacts, the latter as a result of altered health status of livestock and wildlife
2. Achieve universal primary education	Wetland management needs to address the disruptions to ecosystem services that result in water-related diseases. Water-related diseases such as diarrheal infections cost about 443 million school days each year, diminish learning potential, and reduce the coping capacity of local populations for current predicaments and future ecosystem changes	Primary education should include knowledge of health, water, and energy issues at least (a fundamental necessity for urban dwellers who have become more alienated from their surroundings than at any stage in history). Education services can have tendencies to resist increases in attention to such environmental issues at the expense of other subjects
3. Promote gender equality and empower women	Addressing degradation in wetlands, such as water contamination and deforestation, will contribute to the health of women and girls. Women and girls bear the brunt of collecting water and fuelwood and are more vulnerable members of populations to waterborne diseases	Improved wetland management should involve women and girls in a meaningful way, perhaps by recognizing that women can play greater roles in wetland management than they currently do. "Wetland managers," as professions, tend to be dominated by men. Decision-making structures for water resource management, wetland management, and agriculture are also gendered in many parts of the world. These may operate as barriers to achieve this goal

(*continued*)

Table 1 (continued)

Millennium Development Goals (MDGs)	How will intervening in disruption to wetland ecosystem services improve human health and help address the MDGs?	Systemic consequences: where will addressing MDGs need to be aware of the relationship between human health and wetland health?
4. Reduce child mortality	Wetland management will become an essential operational requirement to reduce exposures to waterborne diseases, such as diarrhea and cholera. Prevalence of these diseases is a result of disruption of regulatory services (as a result of overextraction and inappropriate practices)	Interventions such as water treatment facilities (often through aid provision) will usually be technological and infrastructural in the short term to address immediate needs. However, the medium- to long-term goal should be the management of wetland ecosystems to ensure that they can provide suitable water purification and pathogen removal services
5. Improve maternal health	Addressing disruptions to wetland ecosystem services will always include an examination of water quality. Provision of clean water reduces the incidence of diseases that undermine maternal health and contribute to maternal morbidity and mortality	Improving the quality of source water from catchments, reservoirs, and wetlands in general, and distribution infrastructure, may reduce disinfection loads and the likelihood of maternal exposures to these loads
6. Combat major diseases	Up to 20% of the burden of disease in developing countries may be associated with environmental risk factors. Preventive environmental health measures are as important and at times more cost-effective than health treatment. Managing wetlands to enhance ecosystem services with the aim of reducing the likelihood of human exposures to pollutants and infectious diseases is preventive, attending to upstream environmental determinants of health. New biodiversity-derived medicines hold promises for fighting major diseases	Increasing human population sizes as a consequence of successful disease prevention measures may also increase pressure on local water and wetland resources. Wetland management needs to act in concert with water resource management to deal with these foreseeable consequences, for instance, by increasing awareness and thus changing behavior and by incorporating the concept of ecosystem services in prevention strategies. This management needs to be integrated with regional population policies, domestic livestock and wildlife policies (to reduce risk of emerging zoonoses), education, and awareness
7. Ensure environmental sustainability	Current trends in environmental degradation need to be reversed in order to sustain the health and productivity of the world's ecosystems. Wetlands, and the biodiversity they support, encompass many of the key ecosystems of the world and many	Development strategies that aim to safeguard the full range of benefits provided by wetlands might better achieve the goal while minimizing harm to wetlands. This requires recognizing the trade-offs that exist when managing for some ecosystem services like those concerned with

(*continued*)

Table 1 (continued)

Millennium Development Goals (MDGs)	How will intervening in disruption to wetland ecosystem services improve human health and help address the MDGs?	Systemic consequences: where will addressing MDGs need to be aware of the relationship between human health and wetland health?
	of the most productive ones. Wetland management applies directly to this goal	production while trading off supporting and regulating services
8. Develop a global partnership for development	Poor countries are forced to exploit their natural resources, like wetland ecosystems, to generate revenue and make huge debt repayments. Unfair globalization practices export their harmful side effects to countries that often do not have effective governance regimes	Trade, tourism, and migrations of species are often transcontinental. Meaningful wetland management acknowledges that pests and pathogens capable of decreasing ecosystem services and having consequences for the health of local human, domestic, and wildlife communities can be distributed by inappropriately planned and controlled human activities. This needs appropriate recognition in global partnerships for development

Wetland Management and the MDGs

Wetland policy-makers and managers can make a contribution toward the MDGs wherever the close relationships between wetland management and food production, hunger and poverty, climate change, water extraction and use, and waterborne and aquatic vector-borne diseases are present. Contributions can be foreseen along two axes. The first is intervening in the ongoing disruption to wetland ecosystem services so as to help to improve human, domestic, and wildlife health and thereby address the MDGs; this axis is shown in column two of Table 1.

The second axis is shown in column three of Table 1. Methods for seeking to achieve the MDGs, improve human health, and enhance wetland ecosystem services may not necessarily be mutually beneficial – indeed, systemic effects like cross scale interactions and feedback consequences may prove to undermine the originally intended objectives. The activities to address MDGs by the international community, national actions, and actions by sectors other than wetland management need to be more cognizant of the systemic nature of the relationship between these objectives and wetland ecosystem health. Where potential negative consequences are foreseeable, this is no reason to avoid actions that seek to achieve these MDGs; rather, those consequences need to be understood and considered in decision-making.

An understanding of the trade-offs among different wetland ecosystem services and the need for cooperation across sectors is critical in designing further actions in support of the MDGs. For example, it is not uncommon for strategies intended to increase food production and reduce poverty to propose the conversion of marshes to agriculture, conversion of mangroves to aquaculture, and significant increases in the use of fertilizers to increase crop production. This approach, however, will reduce habitat area (and hence the magnitude of services provided by the original habitat), increase the input of water pollutants, remove the natural water filtering service provided by wetlands, and remove ecosystem services, such as storm surge protection, timber and charcoal supply, and fish habitat provided by mangroves, on which local residents in particular rely. This will make the development goal of improved water and sanitation more difficult to achieve and may in fact increase poverty for some groups. In contrast, a development strategy that safeguards the full range of benefits provided by wetlands might better achieve the set of development goals while minimizing future harm to the wetlands.

The Future: MDGs and Their Next Iteration

Some MDGs will be achieved in 2015, others will be within reach, others will remain as urgent priorities for global leaders; indeed attention will be needed beyond 2015. In the meantime, other urgent priorities have emerged. An evaluation of the achievements of the MDGs should examine the degree to which local communities have been involved in decision-making regarding interventions, "atypical" allegiances have been formed, and trust and reciprocity with local communities, across sectors, and across disciplines have been developed. Arguably, progress on MDGs has been limited by system-level constraints, including poor governance, weak institutions, and a lack of appropriately trained personnel. Wetland managers, indeed the wetland sector, will need to be (more) active and participate in the evaluation of the MDGs and the next iteration in order to make the sorts of contributions outlined in this article.

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Non-Governmental Organizations: International and Regional

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Chris Rostron

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Abstract

NGOs play a vital role in delivering wetland conservation at all levels. Their role is often under-recognized as many NGOs do not have a formal status and because they deliver such a diverse set of activities. However, from supporting top-level environmental conventions, carrying out research, engaging local communities, and delivering practical conservation work, NGOs are a powerful tool for the delivery of wetland conservation. International and regional NGOs concerned with wetland conservation are described.

Keywords

Nongovernmental organizations · International · Regional

Introduction

NGOs play a vital role in delivering wetland conservation at all levels. Their role is often under-recognized as many NGOs do not have a formal status and because they deliver such a diverse set of activities. However, from supporting top-level environmental conventions, carrying out research, engaging local communities, and

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delivering practical conservation work, NGOs are a powerful tool for the delivery of wetland conservation.

Wetland NGOs

Delivering work on the ground has long been a central activity of wetland conservation that is delivered by local actors. Many of the challenges that wetlands face are linked to the communities that live in and around them. It is clearly of paramount importance that wetlands are managed well, to provide resources that local people need as well for biodiversity. These range from water itself, food such as fish and birds, building materials, and even cultural and recreational activities. It is in the interest of the local community to manage these resources sustainably and that means they need to be equipped with sufficient knowledge, tools, and resources to achieve this. Whether through managing wetlands sustainably, running engagement and education activities, or carrying out monitoring and practical conservation work, local NGOs are often the best-placed organizations to deliver wetland conservation objectives.

In terms of delivering messages to local people to convince them of the importance of wetlands and their protection, in many instances, local NGOs are more trusted than government, business, or international bodies. They have good local knowledge, are often more likely to be active in the long term, and involve local people in their management or as part of their memberships. They are also seen as more committed to local issues and are not associated with other more political or bureaucratic issues. However, it should be noted that they are often reliant on a few committed individuals, and sometimes have fewer democratic credentials, i.e., they are not voted for by the wider community.

Local NGOs are more often than not part of a wider network of groups, either by sectoral interest or formally as part of a constituted regional, national, or international group. These groupings include regional alliances such as the “mingas” of Ecuador, family groups of indigenous peoples in the same regional area, or groups along geographic features such as rivers or even political boundaries such as local government regions. With increasing communication via internet and social networks, interest groups of NGOs or individuals are also more prevalent, leading to networks that function to promote a specific interest, regardless of geographic proximity.

Regional, National and International NGOs

Regional and national NGOs can exist either as membership groups, with local versions, or as independent groups with a broader interest. So, at this level they can play the role of networks with local representatives, or as international organizations with a campaigning role. Often they fulfil both roles, such as BirdLife or WWF.

Research NGOs play a different role, working with government and private sector groups. Often this will form part of a wider remit, delivering other aspects of wetland conservation. For example, the Tour du Valat (France) has been working for many years on wetland conservation, and is a respected, independent voice for wetland science, from climate change to bird census work. Other NGOs such as WWT (Wildfowl & Wetlands Trust) or Wetlands International also carry out an element of research and hold long-standing datasets for species of wildfowl. Wetlands International is contracted by the Ramsar Convention to carry out the international waterbird census, a highly respected and long-term exercise in estimating the numbers of waterfowl globally.

Network NGOs such as BirdLife, WWF, or IUCN also add huge value to those national and international groups that form part of their membership. In this sense, they create a community of wetland NGOs (note, IUCN includes governmental representatives). This both uses local organizations as a resource to get information, deliver work, or commit finance, as well as providing support and structure to those local groups, through an international voice and trusted expertise. Working through local groups offers a means of validating information used at national level and identifies risks and opportunities. These networks offer a voice at international level too, either directly as stand-alone pressure groups or working in partnership as part of multilateral environmental agreements such as Ramsar. As membership groups they also have resources to carry out wetland research and campaigning and coordination activities for their members.

International NGOs such as WWF and BirdLife play a role to influence and support international policy through multilateral agreements such as Ramsar, the Convention on Biological Diversity, and the Convention on Migratory Species. As officially recognized partners with a defined role, these NGOs are part of the structure of the MEAs, with a history of engagement and commitment to the process. They are generally accepted as "observers" but with the capacity to contribute to debates, provide technical support, and use their national/local contacts to support delivery of the aspirations of the MEA in question.

Less formally established networks such as WWN (related to Ramsar) World Wetland Network, and the Migratory Wildlife Network (related to Convention on Migratory Species) are also actively involved with MEAs, offering an independent and potentially more critical voice. They are not as closely allied to the conventions, and are therefore freer to ensure an independent view of them, particularly when it comes to criticizing the member governments of each convention. Both networks are fairly young, and it remains to be seen how their role will develop. However, it is hoped that they will both provide a strong support mechanism to represent the civil society sector, as well as to influence the MEA decisions on behalf of their members.

Future Challenges

With continuing economic uncertainty in western economies, drastic political changes in North Africa and the Middle East, and massive development and liberalization in the Far East, Russia, and other emerging economies, the role of civil society and NGOs which was already varied is now difficult to summarize or predict. There are both major opportunities for development of their influence and activity, but also growing restrictions in some countries.

From the perspective of MLEAs such as Ramsar and the CBD, the role of NGOs is increasingly important. A focus on indigenous communities and equitable access to natural resources (the Nagoya Protocol of the CBD, see ▶ Chap. 57, "Convention on Biological Diversity (CBD) and Wetland Management") and the impacts of local communities on wetlands are all leading to an increased interest in engaging NGOs and civil society. WWN and the CBD Alliance also offer an increased visibility and opportunity for NGOs to engage in the process of these MLEAs.

Herein lies the challenge to raise the profile, engagement, and influence of NGOs as required by MLEAs while at the same time ensuring that both resources and political support are made available at to them national and local level.

Check for updates

Birdlife International

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Robert J. McInnes

Contents

Abstract

Comprising 120 partners from across the world, BirdLife International is the world's largest nature conservation partnership. BirdLife International is driven by the belief that local people, working for nature in their own places but connected nationally and internationally through a global partnership, are the key to sustaining all life on Earth. BirdLife's origins are in the International Council for Bird Preservation (ICBP), founded in 1922.

Keywords

Non-governmental organization · Waterbirds · Nature conservation · Flyways · International cooperation

Introduction

Comprising 120 partners from across the world, BirdLife International is the world's largest nature conservation partnership. BirdLife International is driven by the belief that local people, working for nature in their own places but connected nationally and internationally through a global partnership, are the key to sustaining all life on Earth.

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Widely recognized as the world leader in bird conservation, BirdLife applies rigorous scientific approaches, informed by practical feedback from projects on the ground in important sites and habitats, in order to implement successful conservation programs for birds and all nature.

Each BirdLife partner is an independent nongovernmental organization (NGO). Most of the partners have a local or international profile known outside of the partnership. This allows each partner to maintain its individual national identity within the larger global partnership. BirdLife partners work together in a collaborative, coordinated fashion across national boundaries to build a global Partnership of national conservation organizations.

The BirdLife Partnership has six Regional BirdLife Coordination Offices throughout the world and a Global Office in Cambridge, UK. Together these are known as “The BirdLife International Secretariat.” The Secretariat coordinate and facilitate the BirdLife International strategies, programs, and policies.

A Brief History of BirdLife International

The International Council for Bird Preservation (ICBP), the organization which grew into the BirdLife International Partnership, was founded in 1922. On midday of 20 June 1922, a group of interested people convened at the private home of the UK Minister of Finance, the Chancellor of the Exchequer. This erudite group included Dr T. Gilbert Pearson, the cofounder and President of the National Association of Audubon Societies (now National Audubon for USA), Frank E. Lemon, the Honorary Secretary of the Royal Society for the Protection of Birds (RSPB) (UK), Jean Delacour, the President of Ligue pour la Protection des Oiseaux (LPO) (BirdLife in France), and P. G. Van Tienhoven and Dr A. Burdet from the Netherlands. The group was united by their passion for birds and they concluded that the only effective answer to the growing trade of wild bird feathers or the increasing threats to migratory birds had to be through coordinated international action. Following on from this first meeting, a declaration of principles was adopted which stated:

> We believe that in organising a world-wide Committee we can be of much aid to each other in our several countries by the interchange of literature bearing on bird study and bird protection; and by united action we should be able to accomplish more than organisations working individually in combating dangers to bird-life.

Among its earliest campaigns, ICBP called for an end to the international trade in wild bird feathers used in the hat-making trade. Outrage at this trade had been behind the foundation of the National Audubon (BirdLife in the US), RSPB (BirdLife in the UK), and Vogelbescherming Nederland (VBN) (BirdLife in the Netherlands) towards the end of the nineteenth century, and remained a concern for ICBP well into the 1950s, when the fashion died out, at least in part because of awareness-raising by ICBP member organizations. Other early concerns, which remain central for BirdLife today, included the protection of birds on migration, the identification and protection of the areas where birds congregate in large numbers, and the most important sites for threatened birds.

Alarm over the growing extinction crisis led to ICBP, other organizations, and governments to create the International Union for the Conservation of Nature (IUCN) in 1948. Within IUCN, ICBP was given responsibility for compiling data on the world's threatened birds. The first Red Data Book for birds was published in 1966. This Red Data Book, and its successors, had a profound effect on the global conservation agenda by setting conservation priorities and galvanizing government, institutional, and donor support for conservation. ICBP was also instrumental in promoting international wildlife laws, most significantly the Convention on the Conservation of Migratory Species (CMS) and the European directives on wild birds and habitats.

ICBP's structure as a "federation of federations" (national sections including conservation organizations, government agencies, universities, museums, and special interest groups) proved too cumbersome for united conservation campaigns. A new vision was needed and this lead to transition from ICBP to the BirdLife Partnership in March 1993.

Global Overview: BirdLife in Numbers

Today, BirdLife International is the world's largest nature conservation partnership with the 120 BirdLife partners having more than 13 million members and supporters (Fig. 1). This comprises 2.77 million members and 10.8 million people who supported BirdLife Partners in 2012 without being members.

BirdLife partner organizations have worked with 7,475 local groups, including action at 2,750 Important Bird and Biodiversity Areas. This work has involved

Fig. 1 The Partners are shown in *dark grey* and BirdLife Country Programmes in *light grey*

2.7 million children. In addition, BirdLife partners manage or own 1,553 reserves or protected areas covering 4.3 million ha of natural areas globally. Across the world, the BirdLife Partnership directly employs 7,400 staff with a combined budget of US$ 539 million.

Vision, Mission, and Global Strategy

The vision of BirdLife International is stated formally as:

> The BirdLife Partnership wishes to see a world where nature and people live in greater harmony, more equitably and sustainably.

The mission of BirdLife International is:

> The BirdLife Partnership strives to conserve birds, their habitats and global biodiversity, working with people towards sustainability in the use of natural resources.

BirdLife International works towards the vision and mission through the following commitments:

- To prevent extinctions in the wild.
- To maintain and where possible improve the conservation status of all bird species.
- To conserve the sites and habitats important for birds and other biodiversity.
- To sustain the vital ecological systems that underpin human livelihoods and enrich the quality of people's lives.
- In the process, BirdLife will empower people and contribute to the alleviation of poverty and strive to ensure sustainability in the use of natural resources.

BirdLife International's global strategy has been developed from the bottom up by the BirdLife Partnership. The strategy directly supports the commitment of the world's governments to take urgent and effective action to halt the loss of

Table 1 BirdLife's strategic objectives

Save species	Conserve sites and habitats	Encourage ecological sustainability	Enable positive change through people
Prevent extinctions Keep common birds common	Identify, conserve, restore, and monitor the sites and habitats important for birds and other biodiversity Promote resilient ecological networks	Demonstrate and advocate nature's values Promote policies that support sustainability	Catalyze support for nature Promote local conservation action Strengthen the global BirdLife Partnership

biodiversity and to achieve the 20 Aichi biodiversity targets by the year 2020. To achieve this, the strategy has the following four pillars which represent BirdLife's approach to conservation:

1. Save Species
2. Conserve Sites and Habitats
3. Encourage Ecological Sustainability
4. Enable Positive Change through People

Each of the four strategic pillars is underpinned by strategic objectives (Table 1).

Conservation International

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Tracy Farrell

Contents

Abstract

Conservation International (CI) operates in more than 30 countries empowering societies to sustainably care for nature, our global biodiversity, for the well-being of humanity – building upon a strong foundation of science, partnership, and field demonstration. CI's Greater Mekong Program over the next 5 years aims to: *(a) conserve 1.35 million ha of terrestrial and freshwater systems and 350 km of rivers that produce key ecosystem services (carbon storage, freshwater, food, health, cultural services, and biological diversity) and (b) mainstream the values of the ecosystem services in the national economy making the distribution of the benefits of these services more equitable among people as well as among generations*. About 2.75 million people indirectly and 9,000 people directly are expected to benefit from this work.

Keywords

Nongovernmental organization · Nature conservation · Mekong River · International cooperation

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Conservation International: Action in the Greater Mekong

Conservation International (CI) operates in more than 30 countries empowering societies to sustainably care for nature, our global biodiversity, for the well-being of humanity – building upon a strong foundation of science, partnership and field demonstration (www.conservation.org). CI's Greater Mekong program over the next 5 year's aims to: *(a) conserve 1.35 million ha of terrestrial and freshwater systems and 350 km of rivers that produce key ecosystem services (carbon storage, freshwater, food, health, and cultural services, and biological diversity); and (b) mainstream the values of the ecosystem services in the national economy making the distribution of the benefits of these services more equitable among people as well as among generations.* 2.75 million people indirectly and 9,000 people are expected to benefit from this work.

The Greater Mekong is the second most biodiverse river system in the world with 1,200 species of fish, 20,000 species of plants, 430 species of mammals, and 800 species of reptiles (CEPF 2012). It is 2,160 km long and flows from China down to Myanmar, Laos PDR, Cambodia and Vietnam, covering an area of about 2.6 million square kilometers and supporting around 326 million people – similar in size to the population of the United States but with a fraction of its wealth (ADB 2012).

The International Union for the Conservation of Nature found that 13% of the freshwater species it assessed were globally threatened, primarily by agriculture and forestry runoff, direct exploitation and/or habitat loss through deforestation, dam construction, and other modifications such as river clearance for navigation (Allen et al. 2012). IUCN also suggested that should all of the hundred plus hydropower dams planned be built on the Mekong, 28% or more of fish species would likely be impacted. And 60% of Mekong people's daily protein comes from fish and fish products (MRC 2005). Fish and other ecosystem services linked to the Mekong such as rice production, transportation, river-based tourism, and sediment deposition were estimated to be worth 33 billion dollars a year; conversely, full hydropower development yields 274 billion dollars in lost services (Costanza et al. 2011).

Where CI: Greater Mekong Works

CI-Greater Mekong works in the 3-S river basin (Sekong, Srepok, and Sesan) which borders Cambodia, Vietnam, and Laos PDR. Although it covers less than 10% of the total Mekong area, the 3-S rivers supply 15% of its sediment and 20% of its water flows (Koehnken 2012; Adamson et al. 2009). CI also works in the Central Cardamoms landscape protecting over 400,000 hectares adopting a "ridge to reef" approach to enhance multiple ecosystem services and protect forested areas. These mountains supply 30% of the Tonle Sap Lake's water supply in the dry season. The Tonle Sap or "Great Lake" work aims to enhance food, water, health, and biodiversity services, and links to development of a more sustainable economy in Cambodia. The lake is also fed by Prey Long forests, where CI works on forest conservation and is developing a Reduced Emissions from forest Deforestation and Degradation

(REDD) project. Within these interconnected focal landscapes and watersheds, CI promotes the protection and valuation of natural capital, sustainable production, markets, and building governance – demonstrating how healthier ecosystems are not only valuable in their own right, but essential for the delivery of healthier economies and for enhancing human well-being.

Landmark Contributions

CI in its 25 years of operation has contributed to the protection of 106 million hectares of land, freshwater, and sea around the world. It was the first to adopt and implement the Hotspots and the High Biodiversity Wilderness Areas methodologies-protecting 53 areas critical for biodiversity. CI then created the Critical Ecosystem Partnership Fund which has funded 137 million dollars of projects and leveraged another $320 million to build capacity for more than 1,600 civil society organizations. CEPF is now funding its second phase of projects in Indo Burma focusing on freshwater and terrestrial ecosystem protection. The 100 million dollar Global Conservation Fund (GCF) invested in 9 million hectares to set aside new and improve the management of existing protected areas – including watersheds and wetlands in Asia, Africa and the Americas.

CI also discovered and documented hundreds of new or endemic species in priority high-biodiversity countries (with 6 of those found in Cambodia); partnered with more than 1,000 local and indigenous organizations to support protection of biodiversity and ecosystems (more than 25 such partners in Cambodia); and impacted policy and corporate decisions that support sustainable development – engaging more than 30 corporate partners. CI-Greater Mekong has granted $4.6 million dollars to partners and trained over 5,000 people in the past 10 years.

Major Activities

Science and Policy

CI science in the Mekong supports policy and more sustainable resource use and decision-making. For example, natural capital metrics are being developed to assess and monitor stocks and flows of ecosystem services; and a Multiple Interface Decision Assessment System will be completed next year to help governments in lower Mekong countries select various hydropower and agriculture development options which have less impact on the Tonle Sap lake. The University of Canterbury in New Zealand and other CI partners are looking at the alternative location, design and operation options as well as payment for ecosystem services options for hydropower. CI-Greater Mekong produced a film “Hydropower Impacts and Alternatives” now being used by NGOS and other partners to promote sustainable 3-S river hydropower development and encourage the use of alternative energy (cambodiahydropower.weebly.com/).

3-S River Basin

The 3-S River Basin provides an important contribution of aquatic biodiversity and ecosystem services linking to the Mekong River, especially with regard to fish habitats (rapids, deep pools, and sand bars) and migration routes. There are currently nine operating dams and 11 projects under construction. Existing dam operations in the Sesan River already cause high and unpredictable fluctuations of water levels downstream and have caused changes in water quality, a major decline of fish populations and other species, and the loss of livelihood and economic security (Piman et al. 2013). The impact of sediment and nutrient flow changes due to both hydropower and land use change on downstream ecosystems such as the Tonle Sap lake Sap lake Sap is also likely to be very significant.

CI-Greater Mekong's work includes supporting and leveraging science depicting the importance of this basin for human livelihoods and ecosystem services – aiming to influence regional hydropower planning discussions. In the future, the program will create trans-boundary protected areas between Cambodia and Laos, and between Cambodian and Vietnam to protect undammed sections of the 3-S rivers. CI-Greater Mekong will also build future scenarios and conduct joint visioning exercises to encourage better cooperation across borders to manage the 3-S rivers for multiple purposes.

Cardamoms/Tonle Sap Scape

The Cardamom Mountain Range covers approximately two million hectares of forested land and is one of the largest remaining areas of natural forest in Southeast Asia. The Central Cardamoms Protected Forest (CCPF) is home to approximately one-third of all Endangered and Rare species listed under the Cambodian Forestry Law, and almost fifty IUCN-listed threatened species (FA 2008). CI-Greater Mekong works with the Forestry Administration and local communities to protect species and their habitats – conducting species research, law enforcement training, and overseeing patrols. Conservation agreements are also implemented to compensate communities for the opportunity costs of conservation (e.g., tilling machines, teacher salaries, etc.).

The CCPF is critical for ecosystem services for thousands of people living within and around its borders, as well as those 30,000 living further downstream in Koh Kong and Pursat provinces (Sousan and Sam 2013). CI-Greater Mekong is also working on sustainable financing options for conservation including a establishing a trust fund, PES projects to fund forest protection in exchange for avoided sedimentation which benefits hydropower developers, and working with economic land concession owners to create sustainable production landscapes.

The Tonle Sap ecosystem depends upon the Mekong, 3-S, and Cardamom Mountains for water, sediment, nutrients, and fish – all of which drive the lake's incredible biodiversity and productivity. According to the Cambodian Fisheries Administration, a single hectare of floodplain in the Tonle Sap lake can produce up to 230 kg of fish a year (FiA 2014). In terms of its value, the overall fishing sector accounts for 10–12% of gross domestic product and contributes more to income, jobs, and food security than in any other country in the world (Baran et al. 2007). The Tonle Sap is very rich in species with over 300 kinds of fish and contains over 1/3 of Cambodia's rarest species (e.g., otters, fish eagles, fishing cats) (CEPF 2012). The lake is vital for national food security, with almost three million people living in or around it directly relying on it to survive and thrive.

CI-Greater Mekong is working with Fisheries Administration and local communities to minimize overfishing and illegal fishing, as well as to protect and replant the flooded forest which is down to 20% of what it once was. Species and flooded forest monitoring take place monthly by trained community members to benefit key species like otters and fishing cats. Dry season pond protection, demarcation, and management of new fish sanctuaries and conservation areas help protect species and ecosystems. These are no take areas which reduce the impacts of fishing pressure and protects critical brood stocks. Research examining the effectiveness of CI's management model is now underway.

CI-Greater Mekong also supports community fisheries – setting them up and training management committees so that they can be in charge of managing their own resources and ultimately receive government financial support to do so. The program also provide business training and helps set up savings groups for women to both increase revenues from fish processing and protect them from income shocks which can place them deeply into debt.

The Greater Mekong program linking science, policy, and field demonstration work is showing how livelihood and biodiversity benefits can result from sound ecosystem management.

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Ducks Unlimited (DU)

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C. Max Finlayson

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Abstract

Ducks Unlimited (DU) is a large private organization that promotes the conservation and restoration of waterfowl and wetlands. It had its beginnings in the aftermath of a report by the More Game Birds in America Foundation in 1935 that recognized the importance of wetlands in Canada for conserving waterfowl in North America. This led to the formation of Ducks Unlimited in Chicago in 1937 by concerned waterfowl hunters with a mission to conserve waterfowl habitat. Ducks Unlimited Canada (DUC) was formed in 1938 and Ducks Unlimited Mexico (DUMAC) in 1970.

Keywords

Waterfowl · Conservation · Hunting · Partnerships · Flyways

Introduction

Ducks Unlimited (DU) is a large private organization that promotes the conservation and restoration of waterfowl and wetlands. It had its beginnings in the aftermath of a report by the More Game Birds in America Foundation in 1935 that recognized the importance of wetlands in Canada for conserving waterfowl in North America

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(Bolen 2000). This was in an era when waterfowl populations in North America were under immense pressure, and hand rearing of game species was considered the main tool for wildlife management. This led to the formation of Ducks Unlimited in Chicago in 1937 by concerned waterfowl hunters under the leadership of Arthur M Bartley and with a mission to conserve waterfowl habitat. Ducks Unlimited Canada (DUC) was formed in 1938 and Ducks Unlimited Mexico (DUMAC) in 1970.

The Mission of Ducks Unlimited is to conserve, restore, and manage wetlands and associated habitats for North America's waterfowl. It promotes responsible waterfowl hunting along with wetland conservation and restoration. These habitats also benefit other wildlife and people. As waterfowl habitat has been degraded and destroyed across much of North America, DU’s vision is to reverse this trend and fill the skies with waterfowl for today, tomorrow, and forever. This is achieved through diverse public and private partnerships to address the range of factors that continue to erode waterfowl habitat across North America. Information on how DU operates and achieves its vision is available at http://www.ducks.org. These activities include working with elected officials and influencing public policy as well as working multilaterally through partnerships with individuals, landowners, agencies, scientific communities, and other entities. It is an example of a grassroots, volunteer-based organization with membership comprising conservationists and outdoor enthusiasts, largely from North America. At the start of 2014, it had 592,388 adult and 48,900 youth members.

The majority of its members are waterfowl hunters. It has an annual income of approximately US$180 million with some 80% or more of this going towards habitat conservation. The income comes from Federal and State habitat reimbursements (37%), conservation easements (13%), sponsors and members (24%), major gifts and donations (21%), and royalties/advertisement (5%). In 2013, it had total net assets of US$ 132 million. It reports to its members through an annual report and national convention and some 4,000 annual events involving members and partners.

Activities

Ducks Unlimited undertakes many activities involving members and partners. These activities are predominantly focused on North American wetlands and waterfowl and also influence and guide similar activities elsewhere in the world. It currently focuses its conservation efforts on the following five regions, all of which are extremely important for waterfowl: Prairie Pothole Region, Western Boreal Forest – Canada, Mississippi Alluvial Valley, Central Valley/Coastal California, and Gulf Coast Prairie. It also has projects in all states and provinces of the USA and Canada and key areas in Mexico and Latin America. It has undertaken more than 20,000 conservation projects in North America. This includes projects across the four flyways – the Atlantic Mississippi, Central, and Pacific flyways. The importance of flyways and the patterns of migration have been investigated for many years through participation in bird banding and more recent technologies for tracking bird movements. These projects have brought together professional scientists from government and nongovernmental organizations, such as DU, and many members of local communities from across the flyways. In addition to collecting valuable scientific information on waterfowl, these programs can support conservation education and awareness initiatives.

Through its many projects and with the support of its members and partners, DU has conserved more than 46,900 km^2 of waterfowl habitat in North America. It has partnerships with a range of corporations, governments, other nongovernmental organizations, landowners, and private citizens to restore and manage areas that have been degraded and to prevent the degradation of existing wetlands. It also works with partners to recommend policies for wetland conservation and promotes the continuation of safe and regulated waterfowl hunting.

Regional conservation initiatives are organized on a geographical basis and serve to link people and resources with critical habitats and recognize the contribution of each region sustaining the future of waterfowl. The following conservation initiatives are underway:

1. **The Alaska Initiative – Born to Fly**: protecting pristine wetland and delta landscapes for breeding and brood-rearing waterfowl.
2. **America's River Initiative**: rebuilding the Mississippi River floodplain by conserving key wintering habitat for waterfowl and reinforcing the region's waterfowling legacy.
3. **Big Rivers Initiative**: restoring the feeding and resting areas along the rivers that make up the Mississippi floodplain and the conserving habitat critical for migratory waterfowl.
4. **Boreal Forest Initiative**: conserving the intact, productive and pristine forests, wetlands, lakes, rivers, and streams.
5. **California Wetlands Initiative**: restoring wintering habitats for waterfowl, other wildlife, and people.
6. **Completing the Cycle Initiative**: ensuring healthy habitat to sustain migratory waterfowl along the Atlantic Flyway waterfowl.
7. **Ducks in the Desert Initiative**: restoring the oases of North America's desert landscape for waterfowl, other wildlife, and people, from the Great Salt Lake to the Lower Colorado River.
8. **Gulf Coast Initiative**: saving critical wintering waterfowl habitat in North America by conserving coastal prairies and marshes along the Gulf Coast.
9. **Great Lakes Initiative**: protecting the waters of the Great Lakes and conserving critical habitat for many species of waterfowl that use this watershed.
10. **Heartland Heritage and Habitat Initiative**: providing innovative water solutions for ducks and people in the Southern Great Plains.
11. **Living Lakes Initiative**: conserving Minnesota and Iowa's wetlands and shallow lakes for waterfowl migrating between the southern wintering and northern breeding grounds each year.

12. **Peaks to Prairies Initiative**: restoring wetlands in the shadow of the Rocky Mountains and conserving the productive breeding populations of ducks.
13. **Preserve Our Prairies Initiative**: conserving North America's highest-priority waterfowl nesting habitat.
14. **Southeast Wetlands Initiative**: supporting the outdoor heritage of the southeastern United States by conserving key habitats for waterfowl.
15. **Southern Prairies & Playas Initiative**: protecting, restoring, and enhancing the grasslands and wetlands of the Southern Great Plains to support migrating waterfowl populations.
16. **Wings and Wetlands Initiative**: assessing impacts of wetland loss and increasing demand for water in Idaho, Oregon, and Washington states.

Progress with such activities and other news are often reported through the web pages of DU, DUC, and DUMAC and also in the DU magazine. The magazines contain many historical and practical articles on waterfowl hunting and are in part supported by revenue from advertisements of waterfowling equipment. Other publications include the Understanding Waterfowl articles that are now standalone and previously presented in the DU Magazine.

While the main focus of DU's work has been in North America, it has assisted with activities elsewhere, encouraging the establishment of similar organizations in New Zealand and Australia, providing technical advice and guidance to the Ramsar Convention on Wetlands, and making its information and knowledge available through individual and institutional networks.

Conservation Methods

DU works closely with wetland and waterfowl biologists and ecologists to assess habitat needs for waterfowl and to monitor how birds respond to environmental pressures. The hallmark of DU's conservation efforts has been habitat restoration and creation. Some of the approaches used to manage wetlands for waterfowl include:

1. **Restoring grasslands** that provide cover and help conceal waterfowl nests and increase the chances of successful hatching.
2. **Replanting forests** that are regularly flooded by overflow from rivers and provide wintering habitat for ducks, and breeding and foraging habitat for other wildlife.
3. **Restoring watersheds** by protecting stream corridors, establishing buffer zones that filter nutrients and sediments, and restoring previously drained wetlands.
4. **Educating landowners** including ranchers to make their land more wildlife-friendly while enhancing the agricultural and recreational values.
5. **Establishing conservation easements** where the landowner agrees to specific development restrictions while maintaining ownership of the land.

The essence of these activities is their reliance on partnerships and science and support of DU's mission and focus on conserving waterfowl both for the present and future.

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International Union for Conservation of Nature (IUCN)

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Claire Warmenbol and Mark Smith

Contents

Abstract

IUCN is a membership Union uniquely composed of both government and civil society organisations. It provides public, private and non-governmental organisations with the knowledge and tools that enable human progress, economic development and nature conservation to take place together.

IUCN is the world's largest and most diverse environmental network, harnessing the knowledge, resources and reach of more than 1,300 Member organisations and some 16,000 experts. It is a leading provider of conservation data, assessments and analysis. Its broad membership enables IUCN to fill the role of incubator and trusted repository of best practices, tools and international standards.

IUCN provides a neutral space in which diverse stakeholders including governments, NGOs, scientists, businesses, local communities, indigenous peoples organisations and others can work together to forge and implement solutions to environmental challenges and achieve sustainable development.

Working with many partners and supporters, IUCN implements a large and diverse portfolio of conservation projects worldwide. Combining the latest science with the traditional knowledge of local communities, these projects work to reverse habitat loss, restore ecosystems such as wetlands, and improve people's well-being.

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Keywords
International organisation · Sustainable development · Biodiversity conservation · Nature-based solutions · Climate change adaptation · Environmental governance · River basin management · Natural infrastructure

Introduction

The International Union for Conservation of Nature (IUCN) was founded in 1948 as the world's first global environmental organization (www.iucn.org). IUCN's work focuses on valuing and conserving biodiversity, ensuring effective and equitable governance of nature's use, and deploying nature-based solutions to global challenges in climate, food security, and development.

IUCN's vision is a just world that values and conserves nature. Its mission is to influence, encourage, and assist societies throughout the world to conserve the integrity and diversity of nature and to ensure that any use of natural resources is equitable and ecologically sustainable.

Conserving biodiversity is central to the work of IUCN. IUCN demonstrates how biodiversity is fundamental to addressing some of the world's greatest challenges such as climate change, sustainable development, and food security. To deliver conservation and sustainability at both the global and local level, IUCN builds on its strengths in the following areas:

Science – 11,000 experts setting global standards in their fields, for example, the definitive international standard for species extinction risk – the IUCN Red List of Threatened Species.

Action – hundreds of conservation projects all over the world from the local level to those involving several countries all aimed at the conservation of biodiversity and sustainable management of natural resources.

Influence – through the collective strength of more than 1,200 government and nongovernmental member organizations, IUCN influences international environmental conventions, policies, and laws.

A Brief History of IUCN

IUCN was born out of a gathering of minds organized by Sir Julian Huxley, Director General of UNESCO, which sponsored a congress held at Fontainebleau, France, on 5 October 1948 to establish a new environmental institution. In total 18 governments, 7 international organizations, and 107 national nature conservation organizations agreed to form this institution and signed a "constitutive act" creating an International Union for the Conservation of Nature.

From this beginning, the overriding strategy and policy of IUCN has been to explore and promote mutually beneficial conservation arrangements that suit those promoting development as well as assisting people and nations to better preserve biodiversity.

At all times, IUCN has heavily emphasized as a key operating principle the strong need to cater to and address the needs of nations, peoples, and communities, so that they can take ownership of future, long-term conservation goals and objectives in their local areas.

IUCN's *World Conservation Strategy* of 1980, based on these principles, announced IUCN's ambitions to more effectively promote dialogue among proponents of conservation and economic development. With the emergence of the concept of sustainable development, IUCN is engaged in nations around the world, making available the services of a large pool of specialists, providing local level advice and conservation services, and expanding its networks of committees and regional advisory bodies.

IUCN and the Ramsar Convention

In its headquarters in Gland, Switzerland, IUCN also hosts the Secretariat of the Ramsar Convention on Wetlands. IUCN has hosted the Ramsar Secretariat since 1987 and has been associated with the Convention throughout its development. Prior to this, IUCN and the International Waterfowl and Wetlands Research Bureau (IWRB), now Wetlands International, jointly served as the Secretariat for the Convention. However today, the Convention is an entirely independent body directly responsible to, and at the service of, its contracting parties (Matthews 1993).

IUCN helped to build support for a global treaty on wetlands by supporting the establishment of a program for the conservation and management of wetlands. This program was called the MAR project which stands for the first three letters of wetlands in different languages – MARecages, MARshes, and MARismas. In part through the development and dissemination of promotional products, this project helped boost the image of wetlands and communicate the many benefits derived from them. The ensuing MAR conference stimulated the collection of scientific wetland inventories, first for Europe and the Mediterranean region, then further globally.

IUCN's Commission on Ecology was designated in 1967 to operate the "Wetlands Convention" and draw up recommendations for parties' wetland conservation. A first draft of the Convention text was discussed at the IUCN Headquarters in Morges, Switzerland, that same year. When the Convention treaty was signed in Iran on 3 February 1971, IUCN accepted the role of bureau duties for the Convention Secretariat, in close collaboration with IWRB.

IUCN and Wetlands

In 1985, IUCN together with WWF started a major wetlands campaign called “Life at the Water’s Edge” (Wetlands Conservation Program 1985–1987: Life at the Water’s Edge, WWF/IUCN). This campaign promoted the need for wetland conservation, produced a vast array of communication materials, and set out a clear wetland conservation program.

Over the past four decades, the multiple roles of wetland ecosystems and their value to humanity have been increasingly understood and documented. That millions of people around the world rely on wetlands for livelihoods is familiar now to many. The Inner Niger Delta, for example, hosts about 20% of the population in Mali and generates on average 90,000 tonnes of fish catch per year. The Lower Mekong Delta supports the world’s most productive inland fisheries, valued at around USD three billion, caught each year. These inland fisheries provide 56 million people with up to 80% of their animal protein intake.

IUCN is working with the Ramsar Convention and Contracting Parties and also with grassroots communities and river basin agencies to implement wetland conservation as part of its water management work. The benefits are clear: not only does it help to conserve biodiversity and livelihoods, but wetland conservation brings benefits to water quality, fisheries, and protection against floods.

For example, IUCN has been working since the mid 1980s in the Inner Niger Delta, an incredibly rich wetland covering over 20,000 km^2 harboring high concentrations of biodiversity (Zwarts et al. 2005). Together with IUCN members such as the NGO Walia and Wetlands International, IUCN developed an approach to flooded forest restoration which gave life again to the delta’s fisheries and soil fertility. This successful example of ecosystem restoration stemmed in part from contributions of traditional landowners, local governments, and technical state services.

IUCN also supported the Government of Botswana to develop the Okavango Delta Management Plan, adopted in 2008. The Management Plan demonstrated how wetland management benefits the health of ecosystems, as well as generates jobs, development, and well-being in the process.

In Jordan, the Azraq Basin is one of the most important recharging ground water basins in the region, supplying water to the capital Amman. The basin also hosts the Azraq oasis, a Ramsar site particularly important for migratory birds, with up to a million birds utilizing the area during the course of a single spring migration. The key challenges facing the Azraq basin are water quality deterioration, soil degradation, and social conflict. The IUCN office in Jordan is working closely with managers of the Azraq Ramsar site, communities, and water resource managers in the region to better protect and begin to restore this vulnerable and precious wetland site.

In Asia, IUCN helped the government of Lao PDR to complete the process of accession to the Ramsar Convention in 2010. In acceding to the Convention, Lao PDR committed to conservation and sustainable use of its wetlands at a crucial point in the nation’s rapid and transformative economic development. Upon accession, Lao PDR designated the country's first two wetlands of international significance, the Xe Champhone Wetlands and the Beung Kiat Ngong Wetlands.

IUCN is increasingly promoting the idea of "nature-based solutions" as part of strategies for tackling the big problems the world faces. What does this mean? A nature-based solution can be peat bogs that function as water filtration plants, mangroves that protect shoreline development, braided floodplains that serve as dikes and levees, or wetlands that operate as treatment of industrial effluent. This "natural infrastructure" does not necessarily replace built infrastructure, but it may be a more affordable and durable option. It complements engineered designs as a counterpart, working hand in hand. By investing in natural infrastructure, we invest in the people closest to its pulse. With the right incentives and the right institutions, we can empower people to become part of the solution.

During the sixth World Water Forum in Marseille, IUCN Director General Julia Marton-Lefèvre concluded the opening ceremony with this solutions focus:

> Every human, now and in the future, should have enough clean water for drinking and sanitation, and enough food and energy at reasonable cost. Nature immeasurably strengthens our ability to meet these basic needs – but not if we squander our natural capital through blindness to the unseen wealth at our fingertips. We should ask –from the onset- how nature can be a solution, how natural infrastructure can help us design, build, manage and use existing water systems for the sake of our human development, our tight budgets, and our thirsty populations. Nature never asks for our pity. Yet as we have discovered, nature is always more than ready to work as our equal partner.

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International Crane Foundation

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Richard Beilfuss

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Abstract

The International Crane Foundation (ICF) is a nonprofit, nongovernment organization that works worldwide to conserve cranes and the ecosystems, watersheds, and flyways on which they depend. ICF provides knowledge, leadership, and inspiration to engage people in resolving threats to cranes and their diverse landscapes. ICF champions cranes as ambassadors for conservation – serving both as sentinels and flagships for safeguarding some of the most important places on Earth.

Keywords

Nongovernmental organization · Cranes · Wetlands · Watersheds · Flyways · Endangered species · Conservation leadership · Captive breeding · Livelihoods

Introduction

The International Crane Foundation (ICF) is a nonprofit, nongovernment organization that works worldwide to conserve cranes and the ecosystems, watersheds, and flyways on which they depend. ICF provides knowledge, leadership, and inspiration to engage people in resolving threats to cranes and their diverse landscapes. ICF champions cranes as ambassadors for conservation – serving both as sentinels and flagships for safeguarding some of the most important places on Earth.

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ICF was founded in 1973 by Dr. George Archibald and Dr. Ron Sauey. ICF's headquarters are located in Baraboo, Wisconsin, USA, with regional offices in China (Harbin), India (New Delhi), Vietnam (Ho Chi Minh City), Cambodia (Phnom Penh), South Africa (Johannesburg), Zambia (Lusaka), and the USA (Corpus Christi, Texas). ICF employs 54 full-time staff members and dozens more project associates, research associates, and interns and sustains a global network of hundreds of specialists working in over 50 countries on five continents. ICF is governed by an international board of 30 directors and has an advisory board drawn from 12 countries.

ICF's headquarters are located on a 240-acre campus in rural Wisconsin. ICF is visited more than 20,000 people each year, and about 30 international colleagues study at ICF annually. The site hosts a captive flock of over 100 cranes, including the only complete collection of all 15 species ever assembled. ICF's campus is a global classroom, featuring live crane exhibits, crane-breeding and crane-rearing facilities, an interactive education center, research library, visitor center, guest house, and nature trails set among 100 acres of restored prairie, savanna, wetland, and forest, and adheres to the highest standards for green building principles.

Crane Conservation

From its beginning, ICF recognized that conserving all 15 species of cranes requires a broad commitment to the people and places essential to cranes. Cranes are among the most endangered families of birds in the world, with 11 of the 15 species threatened with extinction. Many populations are in peril. In sub-Saharan Africa, gray-crowned, black-crowned, wattled, and blue cranes face many threats fueled by rapid population growth and widespread poverty. In Asia, six species are threatened, including Siberian, red-crowned, white-naped, hooded, and black-necked cranes in rapidly developing East Asia, and the sarus crane throughout its range in South and Southeast Asia. The rarest of all cranes, the whooping crane, faces an array of conservation challenges in North America. Consequently, ICF's global priorities span sub-Saharan Africa, East Asia, Indo-Pacific, and North America, while ICF's headquarters in Wisconsin serve as a global center for conservation leadership and training.

ICF's priority programs focus on six conditions essential to securing all 15 species of cranes in the wild:

1. *Mitigating direct threats to crane populations in the wild*. ICF's field programs target the most serious threats to cranes, including illegal trade, shootings, powerline collisions, poisonings, and others. The capture of cranes from the wild for domestication and trade, for example, is one of the most daunting challenges to crane populations, especially in Africa. In collaboration with Wetlands International, ICF studied the black market for cranes in Mali and learned that thousands of cranes were being illegally captured and exported to Europe, the Middle East, and Asia – and 80% or more die through this supply chain. Based on this knowledge, ICF's programs now focus both on reducing the

international demand for cranes by raising awareness among governments, zoo associations, and private collectors and coordinating captive breeding efforts. ICF also supports the most threatened populations with captive propagation and reintroduction efforts, with particular emphasis on establishing a second, self-sustaining population of whooping cranes in the eastern USA.

2. *Securing ecosystems, watersheds, and flyways on which cranes depend, through sustainable resource management*. ICF works to protect and restore vital breeding, staging, and wintering grounds along the flyways traversed by migrating cranes, which include long-term programs for some of the most important wetlands in the world – e.g., the Poyang Lake in China, the demilitarized zone between North and South Korea, the Kafue Flats and Zambezi Delta in southern Africa, the Mekong Delta in Southeast Asia, and the Guadalupe-San Antonio Bay and Estuary in Texas. ICF advances alternative water and land-use practices in these wetlands and the broader watersheds that sustain them, including the environmental flows, invasive species control, fire management, and other land stewardship practices. In southern Africa, ICF is leading international efforts to secure water for key wetlands in the Zambezi River watershed, through controlled releases of water from dams developed for hydropower production.
3. *Bringing people together for conservation action based on their shared values for cranes and the landscapes they depend on*. ICF hosts the Wetlands International – IUCN Species Survival Commission Crane Specialist Group which stimulates and coordinates activities involving over 250 experts from 50 countries. In the 1990s, ICF helped broker agreements between China and Russia to work together to protect and manage crane breeding grounds along the Amur and Ussuri Rivers that divide the two countries. Through the Convention on Migratory Species, ICF brought together 11 diverse nations – including Afghanistan, Azerbaijan, People's Republic of China, India, Islamic Republic of Iran, Kazakhstan, Mongolia, Pakistan, Russian Federation, Turkmenistan, and Uzbekistan – to sign a Memorandum of Understanding for the conservation of Siberian cranes.
4. *Improving local livelihoods and other benefits for communities through the conservation of cranes and their landscapes*. From the rice paddies of north India to the hillsides of Rwanda to the converted agricultural plains of northeast China, cranes live in some of the most densely settled – and impoverished – regions of the world. Conservation solutions in these settings, as well as among more affluent regions, must bring meaningful benefits to local people, and those benefits must be sustained for the long term. At Phu My in Vietnam, for example, locally produced handicrafts from *Lepironia* wetlands are sold in competitive markets of Europe, Japan, and the Middle East. The income generated covers most of the costs for managing this important wetland for sarus cranes and other species. In just 6 years, local income increased fivefold for the community, and crane numbers jumped from 5 to over 250 (30% of the known population).
5. *Empowering conservation leadership for cranes, local communities, and the landscapes they share*. For 40 years and throughout much of the world, ICF has identified and advanced emerging leaders to help cranes and their landscapes. ICF's headquarters, staffed with experts in crane biology, environmental

education, field ecology, captive management, and conservation medicine, are a preeminent center for the training of future leaders. In Southeast Asia, ICF established a network of 18 universities to empower future leaders in wetland ecology and management in the Mekong Region – training more than 250 young lecturers and wetland managers from universities, government agencies, and conservation organizations from eight countries in the region.

6. *Building knowledge for policy and action to secure cranes and crane landscapes*. ICF is a science-based organization whose success is built on its long-term accumulation of knowledge and expertise. At Poyang Lake, ICF research helps national decision-makers understand the importance of natural water-level fluctuations for critically endangered Siberian cranes, many other wetland-dependent species, and for human needs such as transportation, water quality, and fisheries. By linking the ebb and flow of water with the productivity of aquatic plants that cranes feed on ICF research helped quantify the catastrophic impacts that would result from raising and stabilizing waters at Poyang. In India, ICF research uncovered the significant biodiversity value of the densely settled agricultural regions of the northern Ganges plain and identified factors that favor or threaten the functioning of this patchwork of farm fields and small wetlands.

International Peat Society

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Jack Rieley

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Abstract

The International Peat Society (IPS) is a nongovernmental, nonprofit, multidisciplinary organization of scientific, industrial, and regulatory members. As of 2013, it has 1,450 individual and institutional members from 36 countries. It was constituted in 1968 in Quebec, Canada. The IPS is registered in Finland where the IPS Secretariat is located. The society will celebrate its half century in 2018.

Keywords

Nongovernmental organization · Peatlands · Mires · Conservation · Sustainable use

Introduction

The International Peat Society (IPS) is a non-governmental, non-profit, multidisciplinary organization of scientific, industrial and regulatory members. As of 2013 it has 1,450 individual and institutional members from 36 countries. It was constituted in 1968 in Quebec, Canada. The IPS is registered in Finland where the IPS Secretariat is located. The Society will celebrate its half century in 2018.

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The International Peat Society was formally established at the Third International Peat Congress (Quebec, 1968) building on the success of and interest expressed at two previous peat congresses (Dublin 1954 and Leningrad 1963). Much of the stimulus for its establishment, especially in early post World War 2 years, came from the peat industry that was important for energy generation in several countries including Ireland, Finland, Sweden and the Soviet Union. From the outset, in addition to considerations of the production and industrial utilization of peat, the Society also focused on peat resource survey and classification, peatland agriculture, forestry and horticulture and the chemical, physical and biological properties of peat and peat products.

From the outset, the principal aim of IPS has been to promote international contact and co-operation on all matters concerning the study and wise use of mires and peatlands, peat and peat products and related materials in order to advance knowledge and understanding in the cultural, socio-economic, scientific and technical fields. To achieve its goals the IPS, through its nine Commissions and 17 National Committees, organizes conferences, symposia and workshops and publishes research results from science and industry. IPS serves as a forum to get and to keep peat and peatlands experts, from different fields of business, science, culture and regulatory bodies, in touch with each other.

IPS is managed by an **Executive Board**, consisting of the President of the Society, two Vice Presidents and 2-6 ordinary members who are elected for terms of four years on nominations by the IPS National Committees. To reflect the balance of industry and science, Executive Board members come from both areas with emphasis placed on cultural and geographical balance. With the exception of the Presidents, only one Board member from each country can be elected to the Board. Decisions of the Executive Board are implemented by the **Secretariat** that currently consists of the Secretary General (part time) and Communications Manager (full time) who also carry out the routine management of the Society.

The highest decision-making organ of the International Peat Society is the **Annual Assembly**. It comprises one representative from each National Committee. In addition, other members of the IPS can attend as non-voting observers. The Annual Assembly approves the accounts, the annual report, plan of activities and the budget of IPS. It also elects the President, two Vice-Presidents and members of the Executive Board.

At the national level IPS operates through its **17 National Committees** or the Secretariat in countries without one. In countries with a National Committee, all interested organizations, corporations, foundations and individuals are affiliated to the Society through the National Committee concerned.

The main activities of the International Peat Society are conducted through its nine **Commissions** that act as subject groups within specific fields.

- Commission I - Stratigraphy, inventory and conservation of peatlands
- Commission II - Utilization of peat and peatlands for horticulture, energy and other economic purposes
- Commission III - Agricultural use of peatlands and peat
- Commission IV - Chemical, physical and biological characteristics of peat
- Commission V - Restoration, rehabilitation and after-use of peatlands
- Commission VI - Peat balneology, medicine and therapeutics
- Commission VII - Ecology and management of forested peatlands
- Commission VIII - Cultural aspects of peat and peatlands
- Commission IX - Tropical peatlands

Membership of IPS Commissions is open to all members who are interested in the subject. When required, working groups are established to plan and conduct specific projects. For example, in 2005, a joint cross-Commission Working Group on Peatlands and Climate Change was established comprising representatives of all nine Commissions and a number of external experts. This resulted in the publication of a benchmark book in 2008 – “Peatlands and Climate Change” edited by Dr Maria Strack.

The Chairs of the Commissions together with the 2nd Vice President form the Scientific Advisory Board (SAB)of IPS, which elects its own Chair and Secretary annually. SAB normally meets twice a year in association with other IPS meetings or by Internet. The SAB provides the Society with advice and information on matters of scientific, technical and cultural importance, drawing upon the pool of knowledge available in the Commissions and elsewhere and promoting and disseminating research results.

The **International Peat Society** regularly holds Symposia, Workshops and other meetings in different parts of the world. An **International Peat Congress** is organized every 4th year. Moreover, the IPS encourages its members to attend conferences and meetings of other organizations dealing with peat and peatland research.

IPS Member Services

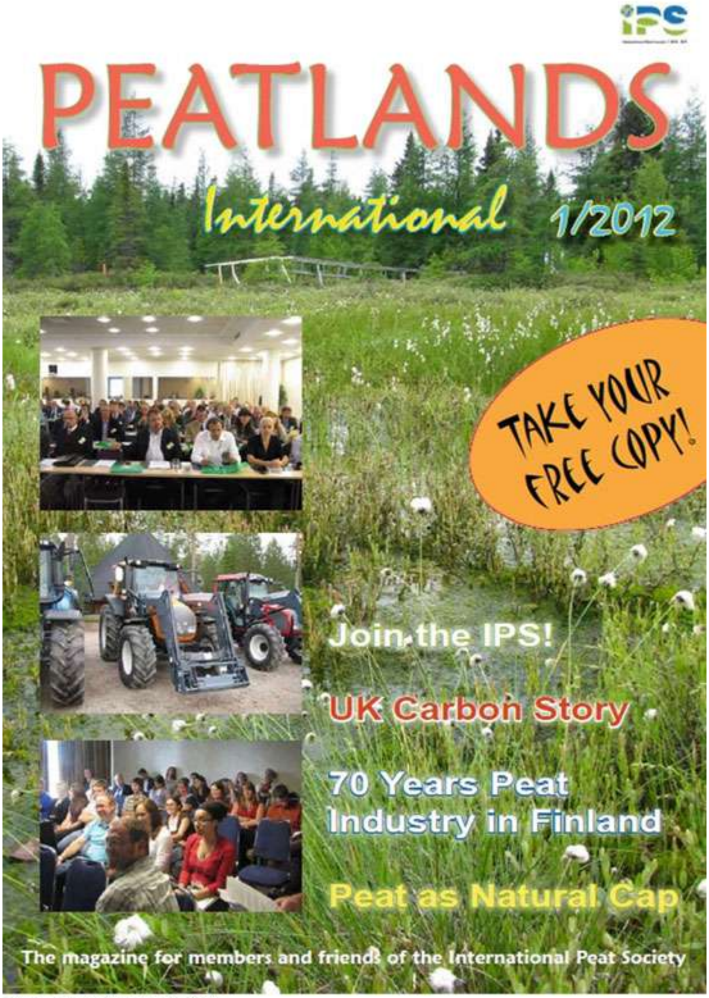
IPS membership confers various privileges, publications and services. The IPS website offers a range of services available to members only who receive a personal user name and password in order to gain access (www.peatsociety.org). Members are able to view and download the proceedings of International Peat Congresses, peat and peatland symposia, workshops and similar events as PDF files for their own use. In addition, it is possible to see and contact other IPS members via the member database, if they have made their data available.

International Peat Congresses

In accordance with its constitution, IPS has held twelve international peat congresses since its inauguration or one every four years, making fourteen in all when Dublin (1954) and Leningrad (1963) congresses are included. The Fourteenth Congress that was held in Stockholm, Sweden in June 2012 was attended by 602 participants from 33 countries. The Fifteenth Congress will be held in August 2016 in Kuching, Sarawak, the first time in a tropical country.

Publications

One of the major tasks of the IPS is publication of information of interest to members and the general scientific community. Besides publishing scientific papers in the International on-line journal **Mires and Peat** and **proceedings** of congresses, conferences and symposia, the International Peat Society also publishes books, policy advisory information and statements, a monthly e-mail newsletter **Peat News** and, twice annually, the magazine Peatlands International. *Peatlands International* consists of about 50-60 pages of background reports on peat and peatlands, reviews of conferences and books, research findings, business reports and internal IPS information.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The magazine for members and friends of the International Peat Society

International Collaboration and Links

Over the years, formal and informal collaboration and links have been established with a number of other organisations with an interest in peatlands and peat including UNESCO, FAO, IUFRO, ISHS, ISSS, IMCG, SWS. IPS has observer status on the Scientific Technical Review Panel of the Ramsar Convention. Individual members of IPS are involved in many national and international agencies and conventions that have interest in peatlands and peat, including CBD and UNFCCC.

Mires and Peat - The joint Scientific Journal of IPS and IMCG

Mires and Peat is an on-line, open access scientific journal published jointly by the International Peat Society and the International Mire Conservation Group (IMCG). It was launched in 2006. Mires and Peat publishes high-quality research papers on all aspects of peatland Science, technology and wise use, including:

- ecology, hydrology, survey, inventory, classification, functions and values of mires and peatlands;
- scientific, economic and human aspects of the management of peatlands for agriculture, forestry, nature conservation, environmental protection, peat extraction, industrial development and other purposes;
- biological, physical and chemical characteristics of peat; and
- climate change and peatlands.

Short communications and review articles on these and related topics are also considered; and suggestions for special issues of the Journal based on the proceedings of conferences, seminars, symposia and workshops are welcomed. The submission of material by authors and from countries whose work would otherwise be inaccessible to the international community is particularly encouraged.

International Water Management Institute 86

James Clarke and Mathew McCartney

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Abstract

The International Water Management Institute (IWMI) is a non-profit, scientific research organization focusing on the sustainable use of water and land resources in developing countries. IWMI works in partnership with governments, civil society and the private sector to develop scalable agricultural water management solutions that have a real impact on poverty reduction, food security and ecosystem health. IWMI is a member of CGIAR, a global research partnership for a food-secure future.

Keywords

Food security · Poverty alleviation · Research · Water

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Introduction

The International Water Management Institute (IWMI) is headquartered in Colombo, Sri Lanka, with regional offices across Asia and Africa. It is one of 15 CGIAR centers which working together and with partners seek to advance agricultural science and innovation to enable poor people, especially women, to better nourish their families, and improve productivity and resilience so they can share in economic growth and manage natural resources in the face of climate change and other challenges seek a mission.

IWMI leads the CGIAR Research Program on Water, Land and Ecosystems (WLE) which combines the resources of 11 CGIAR centers, the Food and Agriculture Organization of the United Nations (FAO) and numerous national, regional and international partners to provide an integrated approach to natural resource management research. WLE promotes a new approach to sustainable intensification in which healthy functioning ecosystems are seen as a prerequisite to agricultural development, resilience of food systems and human well-being.

Mission and Vision

The institute's mission is *to provide evidence-based solutions to sustainably manage water and land resources for food security, people's livelihoods and the environment*. IWMI's vision is for *a water-secure world*.

IWMI works through collaborative research with many partners in the North and South, and targets policymakers, development agencies, individual farmers and private sector organizations.

From the immense damage caused by water-related disasters to the growing problems of water scarcity, IWMI research is seeking solutions to complex problems. Its focus is on the world's poorest and most vulnerable people.

A Brief History of IWMI

IWMI was founded as IIMI, the International Irrigation Management Institute, in 1985. Its original focus was on poorly performing irrigation systems across Asia. IIMI made a major contribution to the improvement of irrigation management by working with stakeholders at all levels on the design, operation and maintenance of irrigation systems. The wealth of knowledge and capacity developed during those early years remains in place today and has fuelled further expansion beyond the Institute's original mandate.

IWMI's early research revealed that water management problems are rarely solely technical, but are often caused by institutional inadequacies. This led to the institution's current emphasis on an interdisciplinary approach to water management. This paradigm has resulted in considerable advances in areas such as water rights, pricing and financing of water resources. IWMI has looked at ways to create positive incentives for farmers and policy makers to manage water more equitably and efficiently. IWMI also carried out some of the early and most significant work on gender and irrigation.

IWMI's unique mix of technical specialists and social scientists has enabled it to tackle complex problems of water management that have confounded the best efforts of policy makers and water users. Pioneering work on groundwater governance, formal and informal water institutions, and broader issues of water and society has made IWMI a market leader in resource management research for policy solutions.

IWMI was awarded the Stockholm Water Prize in 2012.

Research Themes

IWMI's research examines the "water-food-environment nexus", adopting a multi-disciplinary approach and balancing efficiency and productivity objectives with equity and sustainability concerns.

Seven key Research Themes help IWMI to more effectively produce impact for the benefit of poor farmers, whose livelihoods depend on agriculture and for whom access to water for productive purposes is a key constraint. IWMI recognizes the scope for a more proactive role in knowledge application to increase impact-particularly by interacting closely with other research institutes, policymakers, donors, partners and user communities.

1. **Ecosystem Services** recognizes the centrality of ecosystem services for the sustainable intensification of agriculture, and investigates how landscapes, including rivers and wetlands, can be better managed to deliver ecosystem services that are critical for livelihoods and people's well-being
2. **Governance, Gender and Poverty** focuses on promoting gender and social equity, and the sharing of benefits across borders. The theme generates research evidence to foster new and more effective policies, governance and institutional arrangements, and innovative management solutions that seek to optimize the benefits to key stakeholders, including disadvantaged women, men and youth;
3. **Resource Recovery, Water Quality and Health** seeks opportunities to mitigate the negative impacts of human activities on water quality, the environment and health. The theme's current research focuses on safe wastewater use, and business models for resource recovery from wastewater, fecal sludge, and domestic and agro-industrial waste.
4. **Revitalized Irrigation Systems** addresses the complex issues of improving irrigation system productivity and sustainability, looking at policy, institutions and management and technical aspects and solutions. Work includes the benchmarking of irrigated systems and other innovative management tools, as

well as exploring opportunities for scaling up appropriate modernization technologies.
5. **Sustainable Agricultural Water Management** works on sustainable intensification of rainfed agricultural and aquatic systems. The theme investigates technical, policy and institutional interventions to improve access to and management of productive water and land resources for poor and vulnerable agricultural communities.
6. **Water Availability, Risk and Resilience** incorporates hydrological and hydrogeological assessments at multiple scales under current and future climate scenarios, provides services to other themes for modelling of water resources scenarios, and enhances availability and delivery of water resources data through the use of earth observing system satellites and information and communications technology.
7. **Water Futures** conducts research to improve the understanding of the interplay between the decision-making environment and the behaviour of the physical hydrological systems.

Wetland Research at IWMI

IWMI recognizes the importance of wetlands and the vital role that they play in supporting and enhancing the livelihoods of people in developing countries. In many of the countries that IWMI works, wetland agriculture brings opportunities as well as threats. Farmers can enhance the natural productivity of wetlands and a significant proportion of the value of many wetlands is derived from agriculture. However, if insufficient attention is paid to other wetland services, in particular regulating and socio-cultural benefits, wetland degradation results, with a concurrent decrease in agricultural potential. The dilemma remains how to maximize the benefits of agriculture whilst simultaneously minimizing the adverse impacts on other valuable ecosystem services.

IWMIs works on two key objectives for wetland research:

- To protect ecosystem services, livelihood and water productivity in wetlands through better knowledge of how agricultural activities impact wetland functions.
- To reduce poverty through improved agriculture, fisheries, and livestock management in wetlands via identification of effective management strategies and policies to protect vital ecosystem services.

IWMI was elected an International Organization Partner (IOP) of Ramsar at the Convention of the Parties in 2005. IWMI scientists contribute to the Working Groups of the Conventions Scientific and Technical Review Panel (STRP). Over the past few years, IWMI has increased understanding of both the value of wetland agriculture for poverty alleviation and livelihoods as well as the impact of agriculture on wetlands. IWMI has also made significant investments to raise awareness and ensure dissemination and uptake of this knowledge.

Building Capacity

IWMI has a long standing commitment to develop the talents of scientists in the developing world. Hundreds of local researchers have worked for the institution, many gaining PhD's or other qualifications. Many of these individuals are now senior figures in water management and research in Africa and Asia.

Outreach and Uptake

IWMI has consistently sought to turn its science into action. It is strongly committed to the notion that its publicly funded research should be regarded as international public goods, freely available to as many people as possible. IWMI publishes policy briefs and syntheses of current thinking which are available online and though leading digital repositories worldwide. IWMI further disseminates its work online making extensive use of social media, mainstream news outlets, multimedia and other digital formats. IWMI's Water Data Portal allows the public to access its databases online. Its remote sensing and geographical information systems facility, GRandD, gives researchers across the world improved access to spatial data.

Recent Publications

The following are examples of IWMI's research reports in the field of wetland management:

de Silva, S., Senaratna Sellamuttu, S. (2010). *Balancing wetland conservation and development in the Sanjiang Plains: a review of current status and options. Sanjiang Plain Wetlands Protection Project, final report.* Colombo Sri Lanka: International Water Management Institute (IWMI). 81p.

International Water Management Institute (IWMI) (2014) *Wetlands and people.* Colombo, Sri Lanka: International Water Management Institute (IWMI) 32p.

McCartney, M., Rebelo, L-M., Senaratna Sellamuttu, S., de Silva, S. (2010). *Wetlands, agriculture and poverty reduction.*Colombo, Sri Lanka: International Water Management Institute (IWMI). 31p. (IWMI Research Report 137.

Senaratna Sellamuttu, S., Mith, S., Hoanh, C. T., Johnston, R. M.. Baran, E., Dubois, M., Soeun, M., Craig, I., Nam, S., Smith, L. (2010). *Commune agroecosystem analysis to support decision making for water allocation for fisheries and agriculture in the Tonle Sap Wetland System.*Colombo, Sri Lanka: CGIAR Challenge Program on Water and Food (CPWF). 58p. (CPWF Project Report 71)

Society of Wetland Scientists

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Robert J. McInnes

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Abstract

The Society of Wetland Scientists (SWS) is an international organization of approximately 3,500 members dedicated to fostering sound wetland science, education, and management. The Society was formed in March 1980. Arranged globally in chapters and working through specialist interest groups, the mission of the Society of Wetland Scientists is to promote understanding, scientifically based management, and sustainable use of wetlands.

Keywords

Non-governmental organization · International cooperation · Wetland conservation

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Background

The Society of Wetland Scientists (SWS) is an international organization of approximately 3,500 members dedicated to fostering sound wetland science, education, and management. The Society was formed in March 1980. It was the inspiration of Richard Macomber, a biologist with the US Army Corps of Engineers Board of Rivers and Harbors. Back in the 1970s wetland ecologists at the Corps of Engineers did not have access to a suitable forum for discussing wetland science issues free from political concerns. The Society gave them the opportunity to open up dialogues with their colleagues and beyond.

Today, SWS is a dynamic society with members on every continent (with the exception of Antarctica). The Society represents a rich cross-section of disciplines and backgrounds including members from academia, government officers, private consultancies, nongovernmental organizations, and intergovernmental organizations. The Society is assembled into regional chapters both within the USA and beyond. The Society currently recognizes the following chapters:

- USA – Alaska
- USA – Central
- USA – Mid-Atlantic
- USA – New England
- USA – North Central
- USA – Pacific Northwest
- USA – Rocky Mountains
- USA – South Atlantic
- USA – South Central
- USA – Western

The Society also recognizes the following non-USA chapters:

- Asia
- Oceania
- Canada
- Europe
- South America
- International (all other countries)

In addition to the regional chapters, the society also supports special interest sections which allow members to network and promote activities that address subject areas ranging from science to policy. As of spring 2014, the Society has the following seven special interest sections: Biogeochemistry, Global Change Ecology, Peatlands, Ramsar, Restoration, Wildlife, and Women in Wetlands.

Mission and Goals

The mission of the Society of Wetland Scientists is to promote understanding, conservation, protection, restoration, science-based management, and sustainability of wetlands.

To achieve this mission, the Society of Wetland Scientists will:

- Be recognized as the world leader in wetland science
- Lead the education of wetland professionals and the public regarding wetland science and its applications
- Promote the development of wetland science programs around the world, particularly in developing countries
- Foster diversity among SWS members and other wetland professionals
- Promote sound science in wetland policy and stewardship

The Society of Wetland Scientists advocate that wetland science will:

- Form the basis of the public's understanding of the important ecosystem services of wetlands
- Constitute a rigorous interdisciplinary endeavor and body of knowledge
- Provide the foundation for management, restoration, conservation, protection, and policy

Publications

SWS produces an international journal concerned with all aspects of wetlands biology, ecology, hydrology, soils, biogeochemistry, management, laws, and regulations called *Wetlands*. The journal is recognized as the premier outlet for communicating research to an expanding community of international and interdisciplinary wetland professionals. The journal includes full length articles, short notes, and timely review articles.

In addition to the journal, SWS also publishes a quarterly publication called *Wetland Science & Practice* which showcases examples of wetland issues and acts as a communication tool across the Society's membership.

Meetings

SWS holds an annual meeting which brings together wetland professionals from around the world. In addition to the main annual SWS meeting, individual chapters hold their own meetings at a local level.

For more information on the Society of Wetlands Scientists see www.sws.org.

South Africa's National Wetland Rehabilitation Programme: Working for Wetlands

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John A. Dini and Umesh Bahadur

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Abstract

Like many other countries, South Africa has experienced high levels of wetland loss and degradation as a result of human activities. A vehicle through which to undertake much needed extensive wetland rehabilitation was provided through government's Expanded Public Works Programme, which seeks to provide work and training to significant numbers of unemployed people. The resulting programme, Working for Wetlands, now invests substantial public funding in the rehabilitation and wise use of wetlands in a manner that maximizes employment creation, supports small emerging businesses, and transfers skills to its beneficiaries. A series of case studies produced through a four year research

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programme illustrates some of the benefits of this work, including improved livelihoods, protection of agricultural resources, enhanced biodiversity, cleaner water and reduced impacts from flooding.

Keywords
Wetland rehabilitation · South Africa · Working for Wetlands Programme · Poverty reduction · Livelihoods

Background

South Africa's wetlands collectively play an important role in sustaining the country's ecology and economy. Yet despite these vital linkages between wetlands and people, wetlands have been severely affected by the same human activities that have dramatically altered South Africa's landscapes over the past few centuries. Although no systematic national survey of wetland loss has been undertaken, studies in several major catchments have revealed that between 35% and 60% of the wetlands, and the benefits they provide, have been lost or severely degraded (Kotze et al. 1995). It is likely that the extent of wetland loss for the country as a whole lies within this range, and consequently it was not surprising that the 2011 National Biodiversity Assessment identified wetlands as the most threatened ecosystem type in South Africa (Driver et al. 2012).

A pivotal response by the government to this state of affairs was the establishment in 2000 of a national wetland rehabilitation programme, known as Working for Wetlands. The decision to create such a program came about through the convergence of several driving forces. It drew on objectives in environmental, biodiversity, water, and agriculture policies, and capitalized on the growing recognition that wetland degradation is not necessarily permanent, and that it is possible to reinstate at least some ecosystem services through rehabilitation. A foundation was provided for the creation of the program, in the form of another pioneering government initiative. Since 1996, the Working for Water programme has been engaged in removing thirsty invasive alien plants that pose a threat to the country's water security, agricultural productivity, and biodiversity. The nongovernmental Mondi Wetlands Project recognized that the labor-intensive model pioneered by Working for Water would be equally suited to the activities involved in rehabilitating wetlands and lobbied government to begin experimenting in this direction.

Perhaps the most significant factor enabling the emergence of Working for Wetlands was the availability of government funds earmarked for employment creation and poverty reduction, through the Expanded Public Works Programme (EPWP). This government-wide initiative was set up to draw significant numbers of unemployed people into the productive sector of the economy, gaining skills while they work and increasing their capacity to earn income. The ability to turn wetland rehabilitation into a labor-intensive process unlocked a magnitude of financial resources and political support that was previously inconceivable to cash-strapped government departments responsible for biodiversity conservation and natural resource management.

Thus, Working for Wetlands pursues its mandate of wetland rehabilitation and wise use in a manner that maximizes employment creation, supports small emerging businesses, and transfers skills to its beneficiaries. In line with EPWP norms, the program targets those groups most excluded from the mainstream economy, with particular emphasis on women, youth, and people with disabilities.

All rehabilitation interventions aim to improve the condition and functioning of the ecosystem and address both causes and effects of degradation. Typical project activities include:

- Building concrete, earthen, or gabion structures to arrest erosion and trap sediment, and resaturate drained wetland areas
- Plugging artificial drainage channels
- Addressing other causes of degradation, such as poor agricultural practices and invasive alien plants
- Plant propagation, revegetation, and bioengineering
- Building boardwalks, bird hides, and interpretive signboards to enhance the recreational, tourism, and educational value of rehabilitated wetlands
- Concluding contractual agreements with landowners to secure the rehabilitation work, prevent further degradation of wetlands, and influence land use practices
- Providing community members with part-time employment and training to monitor completed rehabilitation once the work is completed

Since its formation in 2000, Working for Wetlands has on average rehabilitated 79 wetlands per year, thereby improving or securing the condition of approximately 6,150 ha of wetland area per year. In the process, the program has provided employment for an average of 1,650 people each year. In line with the emphasis of the EPWP on training, Working for Wetlands has provided a yearly average of 17,300 days of training in both vocational and life skills.

This large-scale investment of public funds in wetland rehabilitation stimulated a range of supporting activities, including the publication by the Water Research Commission in 2008 of an eleven volume series of reports, manuals, and guidelines for wetland assessment and rehabilitation (Dada et al. 2007). This helped to strengthen the scientific and technical foundation for the program's work which, together with an investment in wetland inventory and classification, enhanced the scope to plan and undertake systematic rehabilitation at catchment scale.

Benefits accruing from the wetlands rehabilitated by Working for Wetlands include improved livelihoods, protection of agricultural resources, enhanced biodiversity, cleaner water, reduced impacts from flooding, and sustained baseflows in rivers (Kotze and Ellery 2008). A series of case studies that illustrate some of these outcomes are presented below.

Case Study 1: Healthy Wetlands, Healthy People: Livelihoods and Rehabilitation of Manalana Wetland

The Manalana wetland, near Bushbuckridge in Mpumalanga Province, was severely degraded by erosion that threatened to consume the entire system if left unchecked. Altogether about 70% of the local population make use of the wetland in some way. These wetland users are typically women between 35 and 70 years of age from single-headed households. The wetland was thus considered to offer an important safety-net, particularly for the poor, contributing about 40% of the food grown locally. As a result, Working for Wetlands partnered with a locally based NGO in 2006 to stabilize erosion and improve the wetland's ability to continue providing its beneficial services, while the NGO worked with wetland users to implement more sustainable cultivation practices.

An economic valuation study completed in 2008 (Pollard et al. 2008) revealed that:

1. The annual value of livelihood benefits derived from the degraded wetland was a mere 34% of what could be achieved with an investment in rehabilitation.
2. After rehabilitation, the wetland now contributes provisioning services conservatively estimated at 3,466 Rand (about USD 392) per household per year to some 70% of local households, in an area where 50% of households survive on an income of less than 5,700 Rand (USD 644) per year.
3. The net present value of the livelihood benefits (1,995,885 Rand or USD 225,500) provided by the rehabilitated wetland over a 50-year period is more than double the cost of the rehabilitation interventions (947,328 Rand or USD 107,030), including monitoring and maintenance over the same period, indicating a very favorable return on Working for Wetlands' investment.
4. Manalana wetland acted as a safety net that buffered households from slipping further into poverty during times of shock or stress.

Case Study 2: Restoring the Kidneys of the Landscape

The diverse Rietvlei wetland system is situated immediately upstream of the Rietvlei Dam within a 4,000 ha nature reserve just outside the capital city of Pretoria. The dam has provided Pretoria with drinking water since 1934, producing about 41 million liters per day, or 3% of the city's current requirement. Until recently, the Rietvlei wetlands were heavily eroded and desiccated, having been drained for cultivation and peat mining before the area was proclaimed a nature reserve.

In recent years, the dam has become severely overloaded with nutrients and other pollutants, as its highly urbanized catchment has received increasing volumes of treated domestic sewage and industrial effluent. As a result, the dam is plagued by blooms of blue-green algae, which cause bad tastes and odors in the water that are difficult to remove and require expensive treatment, as well as posing potential threats to health.

Partly in response to this situation, in 2000, Working for Wetlands formed a partnership with the municipality to rehabilitate wetlands upstream of the dam, with the primary objective of improving their ability to purify the water flowing into the dam. Interventions included gabion; concrete; and earthen structures to control erosion, rewet the organic soils, increase retention time of water, and ensure even distribution of flow across the wetland.

Monitoring results show that the rehabilitated wetlands are improving the quality of water flowing into the dam (Masupa and Makhado 2006) with ammonia levels down by 53%, nitrates by 77%, fluoride by 24%, and sulfates by 4%, compared to upstream of the wetlands. This reduction in pollutants entering the dam is contributing to reduced algal growth, thereby reducing the costs of treating the water for human consumption.

Case Study 3: Wetlands and Flood Mitigation in the Krom River

The Krom River in the Eastern Cape Province previously contained some of the largest wetlands of their type in South Africa. However, it is estimated that half of these have been lost as a result of infestation by alien vegetation and destructive human activities such as large-scale cultivation on floodplains. The catchment, which supplies the city of Port Elizabeth with about 40% of its water, has been the focus of concerted action by Working for Wetlands and Working for Water to rehabilitate wetlands and eradicate invasive alien plants. Since 2001, ten large structures have been built to combat erosion that threatened the remaining large intact wetlands.

In 2006, the Krom River experienced its most severe floods since measurements began in 1938, resulting in heavy loss of life and property. The remaining wetlands played a key role in managing the floods, slowing the velocity and destructive potential of the floodwaters, and trapping sediment. The wetlands proved crucial for reducing further potential damage downstream. The Working for Wetlands structures accomplished their purpose and the two main wetland basins emerged from the floods largely unscathed.

There was no happy ending for the rest of the valley. Where wetlands had been destroyed to make way for cultivated fields, parts of the floodplain and river banks were carried away by the floodwaters. Large volumes of sand were dumped onto the remaining fields, and the rest of the sediment was deposited in the Churchill Dam further downstream. In some places, the river was gouged down to bedrock level. Infrastructure and houses were swept away, and many fields on the floodplain were completely destroyed.

Case Study 4: The Story of Pilot Malele

Mr Pilot Malele was unemployed prior to joining Working for Wetlands in 2002. He had completed his secondary schooling, but financial constraints prevented him from continuing his education further. Prior to joining the program, Pilot relied on irregular, short-term, informal jobs to survive, but these did not contribute to improving his quality of life.

Working on the Sand River project, Pilot seized the training opportunities made available through the program and was appointed as a contractor, managing a team of 12 people drawn from his community. The team's main task involved stabilizing gully erosion through the construction of gabions, sloping, and revegetation of the wetland.

Participating in an evaluation of the socioeconomic impact of Working for Wetlands, Pilot indicated that his life had improved tremendously since he joined the program (Schuyt 2005). The formalized and secure employment enabled him to start a family, build a house for his wife and children, and build additional rooms to accommodate his parents and siblings. A particular source of satisfaction was being able to provide his children with the education that his parents could not afford to provide for him. A major highlight of his life was being able to buy a car, something he never dreamt of growing up in a poor family.

Together with another Working for Wetlands contractor, Pilot used the skills he gained through the program to start a company in order to compete for government tenders. Pilot's story, and that of many others employed through Working for Wetlands and similar programs, shows how such opportunities, coupled with honest effort and determination, can change lives and surmount the most disadvantaged of circumstances.

Future Challenges

Despite these successes, significant challenges remain. The program is evaluating whether its protocols for site selection, and the types of rehabilitation interventions chosen, produce the optimal return on investment. This reflection derives from the realization that the program's footprint, in terms of number of wetlands rehabilitated per year, is dwarfed by the number of wetlands requiring rehabilitation by two orders of magnitude. Working for Wetlands is also being challenged to expand its activities to include securing wetlands that are in good condition, rather than the current focus on rehabilitation of degraded wetlands. The major source of the program's funding, the EPWP, continues to dictate the types of activities that can be undertaken. With strong political pressure to maximize job creation, there is an ever-present risk of the means (labor-intensity) becoming the end. Nonetheless, the program is diversifying its funding sources, with increasing interest being shown by private landowners, especially forestry companies. A fledgling mitigation banking system, being piloted by regulatory authorities, also shows promise as a provider of funding and driver of demand for rehabilitated wetlands. Lastly, as the cumulative number of wetlands rehabilitated increases every year, the challenge of monitoring and maintenance of rehabilitated wetlands, and ongoing engagement with landowners and wetland users, grows correspondingly.

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The Nature Conservancy (TNC)

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C. Max Finlayson

Contents

Abstract

The Nature Conservancy (TNC) is a membership-based nongovernmental organization with headquarters in the USA. It has more than one million members and addresses conservation issues across many landscapes and ecosystems, including rivers and wetlands, as well as coastal and marine systems. It works globally to ensure the conservation and restoration of ecologically important lands and waters and to support the ongoing provision of ecosystem services that provide many benefits for people. It comprises a network of offices and field stations across all states of the USA and in more than 35 countries.

Keywords

Nongovernmental organization · Conservation · Restoration · Ecosystem services

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Introduction

The Nature Conservancy (TNC) is a well-established membership-based nongovernmental organization with headquarters in Arlington, Virginia, in the USA. It has more than one million members and addresses conservation issues across many landscapes and ecosystems, including rivers and wetlands, as well as coastal and marine systems. It works globally to ensure the conservation and restoration of ecologically important lands and waters and to support the ongoing provision of ecosystem services that provide many benefits for people. It comprises a network of offices and field stations across all states of the USA and in more than 35 countries.

Its conservation work is based on the best available science, including that undertaken by its own staff as well as by many partner organizations, including governments, research institutions, and nongovernmental and community-based organizations. Information about the organization and its current activities are presented in the Annual Reports (see The Nature Conservancy 2014a) and through an extensive webpage (www.nature.org).

It is governed as a single, tax-exempt organization by a worldwide, volunteer board of directors and is managed from its worldwide office in Arlington. It is organized as a single organization rather than as separate local legal entities with ultimate responsibility for all operations being with the board. Responsibility for day-to-day operations is delegated to the president and chief executive officer with each state and country program being run by a director. Programs in the USA and several in other countries are advised by volunteer boards of trustees.

History

TNC was formerly known as the Ecologists Union that was formed in 1946 with the purpose of taking direct action to save threatened areas. The name was changed to The Nature Conservancy in 1950, and it was incorporated as a nonprofit organization in the District of Columbia in the USA in 1951. Partnerships, including those with governments and the private sector, are at the core of TNC's approach for ensuring it has the means to support public policies that protect and sustain the many and varied ecosystems and biota globally. Its main conservation mechanism for many years has been the Land Preservation Fund that has successfully enabled the organization to provide repayable loans for conservation activities.

The history and milestones of TNC's extensive conservation activities are documented on its webpages and in its annual reports. A number of examples of activities addressing wetlands and wetland management issues are listed below. These are extracted from 2014 Annual Report (The Nature Conservancy 2014a) with further information being available on the webpages in other publications (see www.nature.org/science-in-action/our-scientists/keep-up-with-our-scientists.xml?intc=nature.tnav.science.list).

1. **Caribbean Expansion** – On its southern peninsula and northern coast, Haiti has established its first marine protected areas (MPAs), encompassing a total of 402,881 acres, using conservancy assessments and mapping. TNC scientists are now assisting the Caribbean nation in managing its new MPAs, employing drones to help map and monitor the most important coastal sites. The conservancy is also training the staff of Cuba's National Center for Protected Areas.
2. **Gulf Coast Land and Water –** Powderhorn Ranch, a mosaic of dense live oak forests, coastal prairies, salt marshes, and wetlands, was secured by a partnership between the conservancy, The Conservation Fund, and the Texas Parks and Wildlife Foundation. The ranch's tidal bayfront protect vitally important seagrass beds and mollusc reefs. The National Fish and Wildlife Foundation funded a significant portion of this project using fines from the Deepwater Horizon oil spill. Powderhorn is slated to become a state park with ownership turned over to the Texas Parks and Wildlife Department.
3. **Technology Aids Birds** – Using satellite images and data crowdsourced from eBird.org, TNC scientists mapped when and where migratory birds, including sandhill cranes, need habitat in California's Central Valley. A TNC team, including an economist, then implemented a first-of-its-kind auction to rent and flood farm fields at the exact time and location needed by the birds. The pilot was a success, with more than 40 participating farms creating additional areas of wetland habitat at a critical time for migratory birds.
4. **The Colorado Flows –** Using an innovative water-banking model and a trust to acquire water rights, TNC joined with partners to ensure that the Colorado River delta received dedicated water flows for the first time in nearly half a century. Due to over-allocation of the Colorado River for cities, agriculture, and other uses in the USA and Mexico, the river no longer reaches the Gulf of California – harming fish and shrimp habitat – and has stopped regularly flowing through its delta, a critical habitat for resident and migratory bird species.
5. **Re-creating Floodplains –** TNC has joined with public and private partners to launch Floodplains by Design, an innovative effort along the major rivers that flow into Washington state's Puget Sound. Over the coming 20 years, the effort will re-engage more natural floodplains throughout the state that have been cut off from rivers because of residential, agricultural, and industrial development and earlier efforts to control floods. Taxpayers have paid more than $1.4 billion in flood damage in the area since 1990. The large-scale effort seeks to restore salmon populations, reduce flood hazards, increase agricultural viability, improve water quality, and enhance outdoor recreation.

Mission

The mission of The Nature Conservancy is "to conserve the lands and waters on which all life depends," and the vision is "a world where the diversity of life thrives, and people act to conserve nature for its own sake and its ability to fulfill our needs and enrich our lives." The mission and vision are achieved through the activities and efforts of its members, partners, and employees and guided by a framework called Conservation by Design that provides a systematic approach to determine where to work, what to conserve, what strategies should be used, and how effective these have been (The Nature Conservancy 2014b). The approach outlined below sets out the values by which TNC works:

1. **Integrity Beyond Reproach**: We will meet the highest ethical and professional standards in all of our organizational endeavors, and, in doing so, we hold ourselves accountable to our mission and to the public.
2. **Respect for People, Communities, and Cultures**: Enduring conservation success depends on the active involvement of people and partners whose lives and livelihoods are linked to the natural systems we seek to conserve. We respect the needs, values, and traditions of local communities and cultures, and we forge relationships based on mutual benefit and trust.
3. **Commitment to Diversity**: We recognize that conservation is best advanced by the leadership and contributions of men and women of diverse backgrounds, beliefs, and cultures. We recruit and mentor staff to create an inclusive organization that reflects our global character.
4. **One Conservancy**: Our strength and vitality lie in being one organization working together in local places and across borders to achieve our global mission. We value the collective and collaborative efforts that are so essential to our success.
5. **Tangible, Lasting Results**: We use the best available science, a creative spirit, and a nonconfrontational approach to craft innovative solutions to complex conservation problems at scales that matter and in ways that will endure.

From an increasing position of strength, based on successful fundraising and strong partnerships, TNC has committed to using its resources wisely, to achieving effective conservation outcomes, and to set even higher goals in the face of increasing pressure from, for example, deforestation, overfishing, and carbon emissions (The Nature Conservancy 2014a).

For many years, TNC has seen its workplace as the whole planet and has focused on global challenges to land, water, oceans, cities, and climate. This requires an expanding effort to "protect and restore landscapes, rivers and oceans at an unprecedented scale" and to "transform how we use the world's natural resources by affecting policy and practices locally and globally" and "inspire global action by people who value nature and its role in ensuring thriving communities and dynamic economies" (The Nature Conservancy 2014a).

Ongoing Activities

The need to protect nature is what propelled The Nature Conservancy into existence in 1951. Initially, protection took the form of buying land and setting it aside, using a mix of land purchase and partnership arrangements. This led to the adoption of a suite of diverse actions that extended across entire land and water systems, including areas where people obtained their daily sustenance directly from nature.

TNC has undertaken a large number of wetland-oriented activities including the establishment of freshwater and coastal/marine protected areas and the instigation of conservation and restoration activities. These have included efforts to determine and deliver appropriate environmental flows to rivers and floodplains that had formerly been degraded by the diversion of water for consumptive purposes. This included supporting the 10th international river symposium and international environmental flows conference, held in Brisbane, Australia, September 2007 (Anon 2007). The conference was attended by more than 800 participants who agreed the Brisbane Declaration and affirmed that "Environmental Flows are essential for freshwater ecosystem health and human well-being" and supported previous research conducted by TNC for ensuring the sustainable management of global water resources (Richter 2014).

TNC has also embarked on many wetland-related projects in the coastal and marine environment with one recent initiative directed towards developing methods to restore oyster reefs. These are a natural component of the marine environment that have suffered extensive degradation with an estimated 85% being lost. Given their value for biodiversity and, for example, coastal protection, there are increasing efforts to restore them and have them recognized through the Ramsar Convention on Wetlands (Kasoar et al. 2015). These efforts are leading to community-government partnerships to support the effort needed to repair the damage done in the past and restore at least some of the coastal diversity that previously occurred in many locations around the world, and highlighting the type of work that the TNC has successfully undertaken over many years.

Future Challenges

The challenge for TNC is to continue to advance conservation around the world through working with scientists and collaborators at the local and global scale to leverage results and to confront conservation threats at the largest scale.

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WetlandCare Australia

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Louise Duff and Cassie Price

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Abstract

WetlandCare Australia (WCA) is a leading not-for-profit company that has been working with communities to protect, restore, and promote healthy wetlands since 1991. WetlandCare Australia was founded to undertake practical projects with long-term benefits for wetlands across Australia. The organization implements a broad range of natural resource management and capacity building projects.

Keywords

Australia · Capacity building · Nongovernmental organization

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Introduction

WetlandCare Australia (WCA) is a leading not-for-profit company that has been working with communities to protect, restore, and promote healthy wetlands since 1991. WetlandCare Australia was founded to undertake practical projects with long-term benefits for wetlands across Australia. The organization implements a broad range of natural resource management and capacity building projects.

WetlandCare Australia has a board of directors who fulfill their role on a voluntary basis. WetlandCare Australia's directors have broad experience, bridging wetland management, conservation, and administration. A staff of 15 covers management and administration, ecology, education and communications, natural resource management, and on-ground work. The organization has five offices on the east coast of Australia from the city of Newcastle in the Hunter Region of New South Wales to Townsville in the wet tropics of Queensland.

As a nonpolitical science-based organization, WetlandCare Australia is dedicated to delivering effective and sustainable outcomes through the use and application of science-based techniques in the restoration and preservation of wetlands. To achieve this, the organization is supported by a Scientific Advisory Panel.

Major Areas of Activity

WetlandCare Australia's conservation strategy sets the organization's goals across five program areas:

Great Barrier Reef Catchments: Improving the health, function, and resilience of aquatic ecosystems across Great Barrier Reef catchments to protect the Reef

Blue Carbon: Mainstreaming a blue carbon agenda into national climate change initiatives

Eastern Wetlands and Water: Improving conservation, management, and sustainable use of wetlands and environmental water across the east coast of Queensland and New South Wales

Australian Wetland Biodiversity: Promoting protection and conservation of Australia's unique wetland flora and fauna

Ramsar Program: Undertaking communication, education, participation, and awareness activities to improve recognition and build capacity for the implementation of the Ramsar Convention in Australia

WetlandCare Australia achieves these goals by working in collaboration with government agencies, landholders, indigenous Australians, community organizations, and business sector. The scope of work the organization undertakes includes:

Wetland classification, assessment, and mapping
Wetland sub-catchment management planning
Wetland restoration and rehabilitation

Wetland protection and conservation
Communications, education, conferences, workshops, seminars, and webinars
Project monitoring, analysis, reporting, and evaluation

Landmark Contributions

WetlandCare Australia's first major project was the restoration of Banrock Station, a 3,400 ha complex of wetlands and native vegetation in South Australia. The project commenced in 1992 when volunteers started a rehabilitation project by installing flow control structures.

In 1994, BRL Hardy Wine Company purchased the property and commenced export production of wine under the Banrock Station label. The company produces over 30 million bottles of wine annually and continued to work in partnership with WetlandCare Australia for almost a decade. The project team manipulated water flow into the Banrock Swamp to reduce the number of introduced European carp, excluded sheep from the swamp, and eradicated rabbits by introducing a control virus. In 1995, the company expanded its rehabilitation program to address management issues on the adjoining Banrock Station Wetland Complex and established a sponsorship and education program to support wise use of wetlands in Australia and internationally.

Through the dedicated work of WetlandCare Australia and Banrock Station Wines, this wetland complex stands as a model for the Ramsar Convention's wise use principle and has now been listed as a Wetland of International Importance under the Ramsar Convention.

Major Projects

WetlandCare Australia has a number of major projects underway.

Coastal 20

WetlandCare Australia is working in partnership with community, government, and industry bodies to rehabilitate 20 wetlands on the east coast of Queensland and New South Wales. These 20 coastal wetlands are biodiversity hotspots and have regional and international significance. They face significant and increasing human and environmental pressures threatening their environmental, cultural, social, and economic values. WetlandCare Australia established regional stakeholder control groups to identify priorities and consult on threat management. The project team undertook ecological assessment of all sites and developed Site Action Plans agreed by the control group. On-ground works include fencing, weed control, feral animal control, community engagement, and education. Coastal 20 is a 2-year program from July 2011 to June 2013 and is funded by the Australian Government through its Caring For Our Country program.

Hunter Estuary Shorebird Protection Program

WetlandCare Australia has a 2-year program of works underway to enhance habitat and control threats to migratory and resident shorebirds in the Hunter Wetlands National Park in New South Wales. The project includes revegetation, weed control, fox control, bird monitoring surveys, and community education. It is being undertaken in collaboration with the NSW National Parks and Wildlife Service, Hunter Bird Observers Club, Hunter-Central Rivers Catchment Management Authority, and Hunter Wetlands Center. The project time frame is July 2011–June 2013 and is funded by the New South Wales State Government through its Environmental Trust.

Maintaining Australia's Biodiversity Hotspots

The Australian Government's Maintaining Australia's Biodiversity Hotspots program used landholder stewardship agreements to support a broad range of management actions on private land tenures nationally. WetlandCare Australia facilitated implementation of the program in the Torrington and Ebor-Dorrigo-Coffs Harbor regions of New South Wales. The organization undertook assessment and preparation of site action plans for numerous properties and managed a competitive bidding process resulting in 23 stewardship agreements. Landholders completed a range of management actions including wetland rehabilitation, weed and pest control, grazing exclusion, and fire management over 3,700 ha of native vegetation with high conservation value.

Australian Wetland Network Secretariat

WetlandCare Australia hosts the Secretariat of the Australian Wetland Network. The network represents over 70 nongovernment organizations (NGOs) working to address wetland conservation across Australia. The Australian Wetland Network coordinates consultation and reporting for the NGO sector in the lead up to the triennial Ramsar Conference of Contracting Parties (COPs) and has attended the past two COPs. The Secretary also serves as the Oceania Coordinator for the World Wetland Network.

Sustainable Wetlands on Coastal Landscapes

WetlandCare Australia (WCA) has been working with coastal NSW Catchment Management Authorities (CMAs) to better understand their wetland priorities. Between 2004 and 2011, WCA has compiled comprehensive wetland mapping for each CMA. The mapping is populated with each wetland's conservation values and potential threats to assist in determining priorities for investment in wetland works.

Further to this, WCA developed and provided CMAs with a decision support database that analyzes conservation value and threat and returns a priority list of wetland complexes for a given area. All CMAs continue to utilize this data in their project and investment planning. The result has been targeted on-ground outcomes for priority wetlands in each region.

Delivering Biodiversity Dividends for the Barratta Creek Catchment

WetlandCare Australia has a project underway to protect and improve management of the Barratta Creek Catchment. This catchment forms the main artery of the Bowling Green Bay wetlands, the only Ramsar site in north Queensland. Barratta Creek is one of the most high-integrity floodplain creek systems on the developed east coast of Queensland. Since the introduction of intensive irrigated agriculture, the creek and associated wetlands have suffered impacts including invasive aquatic and terrestrial weeds, altered fires regimes, and excessive and nutrient-rich tailwater flows. WetlandCare Australia is working with multiple stakeholders to improve biodiversity outcomes through integrated catchment management. The project is supported by the Australian Government's Clean Energy Future's program through its Biodiversity Fund.

Starting in Your Backyard

WetlandCare Australia appreciates that community education and engagement are essential to protect human impacts on wetlands. This program engages residents living in wetland catchments to plan and commit to a program of activities that will reduce the impacts of their home and family on wetlands in their sub-catchment. WetlandCare Australia developed resources and undertook a series of face-to-face "train the trainer" workshops and webinars to build the capacity of local environmental educators throughout New South Wales, Australia. The program was funded by the New South Wales Government through its Environmental Trust.

Further information about WetlandCare Australia can be found at www.wetlandcare.com.au

http://www.youtube.com/user/WetlandCareAust?feature=guide

Check for updates

Wetlands International

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Jane Madgwick

Contents

Abstract

Wetlands International is the only global nongovernmental organization (NGO) dedicated fully to the wise use of wetlands. It was established in 1995 through bringing together three regional NGOs: International Waterfowl and Wetlands Research Bureau (IWRB), the Asian Wetland Bureau (AWB), and Wetlands for the Americas (WA). Wetlands International is neither a nature conservation organization nor a development one. Instead, Wetlands International connects conservation and development goals. This is because for wetlands, nature and people's well-being are so interdependent. Wetlands International's dual focus is on people and nature, and its broad array of partners and members, both governments and NGOs puts it in a favorable position to bring many different sectors together to tackle the pressing challenges of our time – food and water security, climate change, and biodiversity loss.

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Keywords
Nongovernmental organization · Wise use · Waterbirds · Conservation · Development · People · Livelihoods

Introduction

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A Network Organization with Strong Roots in Developing Countries

Wetlands International came about as the result of a close working relation between the International Waterfowl & Wetlands Research Bureau (IWRB) that was created in 1954, the Asian Wetland Bureau (AWB) – initiated as INTERWADER in 1983 – and Wetlands for the Americas (WA) formed in 1989. These three organizations started to work closely together in 1991 and joined to form one single organization in 1995, recognizing that global wetland challenges required a larger, concerted global effort.

Wetlands International is a global network organization working worldwide through, by 2016, 18 offices and through a broad range of partnerships in around 100 countries. All offices share expertise and collaborate to achieve the same set of global goals, using a common set of institutional policies and standards. The head office in the Netherlands guides and supports the work of the network while the regional and national offices are responsible for their programs, staffing, and resources. These offices are established as legal entities in their own countries and are linked to the head office through a Global Network Partnership Agreement that enables collaboration and mutual accountability.

The Association Wetlands International enables governmental and nongovernmental membership to the organization and adopts the long-term strategy of the organization, which is guiding the activities of all offices in the network. The Foundation Wetlands International has the specific to guide and support the office network. Its executive management and the supervision thereof are in the hands of the Management Board and the Supervisory Council, respectively. Members of the Supervisory Council of the Foundation are also members of the Board of the Association.

How Wetlands Can Be Solutions

Securing Water Supplies

As a result of bringing knowledge together at a landscape or basin scale on the values of wetlands such as in supporting water regulation and purification, Wetlands International have been successful in stimulating investments in wetland conservation and restoration, for example to support the sustainability of drinking water and sanitation schemes. In some cases, such as in India and Mali, Wetlands International's evidence has led to reorienting major water infrastructure developments and operations. Wetlands International demonstrate tools and approaches to map wetland values and to connect this with integrated water resource management, working closely with a number of national and local governments. At the global level, Wetlands International are joining forces with major businesses and industry to develop business cases for investing in wetlands as natural water infrastructure to reduce water risks for the long-term.

Reducing Natural Disasters

In working with governments, communities, and private sector partners, Wetlands International place emphasis on reviving wetland ecosystems which grow and adapt to changing environments, rather than physical structures, which cannot. Wetlands International enables local communities to identify the root causes of water-related hazards and to restore and manage wetlands such as along vulnerable coasts, deltas, or floodplains, so as to increase their resilience to severe events. Wetlands International's evidence-base helps to demonstrate to governments that sustaining and restoring wetlands is a cost-effective strategy for disaster risk reduction and climate change adaptation, with strong benefits for poverty reduction and biodiversity conservation.

Meeting the Growing Demand for Food

In the major wetlands where we work, Wetlands International help to integrate wetland values as part of agricultural and fishery systems, including through applying local knowledge. For example in Mali, Wetlands International and its partners are advocating for equitable water sharing and better water governance to sustain fisheries through a participatory approach whereby traditional fisheries management is integrated in river basin-wide food security and environmental governance. In South America, Wetlands International are bringing information to the Roundtable for Sustainable Soy on the trends and impacts of soy cultivation on wetlands and water quality and are working with producers to minimize and mitigate wetland-related threats. Here Wetlands International has also promoted agricultural practices that enhance biodiversity in rice fields. In Indonesia, we work with smallholders to integrate mangroves into shrimp pond design and management, demonstrating how wetland rehabilitation can contribute to livelihood development and food security.

Reducing Greenhouse Gas Emissions

Due to Wetlands International's campaigns, the vital role of wetlands and especially peatlands in storing carbon (more than double that in all the worlds' forests combined) is now recognized by the UN Convention on Climate Change. However, the drainage and degradation of peatlands is causing major greenhouse gas emissions. Wetlands International's projects in Europe and in tropical peatswamp forests have shown the possibilities to halt and reverse this, turning them again into carbon sinks.

Mission, Vision, and Strategy

In 2015, Wetlands International members adopted its 2015–2025 strategy and committed to reviewing its strategic outlook on a regular basis. Every five years, a consultative process with stakeholders and members in all regions helps to set the organization's long-term goals linked to our mission and vision and to establish specific targets associated with a ten year outlook. This helps ensure that the organization remains relevant to the global agenda and the key threats to wetlands/ the key opportunities for wetland conservation and wetland management to contribute to sustainable development.

Wetlands International believes that people's spiritual, physical, cultural, and economic well-being depends on safeguarding and restoring wetlands worldwide. We consider that far greater prominence needs to be given to the conservation and wise use of wetlands as a contribution to sustainable development. We call for improved governance and concerted action by all sectors of society, from global to local levels, to secure wetlands and their range of values and services for current and future generations. More specifically,

Wetlands International's long-term Vision

is a world where wetlands are treasured and nurtured for their beauty, the life they support, and the resources they provide

Wetlands International's Mission

is to sustain and restore wetlands, their resources, and biodiversity

Wetlands International's Goal

is that wetlands are wisely used and restored for the role they play in:

- Improving human well-being and local livelihoods
- Conserving biodiversity
- Sustaining the water cycle
- Reducing climate change and its impacts

Wetlands International's targets and strategic interventions *focus in key geographic areas and on issues where* wetlands make the most difference for people and nature. Our strategic intent for the period 2015–2025 sets out our strategic interventions according the following five target areas:

1. Conserve and restore wetland habitats, sites, and species
2. Conserve and restore peatlands as an integral part of landscape planning and management
3. Conserve and restore wetlands as natural water stores
4. Conserve and restore floodplain systems as blue lifelines in the desert
5. Conserve and restore wetlands as integral part of resilient coastal landscapes

Partnerships

Wetlands International works in partnership with many different organizations on wetland conservation and natural resource management including NGOs, knowledge institutes, business, and industry and international conventions.

The changing governance landscape means that companies are being held more accountable by their stakeholders and consequently must be more transparent in their use of natural resources and impacts on the environment. Companies collaborate with us to improve their environmental and/or social performance in relation to wetlands. We help them to mitigate risks, including physical, regulatory, reputational, and financial risks, as well as to share perspectives and knowledge for innovative solutions where wetlands have a role to play.

Companies like Shell and engineering and dredging firms like Boskalis and Haskoning collaborate with us to help them reduce their environmental impacts and to combine expertise in developing and implementing nature-based engineering solutions. Through our membership of the Dutch-based Ecoshape consortium, we join private sector companies in promoting the building with nature approach, in developing business cases and creating the enabling environment for implementing projects through engaging governments and communities in southern countries, where we are based.

Wetlands International engage and collaborate with water and ecological knowledge institutes in each region, often playing the role of brokering wetland knowledge and technical advice to communities, governments, and private sector players so as to tackle specific challenges. This way of working that combines technical expertise, practical experience, and policy advocacy allows us to scale up successful wetland solutions to have a global impact.

As well as working with a myriad of environmental NGOs and NGO networks globally and in each region, we are close partners with numerous development and humanitarian NGOs and work with their networks in a number of developing countries. For example:

Partners for Sustainable Development: Under the Ecosystem Alliance, we work in partnership with IUCN Netherlands and Both Ends to work in the field (Senegal, Kenya, DRC, Argentina, Indonesia) and at international policy fora to promote the role of ecosystems for people, to green economical trade chains such as soy and palm oil, and to show the importance of wetlands and other ecosystems for climate regulation.

Partners for Disaster Risk Reduction and Climate Change Adaptation: We are member of Partners for Resilience; a partnership between Red Cross (Netherlands), Red Cross Climate Centre, Care (Netherlands), and Cordaid for community – ecosystems based approaches for disaster risk reduction and climate change adaptation. Under the partnership, we lead projects in Mali, India, and are strongly involved in Kenya, Indonesia, Nicaragua, and Guatemala.

Wildfowl and Wetlands Trust

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Martin Spray

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Abstract
Established in 1946 by Sir Peter Scott, one of the most influential conservationists of his time, the Wildfowl & Wetlands Trust (WWT) has grown to become a leading UK-based wetland conservation organization. WWT works across the UK and around the world with a diverse range of partners to create, manage, and restore wetlands. WWT also engages with and works to inspire people, governments, and businesses to take direct action to save wetlands and their wildlife and provide the tools that enable them to do so.

Keywords
Nongovernmental organization · Conservation · Wetland creation · Restoration · Management

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Background

Established in 1946 by Sir Peter Scott, one of the most influential conservationists of his time, the Wildfowl & Wetlands Trust (WWT) has grown to become a leading UK-based wetland conservation organization.

WWT works across the UK and around the world with a diverse range of partners to create, manage, and restore wetlands. WWT also engages with and work to inspire people, governments, and businesses to take direct action to save wetlands and their wildlife and provide the tools that enable them to do so.

WWT and UK's Wetland Reserves

WWT is managing nine wetland reserves and associated land holdings totaling 2,622 ha across the British Isles. This land is of high ecological value, as recognized by the UK, European, or International designations applied to all or parts of our reserves. Six are Wetlands of International Importance as recognized under the Ramsar Convention.

Case Study: An Example of Enhancing Our Reserves Through Habitat Creation at Welney, Norfolk

WWT Welney covers the lower portion of the Ouse Washes in Eastern England, constructed some 300 years ago to drain fenland for agriculture, and holds national, European, and International designations due to its international importance for breeding and wintering birds. However, the Ouse Washes are listed on the Montreux Record of the Ramsar Convention because they are deteriorating due to the impacts of spring/summer flooding with key species declining via subsequent loss of eggs and chicks.

In addition to working with the relevant authorities to try to minimize the impacts of this unseasonal flooding on the Ouse Washes, WWT has also been working to create new areas of habitat adjacent to the Ouse Washes to provide additional and alternative habitat for breeding and wintering birds.

Since 2008, WWT have been creating new wet grassland habitat on former agricultural land. Initial investigations included topographical survey work to ascertain spot heights at key locations and soil analysis to establish the varying depths of peat and underlying clay across the site. Based on this, the design focused on altering the topography to accentuate the natural contours. Transformation into wet grassland was achieved by digging a system of ditches, channels, and scrapes; inserting a waterproof liner; and seeding with native grasses. The sites have been engineered to allow close control of water levels, and grazing by sheep and cattle has been established to manage the grassland, with an electric fence serving the dual purpose of containing the livestock and deterring terrestrial predators.

So far the project has been extremely successful. In addition to up to 500 wigeon *Anas penelope* recorded daily during the first year, snipe *Gallinago gallinago* and jack snipe *Lymnocryptes minimus* have been recorded along with more than 120 lapwing *Vanellus vanellus*, 200 golden plover *Pluvialis apricaria*, and 200 whooper swans *Cygnus cygnus*. In 2011 black-tailed godwits *Limosa limosa* bred successfully – a species whose breeding areas have become highly restricted in recent years and that are at risk of being lost from the Ouse Washes altogether.

Demonstrating Wetland Management and Multifunctionality

WWT works on a variety of wetlands in the UK and internationally that best demonstrate their values and benefits to wildlife and people.

Wetland Treatment Systems

Over the past 20 years, WWT has built-up extensive experience of constructing and managing treatment wetlands at its centers. As of 2013 there are a total of 18 systems across all nine centers designed to tackle a variety of effluents. WWT design systems to mimic natural wetland habitats including pools, marshes, and reedbeds which can support a rich diversity of wetland plants and macroinvertebrates.

They perform a vital function in providing clean water for the wetlands in our reserves and visitor centers but equally also protect the sensitive wetland habitats that water is discharged into, including Strangford Lough and the Severn Estuary, both designated as Wetlands of International Importance.

WWT also undertakes a program of research and development on these systems to ensure they are effectively performing the functions they were designed for and to explore ways to enhance the systems and maximize the full range of benefits that these systems can provide.

WWT has extended this work to include, for example, the design and establishment of a number of treatment systems in Vientiane, Laos and, in the UK, systems for the treatment of waste water from breweries, schools, equine clinics, landfill sites, and farms. WWT are also investigating phosphate cycling under different management options within wetland treatment systems to increase their capacity for phosphate removal and have undertaken innovative research to investigate the benefits of these systems in tackling disease and pathogens.

Our Work Overseas

While being UK-based, WWT also works internationally. The following provides examples of some of the overseas work conducted by WWT.

Cambodia

Since 2010, WWT has been working on a project to protect two wetland sites, Boeung Prek Lapouv (BPL) and Anlung Pring (also known as Kampong Trach) wetlands in the floodplain of the Mekong River in Cambodia. The wetlands are among the last strongholds for the eastern sarus crane *Grus antigone sharpii* and support numerous other threatened wetland species. The sites are also vitally important to local communities, but are under increasing pressure from unsustainable practices.

The main threats to the sites are the exploitation of wildlife, agricultural encroachment, agrochemicals pollution, inappropriate fishing methods, hydrological changes and consequent vegetation changes, and invasive alien plants. WWT is working with three local Cambodian NGOs and in collaboration with the Cambodia Programme of Birdlife International to develop the skills of local conservation groups to better manage the sites and to work with local communities.

The first major success of the project was achieved in January 2011, when Anlung Pring joined BPL in being officially designated by the Cambodian Government as a special protected area for the cranes under the management of the Forestry Administration (FA), in the Ministry of Agriculture, Forestry and Fisheries (MAFF). Since its designation, we have been working with the FA and other governmental and nongovernmental stakeholders to develop a management plan for the site that will identify the key features of the site and prescribe some key management activities.

Beyond the boundaries of the reserves, WWT are also working with local Cambodian NGOs to improve the agricultural and other practices of local communities so that they are less harmful to the wetlands but still provide sustainable livelihoods. WWT are also working to identify alternative livelihood options that can supplement existing options and replace some of the more damaging ones. Among the options being explored are the development of “crane-friendly rice,” ecotourism opportunities, and associated local handicraft products.

Nepal

The wetlands at Koshi Tappu in eastern Nepal are vitally important, not just for Nepal’s last wild water buffalos *Bubalus arnee* and the tens of thousands of waterbirds the wetlands support but also for the people who live there and depend on wetlands for their livelihoods. This dependence results in pressure on wetlands in and around the reserve, so WWT have helped alleviate these pressures by providing sustainable alternatives that support local livelihoods.

In collaboration with Bird Conservation Nepal, WWT has investigated wetland resource use and identified barriers to people obtaining a sustainable livelihood. WWT found several livelihood options that offer sustainable alternatives to current resource use and provided investment and training to enable the poorest most wetland-dependent people locally to benefit from them.

For example:

- Fish farming in ponds dug from agricultural land in the reserve buffer zone provides a good alternative to the capture of wild fish for local fish-dependent people.
- Weaving mats using *Typha* offers an alternative to fishing as a livelihood, enabling women in particular to generate an income.

WWT has also found two good uses for invasive nonnative plants. These both support local livelihoods and provide an incentive for people to remove them from waterbodies:

- Using water hyacinth *Eichhorniacrassipes* to make compost reduces the need to purchase chemical fertilizers.
- Charcoal made from invasive *Ipomoea* and *Lantana* is made into briquettes. These provide a smokeless fuel which is more efficient and cheaper than firewood and reduces demand for animal manure for fuel (thus increasing the amount available as fertilizer on farmland).

A cost-benefit analysis of the three livelihood alternatives that provide an income (fish farming, mat weaving, and briquette making) showed that these sustainable options give excellent returns per unit area of land used compared to conventional crop-growing (multiple cropping of rice, maize, wheat, and fodder).

Work is ongoing to encourage the uptake of these livelihoods and to inform the revision of the site management plan by the Department of National Parks and Wildlife Conservation. In order to support this we have produced a set of wetland management guidelines and are continuing to work with Bird Conservation Nepal to monitor the situation and measure the benefits that uptake of livelihood options bring to both people and biodiversity, and to determine whether further conservation and development work is necessary.

Health in Wetlands

WWT has a long history and international reputation in wildlife health and has contributed to the Ramsar Convention's work on healthy wetlands and healthy people. One of the most important outputs of this work was the crafting, with colleagues from other organizations, of Resolution XI.12 on an ecosystem approach to health. The Resolution, which was adopted by the 163 contracting parties to Convention in the summer of 2012, explicitly recognizes the interconnectivity of health of all life within wetlands and how maintenance of well-functioning wetlands promotes health across these sectors.

As a resource to help wetland managers achieve health in wetlands, the Ramsar Wetland Disease Manual (Cromie et al. 2012), produced by WWT for Ramsar's Scientific and Technical Review Panel, was launched in 2012 at the same time as the

Resolution. The Manual recognizes the importance of the role of wetland managers in the prevention of health problems and represents a practical resource for this key stakeholder group as well as policy makers. It provides an overview and the principles of animal health and disease management, specific and generic practices for preventing and controlling diseases in wetlands, user-friendly factsheets of a wide range of priority animal diseases of wetlands, and case studies. Although focused on wetlands, many of the principles and practices are appropriate for other habitats.

Future Challenges?

Throughout history wetlands have suffered from being been misunderstood and undervalued, leading to more rapid loss and degradation than most other habitat types. Those wetlands that remain today are faced with ever increasing pressures, and their piecemeal degradation and destruction continues. WWT has identified the following particularly significant threats to wetlands:

- Direct destruction of wetlands
- Overexploitation of natural resources
- Pollution
- Invasive species
- Disturbance
- Flow modification
- Disease

These well-known and long-standing challenges are now also being exacerbated by new threats and drivers such as climate change and unsustainable (and increasingly globalized) patterns of consumption of natural resources. A further complicating factor for wetlands is that, due to their position within the landscape, wetlands are often the "receivers" of multiple threats acting in combination and often originating from locations far removed from the wetlands themselves.

Fundamentally, many of these threats have their origin in the lack of understanding and appreciation of the importance and functions of wetland ecosystems. Changing this perception is, and will continue to be, at the heart of WWT's mission.

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World Wetland Network

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Chris Rostron

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Abstract

The World Wetland Network (WWN) acts as a support network for relatively small NGOs engaged in wetland conservation. This may not be their main activity, but broadens involvement of NGOs or CSOs (civil society groups) that recognize the importance of wetlands not only for biodiversity, but also for human and economic well-being. WWN aims to bridge the gap between multilateral environmental agreements and the NGOs that deliver wetland work at ground level. Although the Ramsar International Organization Partners (WWF, IUCN, BirdLife International, Wetlands International, IMWI, and WWT) are already an integral part of the Ramsar structure, they represent a distinct international focus. WWN works to ensure that the interests of smaller NGOs are better integrated, raising their profile as significant deliverers of wetland conservation on the ground.

Keywords

Capacity building · Civil society · Network · Nongovernmental organization

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Introduction

The World Wetland Network (WWN) acts as a support network for relatively small NGOs engaged in wetland conservation. This may not be their main activity, but broadens involvement of NGOs or CSOs (civil society groups) that recognize the importance of wetlands not only for biodiversity, but also for human and economic well-being. WWN aims to bridge the gap between multilateral environmental agreements and the NGOs that deliver wetland work at ground level. Although the Ramsar IOPs (international organization partners), such as WWF, IUCN, BirdLife, Wetlands International, IMWI and WWT, are already an integral part of the Ramsar structure, they represent a distinct international focus. WWN works to ensure that the interests of smaller NGOs are better integrated, raising their profile as significant deliverers of wetland conservation on the ground.

Origins

WWN arose from a perceived need for better organization among NGOs at the Ramsar Conference of the Parties (COP10 in Changwon, Repbulic of Korea, in 2008). NGOs had been active at previous COPs and often achieved input to the COP proceedings, formally during the plenary and work groups, informally behind the scenes, or through actions outside the COP proceedings. However, NGO representatives felt that their sector could work more effectively if it adopted a more structured approach, offering support and coordination for the diverse range of groups that attend the Ramsar COPs, or that work to deliver wetland conservation.

The network is not a formally constituted group, but is run by a group of NGO representatives, the WWN Committee, put forward at the Ramsar COP10, and hosted by the Wildfowl and Wetlands Trust, UK (WWT). WWN had nearly 2000 members as of 2012, many of whom signed up as part of the process of voting for their wetlands during the two rounds of the Globe Awards. Membership is free and open to all CSO groups and individuals that are actively engaged in wetland conservation.

The key aims of the WWN are as follows:

(a) Facilitate and coordinate the involvement of wetland NGOs at every Ramsar COP, particularly through the international NGO meeting, and help influence similar Multi-lateral Environmental Agreements.
(b) Raise the profile of wetland NGOs and their role in the local delivery of international wetland conservation agreements.
(c) Create an international platform to share knowledge and spread messages about the best practices for wetland conservation and wise use.
(d) Take positive action to support good wetland management, and offer support to local NGOs in their work to protect internationally important wetlands.

Actions at the Ramsar COPs

Every three years, the Ramsar Convention holds a Conference of the Parties (COP), with 2012 seeing the 11th COP. At COP10, 2008, in the Republic of Korea, an International Wetland Conference was held, which resulted in the establishment of the WWN. The following COP11 was the first conference that saw WWN playing a really active role. Preparation for the Ramsar COP11 started well in advance, recognizing the need of smaller NGOs to understand how Ramsar COPs function and the opportunities for NGO involvement during the process. A preparatory meeting was held at CENEAM, Segovia, Spain, bringing together representatives of the WWN committee and some of the IOPs, who are well experienced in previous COPs. The meeting involved an initial review of the voting for the Wetland Globes, a detailed review of the draft resolutions that were proposed for discussion at the COP, and planning for NGO engagement at COP11, including the structure of the international NGO meeting to be held at the start of the COP.

Immediately before the start of the COP11, the international NGO conference was convened, with speakers from the Secretariat and NGO representatives that had been involved heavily in the previous COPs. The event brought NGO delegates together to meet and discuss issues, provided guidance on how to engage during the COP, carried out work on identifying which Draft Resolutions should be of highest priority for engagement, and prepared the opening NGO statement for the COP, read out by a Romanian NGO representative to Ramsar delegates during the opening plenary. Those NGOs attending prepared a closing statement which the WWN Chair read out during the closing ceremony. This gave support from civil society, but also reiterated the need to recognize NGO input at both the COP and within Ramsar member countries.

Wetland Globe Awards

The Wetland Globe Awards is an initiative developed by WWN to draw attention to the role of civil society in wetland management, highlight best practice through case studies, draw attention to wetlands under threat, and offer an opportunity for civil society to register its opinion on local wetlands at an international level. The awards are made for the wetland itself, not to any specific organization, emphasizing the concept that wetlands need all sectors (governmental, business, and civil society) to work together to protect them.

Lake Natron received the Blue Globe award at CBD COP10 in 2010, for not allowing development of a soda ash plant, which would have severely compromised the breeding ground of the lesser flamingo. Shortly after, the development was proposed again, but the Globe Award was used as one of the reasons in the campaign against it. The proposal was eventually turned down again.

Future Challenges

WWN continues to have regular meetings of the committee, and subregional representatives are active across the globe to varying degrees. WWN will also continue to run the Wetland Globe Awards and has ambitions to create a civil society monitoring tool for wetlands. The challenge is to secure sustainable funding to allow development of these ideas. WWN is not an independently constituted group, but hosted at WWT, and therefore cannot fund-raise on its own. Consequently, the challenge is to get enough activity and resources to attract funding, and then justify constitution and independence.

94 World Wide Fund for Nature (WWF)

C. Max Finlayson

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Abstract

The World Wide Fund for Nature (WWF) is an international nongovernmental organization that was founded in 1961 and is based in Switzerland. It was formerly called the World Wildlife Fund, which remains the official name in Canada and the United States, and was changed in 1986 to better reflect the scope of its activities. In 2001 the original acronym of WWF became its one, global name. Its logo is the highly recognizable black-and-white panda that has come to be seen as a symbol for the conservation movement as a whole. Ongoing conservation activities include those that support wetlands and their species, as well as specific initiatives such as the Living Planet Index.

Keywords

Conservation · Biodiversity · Nature · Sustainability

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Introduction

The World Wide Fund for Nature (WWF) is an international nongovernmental organization that was founded in 1961 and is based in Gland, Switzerland. It was formerly called the World Wildlife Fund, which remains the official name in Canada and the United States, and was changed in 1986 to better reflect the scope of its activities. In 2001 the original acronym of WWF became its one, global name. Its logo is the highly recognizable black-and-white panda that has since come to be seen as a symbol for the conservation movement as a whole. Information on the organization and its many activities is available through its annual reports (see WWF 2015) and web pages (http://wwf.panda.org), including those for its offices around the world. This information forms the basis of the text that follows.

WWF is now one of the world's largest conservation organizations with over five million supporters. It has invested in excess of US$1 billion in more than 12,000 projects since 1985 and currently has conservation and environmental projects in more than 100 countries. These projects contribute to the general goal of finding a balance between biodiversity and the demands that humans place on natural resources, including biodiversity.

It is an independent foundation registered under Swiss law and is governed by a board of trustees under an international president. More than half of its funding comes from individuals and bequests, with the remainder from governmental sources and corporations. An international secretariat is based in Gland, Switzerland, with the role of leading and coordinating the network of WWF offices around the world through the development of policies and priorities, partnerships, and international campaigns. The network of offices comprises those that raise funds and work autonomously and those that work under the direction of one of the other offices.

All offices undertake conservation work, including practical field projects, scientific research, advising local and national governments on environmental policy, promoting environmental education, and raising awareness of environmental issues. Those that work independently also contribute funding to WWF's global conservation program. All offices contribute to an enormous pool of environmental expertise and knowledge.

History

WWF was formed by a small group of scientists and advertising and public relations experts committed to establishing an international organization to raise funds for conservation. In April 1961, they signed a declaration entitled "We Must Save the World's Wild Life" that became known as the *Morges Manifesto*. Its operations were based in Morges, Switzerland, where IUCN was then based. Offices were established in different countries and National Appeals launched, with two-thirds of the funds they raised being sent to the international secretariat and the remainder retained for their own conservation projects. Its work was based on the best available science and included educational activities as well as wildlife surveys and antipoaching projects. Many of the animals and habitats that featured in the early work of WWF have assumed an iconic status and still feature. This includes the assistance provided to establish the Charles Darwin Foundation Research Station in the Galápagos Islands and in 1998 the Galápagos Marine Reserve. It also included support for the purchase of land in the Guadalquivir Delta marshes in Spain that later became the Coto Doñana National Park – an important site for migratory water birds and nowadays an internationally important wetland (Ramsar site).

As the success of individual projects became known, WWF encouraged the development and implementation of a more comprehensive conservation effort for entire biomes as well as species across their range. This included support for the development of the Ramsar Convention on Wetlands that was signed in 1971. The Convention addressed both the wise use and conservation of wetlands, and through its wise use, provisions (Finlayson et al. 2011) became a forerunner of the sustainable development principles that were being developed by WWF and its partners.

Despite the impressive effort and conservation successes through an expanded global presence, the organization realized that parks and crisis-led conservation efforts were not enough. This led to a greater emphasis on the importance of conservation for people and the integration of conservation with development. This led to the World Conservation Strategy (IUCN et al. 1980) which integrated conservation and the sustainable use of natural resources and explained conservation objectives in terms of the benefits to people. By the 1990s, the link between biodiversity, human activities, and well-being and livelihoods had become more widely accepted and are now explicitly included in WWF's mission and operating procedures that focus attention on ecoregions and with increased engagement with business and new partnerships in support of sustainable development.

WWF continued to support international efforts and was instrumental in establishing the Convention on Biological Diversity (CBD) in 1992. Since then, WWF has continued to support the Convention and lobby for specific outcomes and programs to address the conservation and use of biodiversity. In 2002, the Convention adopted a program of work to significantly reduce the rate of biodiversity loss by 2010. It further successfully lobbied for a program of work on protected areas, including concrete targets and timelines. The outcomes of such work are presented in many fora and publications, including those for protected areas, including for freshwater systems (see Pittock et al. 2014).

Mission

WWF's mission is to stop the degradation of the planet's natural environment and to build a future in which humans live in harmony with nature, by conserving the world's biological diversity, ensuring that the use of renewable natural resources is sustainable, and promoting the reduction of pollution and wasteful consumption.

In response to this mission, WWF has directed its work toward reducing the impact of people on the biological diversity of deserts, forests, freshwater, marine, and mountain ecosystems, as well as in specific places. The latter have been identified on the basis of the best available scientific evidence as important for irreplaceable and threatened biodiversity or representing an opportunity to conserve a representative and large example of a specific ecosystem. Over time, the focus has moved away from localized efforts to conserve single species and individual habitats to an ambitious and strategic approach for global conservation and sustainable development.

Ongoing Activities

WWF has from its founding worked with partners, including UN organizations, development agencies, and other nongovernmental organizations, as well as business and industrial enterprises. The latter have become increasingly important in some of WWF's conservation projects, although at times this courts controversy. By working with private businesses, WWF is seeking to change behavior and achieve conservation results that would not be possible otherwise.

Given restraints on resources and the critical nature of many conservation issues, WWF has focused its considerable efforts on a number of Global Initiatives. These are seen as visionary and large-scale efforts with the potential for broad positive impacts across species and ecoregions and form the centerpieces of a strategic conservation plan. For WWF this was based upon an analysis of the global distribution of biodiversity resulting in a map now known as the Global 200 (Olson and Dinerstein 2002). The Global 200 began with WWF asking the question: "if we wish to conserve biodiversity, where should we be investing our precious conservation funds?" It was an exercise in prioritization, recognizing that if the available resources were spread too thinly they could not achieve the desired result; then how should they be focused?

WWF's focus is very much on the Earth's most special places with the expectation that it is possible to conserve most of life on Earth by protecting the most exceptional ecosystems and habitats. This includes places that are particularly rich in biodiversity, those that support unique animals and plants, and those that are like no other. Priority places include deserts, forests, freshwater, marine, and mountains. Endangered species are also addressed, including flagship and footprint-impacted species. Flagship species are iconic animals that provide a focus for raising awareness and stimulating action and funding for broader conservation efforts. Footprint-impacted species are those whose populations are primarily threatened because of unsustainable hunting, logging or fishing. It is expected that by strategically focusing efforts on these species, there will be positive outcomes for the many other species that share their habitats and/or are vulnerable to the same threats. They have also chosen five priority areas to reduce humanity's ecological footprint – the amount of land and natural resources needed to supply our food, water, fiber, and timber and to absorb our CO_2 emissions. These are carbon, energy, and climate (energy use, impact of forest loss, and the need for a new global policy on climate change), sustainable cities (transition of cities for sustainable development), farming (food, fiber, grazing, aquaculture, and biofuels), fishing (overfishing, illegal and unregulated fishing, bycatch, and poor management and procurement), forestry (timber, paper, pulp, and fuel wood), and water (dams, irrigation, and drinking supplies).

In among the many projects that have been successful, the implementation of the Living Planet Report that was first published in 1998 has resulted in the production of an important publication with information on the conservation status of vertebrate animals and an estimate of the human footprint. This information is produced as the Living Planet Index and the Ecological Footprint (WWF 2014). The former shows that over the period 1970–2010 approximately 50% of vertebrate population for which data were obtained are declining, while at the same time the ecological footprint of humans has more than doubled. The freshwater index shows the greatest decline of any of the biome-based indices with an average decline of 76% in the monitored populations between 1970 and 2010. The ecological footprint shows that 1.5 Earths are now required to meet the demands humanity makes on nature each year.

In response to these trends, WWF continues to provide a lot of support for the aims and goals of the Ramsar Convention on Wetlands. This includes through the formal role of an International Organization Partner and participation in the governance and technical bodies of the Convention. It has continued its efforts to support countries to join the Convention and to nominate and manage wetlands of international importance. In this manner, it is both building on its history in helping to establish the Convention and extending its partnerships and approaches to reduce the degradation of the planet's natural environment and to build a future in which humans live in harmony with nature.

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Section IX

Wetland Law and Policy

Mark Everard

Wetland Law and Policy: Overview

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C. Max Finlayson and Royal C. Gardner

Contents

Abstract

The term "wetland law and policy" refers to the legally related rules developed by governments that pertain to activities that affect wetlands. When used in a general sense, wetland law and policy encompasses a broad range of instruments, including "*. . .legislation, such as statutes, acts, decrees, and ordinances; regulations and other rules promulgated by agencies that have the force of law; and policies, which depending on the jurisdiction may also have the force of the law or may merely provide principles or rules that guide a decision-making process.*" It can also include judicial decisions that apply or interpret the legislation, regulations, and policies. Wetland law and policy may govern activities that have the potential to harm wetlands as well as activities that may benefit wetlands and the ecosystem services they provide.

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Wetland law · Wetland policy · Legislation · Policies

Introduction

The term "wetland law and policy" refers to the legally related rules developed by governments that pertain to activities that affect wetlands. When used in a general sense, wetland law and policy encompasses a broad range of instruments, including "*. . .legislation, such as statutes, acts, decrees, and ordinances; regulations and other rules promulgated by agencies that have the force of law; and policies, which depending on the jurisdiction may also have the force of the law or may merely provide principles or rules that guide a decision-making process*" (Gardner et al. 2012). It can also include judicial decisions that apply or interpret the legislation, regulations, and policies. Wetland law and policy may govern activities that have the potential to harm wetlands as well as activities that may benefit wetlands and the ecosystem services they provide.

Law, Policy, and Wetlands

The United States experience with its legal definition of the term "wetland" provides an example of the differences between statutes, regulations, and guidance documents. As Fig. 1 indicates, the United States Congress (the national legislature) enacted the Clean Water Act, a statute that regulates activities in "waters of the United States." The agencies charged with implementing the Clean Water Act then issued a regulation defining "water of the United States" to include wetlands. Further, more detailed technical guidance was developed through wetland delineation manuals, which are used to determine the boundaries between a wetland and an upland. The statute and regulations are law in the sense that they are binding and have the "force of law." The delineation manuals, on the other hand, are more akin to policy or guidance documents, which do not have the force of law. Nevertheless, application of such guidance can have legal implications.

While wetland laws and policies can differ significantly from jurisdiction to jurisdiction, they can generally be categorized as regulatory or nonregulatory approaches (Gardner 2003).

- A regulatory approach suggests that government permission is needed before a proposed action that could affect wetlands moves forward. In some cases, violators may be subject to administrative, civil, or even criminal penalties.
- A nonregulatory approach establishes incentives (financial or otherwise) that encourage voluntary actions to conserve or protect wetlands.

Regulatory and nonregulatory wetland laws and policies can take many different forms. Many variations exist in different countries, shaped by their particular governance and legal systems, as well as local customs and practices. The

Statute Clean Water Act (enacted by Congress)	33 U.S.C. §1344	Section404: Corps of Engineers may issue permits for discharge of dredged or fill material into the "navigable waters"
	33 U.S.C. §1362	Section502: The term "navigable waters" is defined as "the waters of the United States, including the territorial seas"
	33 C.F.R. §328.3(a)(3)	Definition of "waters of the United States" includes"wetlands... the use, degradation or destruction of which could affect interstate or foreign commerce"
Regulations Corps of Engineers Regulatory Program (promulgated through notice-and-commen trule making and codified in the Code of Federal Regulations)	33 C.F.R. §328.3(a)(7)	Definition of "waters of the United States" includes"[w]etlands adjacent to [other] waters [of the United States]"
	33 C.F.R. §328.3(b)	"Wetlands" is defined as "those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support... a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas."
Guidance Corps of Engineers Regulatory Program (may not be subjected to notice-and-comment rule making and is not codified in the Code of Federal Regulations)	Technical Report Y-87-1, U.S. Army Corps of Engineers Waterways Experiment Station	Wetland Delineation Manual detailing indicators for wetland hydrology, hydrophytic vegetation, and hydric soils
	Regional Supplements	Regional supplements issued (or in process of being issued) for: Alaska; Arid West; Atlantic and Gulf Coast; Great Plains; Western Mountains; Mid-West; Caribbean Islands; and Northcentral and Northeast

Fig. 1 The interplay of statutes, regulations, and technical guidance in the United States (From *Lawyers, Swamps, and Money*, by Royal C. Gardner. Copyright © 2011 Royal C. Gardner. Reproduced by permission of Island Press, Washington, D.C.)

importance of the latter should not be underestimated, whether dealing with traditional access or property rights of local or indigenous communities or with more contemporary commercial enterprises.

Regulatory Approaches in Application of Wetland Law and Policy

In its simplest form, a regulatory approach involves a prohibition and a permit system. An activity, such as filling in a wetland, is prohibited unless the appropriate government agency or official grants permission for the activity to proceed. Often, an environmental impact assessment or study is required prior to the permit decision.

Ideally, an environmental impact assessment promotes informed decision-making. Such studies will typically consider the environmental impacts of the proposed activity, any feasible alternatives to the proposed activity, and the environmental impacts of those alternative courses of action. A "no action" or "status quo" (also known in the UK and elsewhere as "do nothing") option is usually included in the assessment, often serving as a counterfactual. In the wetland permit context, the "no action" alternative would be the denial of a permit, in which case the proposed activity could not legally proceed.

The use of environmental impact assessments became prevalent in the United States after the enactment of the National Environmental Policy Act (NEPA) in 1970. NEPA is a broad "stop and think" statute. It requires an environmental study prior to federal agencies taking certain actions. NEPA applies both to government projects that a federal agency itself performs (e.g., a dam construction project) and to private projects that require a federal permit (e.g., the construction of a housing project on privately owned wetlands). The use of environmental assessments is now a common tool throughout the world (Wood 2003) and a recommended practice by the Ramsar Convention on Wetlands that also encourages the adoption of Strategic Environmental Assessment to ensure a broader consideration of the multiplicity of activities that can impact adversely on wetlands (Ramsar Convention Conference of the Parties 2008).

It is important to note that laws and policies that require environmental impact assessments are frequently *process-focused*. In other words, the legal instrument dictates that a particular process must be followed to encourage informed decision-making. The resulting environmental study, while educating decision-makers about impacts and alternatives, does not necessarily dictate a result. A governmental agency or official may still choose an alternative that is harmful to the environment and wetlands. However, it needs to be a decision made with the awareness of the environmental and wider consequent socio-economic consequences. It is also important to note that although environmental impact assessment may be required by law, the recommendations and conclusions in these studies often have no legal effect by themselves. To be legally binding, the recommendations and conclusions should be incorporated into the terms and conditions of a permit authorizing an activity or project and potentially subject to compliance measures.

The requirement for permits for activities affecting wetlands may flow from a wetland-specific law or policy. For example, Uganda has a specific National Wetland Policy that encourages the avoidance of wetland impacts, stating that there will be "*. . .no drainage of wetlands unless more important environmental management requirements supersede*" and "*. . .only those uses that have been proved to be nondestructive to wetlands and their surroundings will be allowed and/or encouraged.*" In other cases, permit requirements for wetland impacts are the result of broader water-related laws (such as the Clean Water Act in the United States or the Water Framework Directive in the European Union). Also common are permit requirements of general environment or conservation and biodiversity-related laws and policies that encompass wetlands, as well as those that are targeted

Table 1 The legal and policy framework for the alternatives analysis (From *Lawyers, Swamps, and Money*, by Royal C. Gardner. Copyright © 2011 Royal C. Gardner. Reproduced by permission of Island Press, Washington, D.C.)

Statute (enacted by Congress)	Clean Water Act Section 404(b)(1) The Corps will make permit decisions by applying guidelines developed by the EPA	
	⇓	
Regulations (promulgated by EPA through notice-and-comment process)	Section 404(b)(1) Guidelines for Specifications of Disposal Sites for Dredged or Fill Material 40 C.F.R. Part 230	
	⇓	⇓
Guidance (issued by EPA and Corps without public notice and comment)	1993 Memorandum to the Field: Appropriate Level of Analysis Required for Evaluating Compliance with the Section 404 (b)(1) Guideline Alternatives Analysis	1995 Memorandum to the Field: Individual Permit Flexibility for Small Landowners

mainly at conserving resources such as fisheries or water. The importance of the latter should not be underestimated, as many wetlands are under private ownership and used by individuals or communities for commercial or subsistence purposes.

Many permit schemes call for regulatory agencies to use some type of avoid-mitigate-compensate framework. Under such a framework, a wetland law or policy may express a preference that adverse wetland impacts be avoided to the extent feasible. Table 1 provides an example of the sources of the “avoidance” step or alternative analysis in the United States, ranging from the general statutory authority to the more detailed regulations and policy documents.

If wetland impacts cannot be avoided, the impacts need to be mitigated or minimized. Any remaining impacts then should be compensated for, that is, offset by wetland restoration, enhancement, creation, or preservation projects. The importance of wetland restoration has been recognized and is now widely practiced in many countries whether through regulatory mechanisms associated with permits and compliance arrangements or through community effort, such as the considerable involvement of nongovernmental and community-based organizations in wetland restoration in northern America and parts of Europe.

Offset mechanisms in a permit context can take several different forms. The simplest approach is where the permittee itself does the offset project or hires an environmental consultant or engineer to carry out the project. Many studies have found that such “permittee-responsible” offsets have not achieved the desired ecological results (National Research Council 2001). Accordingly, some countries have turned to “wetland banking” or “wetland mitigation banking” to provide offsets to wetland impacts. As described in Ramsar Resolution XI.9 (2012), wetland banking occurs where “*. . . a site owner generates compensation credits through the restoration, enhancement, creation and/or preservation of wetlands. The amount of credits generated is based on the ecological improvements at the site. Credits are* *then sold to developers to offset adverse wetland impacts to the same type of habitat elsewhere*." Wetland banking requires a well-developed legal system, relying on legal rules related to "*. . . property rights (e.g., conservation easements), enforcement authority, and commercial and financial law (e.g., letters of credit, performance bonds, endowment accounts)*" (Gardner et al., in press). A third offset mechanism is called "fee mitigation" or "in-lieu fee mitigation" where a permittee would pay money into a fund which would then finance offset projects.

Many regulatory schemes contain enforcement provisions. If a developer fills a wetland without obtaining the necessary permit, it could be subject to administrative or judicial fines and penalties and even be ordered to restore the site. In rare cases, violators have been subjected to criminal prosecution (Gardner 2011). Permittees may also be subject to enforcement actions if they do not comply with the conditions of the permit. An enforcement action is typically initiated by a government agency or ministry. In some countries, such as the United States, individuals or NGOs may bring a "citizen suit" and sue an alleged violator directly if the government has failed to do so.

The denial of a wetland permit also has legal consequences and may affect property or use rights. In countries where wetlands can be privately owned, a property owner that has been denied a permit may claim that its property rights have been interfered with to such an extent that the government should provide financial compensation. In such cases, the property owner would file an administrative or judicial action to seek such payments.

Nonregulatory Approaches

In addition to the regulatory approaches outlined above, wetland laws and policies may take a nonregulatory approach. Sometimes these approaches are formally reflected in national policies, such as those enacted by Australia and Canada. These are based on a willingness to support common targets or activities, at times with substantive incentives, such as the provision of funding for management and restoration activities or capacity building. The abovementioned national policies are examples of nonregulatory approaches in response to the recommendations from the Ramsar Convention on Wetlands for Contracting Parties (countries) to develop and implement policies for the wise use of wetlands. The success of such measures is debatable as, for example, only 47% of the 160 Contracting Parties to the Convention had reported that they had taken steps to develop National Wetland Policies and incorporate wetlands into a national strategy for sustainable development (Finlayson 2012).

Finlayson (2012) further reported that fewer than half of the Contracting Parties reported activities in response to many of the goals and strategies contained in the Convention's Strategic Plan. As many of the decisions taken by the Convention are not binding, this may not be a surprising outcome. However, it does beg a question about the extent to which such nonregulatory approaches are effective. Finlayson et al. (2011) also considered whether nonregulatory approaches at an international level could be effective and concluded that "*Initial findings indicate that those countries that report better implementation are also reporting that their* *wetlands are in a relatively better state. In particular, this appears to be the case for countries that have established national policy/legislative frameworks and that are undertaking a wide range of implementation activities both nationally and on-the-ground.*"

Canada, the United States, and the countries in the European Union have many nonregulatory programs that encourage wetland restoration (Gardner 2003). In some cases, farmers are paid to convert agricultural lands back to wetlands. For example, in the United States, the Wetlands Reserve Program, administered by the Department of Agriculture offers farmers to voluntarily protect and restore wetlands. The level of cost-sharing payments depends on the duration of the protective measures. More than 11,000 landowners have participated in the Wetlands Reserve Program, covering more than 930,000 hectares of land (US Department of Agriculture 2011).

Another nonregulatory approach involves tax incentives. In Canada, the province of Ontario has a Conservation Land Tax Incentive Program that encourages the protection of provincially important wetlands. Property enrolled in the program can be exempt from property taxes.

In recent years, there has been wider recognition of the importance of traditional access and customs of indigenous peoples, such as in northern Australia where land rights and traditional ecological knowledge have supported both regulatory and nonregulatory approaches for managing wetlands (Finlayson et al. 1998). The importance of local customs and practices has also been formally recognized through the global Strategic Plan for Biodiversity 2011–2020 and the Aichi Targets (Convention on Biological Diversity Conference of the Parties 2010) which includes a target whereby "*By 2020, the traditional knowledge, innovations and practices of Indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, as respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the Convention with the full and effective participation of Indigenous and local communities, at all relevant levels.*" This is an important target but, like the 2010 Biodiversity target that was not met (Armenteras and Finlayson 2012), is also nonbinding and dependent on nonbinding responses by national governments.

The nonbinding nature of approaches to encourage the effective management of wetlands may allow flexibility and encourage sectors and individuals to collaborate and seek joint solutions. This is particularly important when resources for implementing policies are not available or where [potential] policies may be seen as intrusive, inflexible, or practically unenforceable. The importance of community-based or nongovernmental organizations in supporting the delivery of nonregulatory approaches is widely recognized and, in some instances, has provided the basis for bringing governments and others together to explore and develop joint outcomes. At an international level, the development of the Ramsar Convention is an explicit example whereby the development of an intergovernmental treaty was spearheaded by nongovernmental organizations. Such organizations continue to play an important role in developing policy and responding to recommendations made by governments through the Convention processes.

The Influence of International Law on Domestic Wetland Law and Policy

Although wetland laws and policies are primarily domestic-based (i.e., nationally or locally based), international legal instruments, primarily through conventions and other agreements among countries, can influence these laws and policies. However, in practice, the legal effect of a convention within a particular country varies widely. The extent to which a convention or treaty applies directly in a country depends in part on whether that country subscribes to a monist or dualist approach to international law (Bruch 2006).

A monist approach generally means that "international law is part of the domestic law of the country" (Bruch 2006). This means, in some countries, that an international convention is viewed as controlling or overriding domestic or national law. In other monist countries, an international convention has the same authority as a statute or legislative decree, while in other countries domestic laws trump conventions. Despite these variations within monist countries, an important distinction is when such a country joins a convention that agreement "is directly applicable" if its provisions are sufficiently clear.

In contrast, countries that have a dualist approach to international law consider international law to be separate from domestic law. Consequently, an international legal instrument, such as the Ramsar Convention, does not immediately affect a dualist country's domestic legislation. In order for the convention to apply within that jurisdiction, the dualist country would have to enact implementing legislation, also known as the "act of transformation" (Bruch 2006).

In very broad terms, civil law countries follow the monist approach, while common law countries use the dualist approach. Some countries, such as the United States and New Zealand, follow a mixed approach (Shelton 2011).

Challenges

It is important to note that legal regimes, whether regulatory or voluntary, do not necessarily translate to wetland conservation on the ground. Wetland law and policy is often only effective if there is effective enforcement, which requires appropriate investment in administrative entities empowered to protect wetlands.

It is also important to recognize that wetland laws and policies need to be suited to specific socio-economic contexts. For example, an approach that works in urban areas in the United States may not be appropriate for rural KwaZulu-Natal where different (often informal) governance systems operate.

Finally, a mix of regulatory and nonregulatory approaches is required for effective wetland conservation. Sometimes, a nonregulatory approach is more effective than a regulatory approach, especially if the stakeholders (e.g., farmers in the United States) are particularly resistant to or suspicious of traditional regulatory mechanisms. Laws and policies should be developed as part of a coherent broader policy toolkit serving as "societal levers" (sensu Everard et al. 2014), shaping local decisions and actions, including *inter alia*: top-down statutory regulation and levies, bottom-up initiatives including quality assurance networks or community-based partnerships, formal incentives, common law, voluntary market-based schemes such as "payments for ecosystem services," offsetting, and informal agreements and protocols.

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National Wetland Policies: The Basics

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Royal C. Gardner

Contents

Abstract

A national wetland policy is a published governmental statement that articulates how wetland conservation and management is to be achieved within a particular jurisdiction. The policy may be contained in a legally binding instrument, such as a statute or decree, or it may be contained in a nonlegally binding aspirational document. Ideally, a national wetland policy includes assessable goals, timelines for achievement of those goals, roles and responsibilities of various actors, and budget commitments that provide adequate financial support for implementation.

Keywords

Law · Policy · Ramsar convention

Introduction

A national wetland policy is a published governmental statement that articulates how wetland conservation and management is to be achieved within a particular jurisdiction. The policy may be contained in a legally binding instrument, such as a statute or decree,

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or it may be contained in a non-legally binding aspirational document. Ideally, a national wetland policy includes assessable goals, timelines for achievement of those goals, roles and responsibilities of various actors, and budget commitments that provide adequate financial support for implementation (Ramsar Convention Secretariat 2010).

One driver for national wetland policies is the Ramsar Convention. Countries that join the Convention (known as Contracting Parties) undertake, inter alia, a commitment for the wise use of wetlands within their territories. A national wetland policy is considered a key feature for satisfying a Contracting Party's wise use obligation. Indeed, the annex to Ramsar Recommendation 4.10 (1990) stated that "[i]*t is desirable in the long term that all Contracting Parties should have comprehensive national wetland policies*, [which] *should as far as possible address all problems and activities related to wetlands within a national context.*"

The Ramsar Convention's Strategic Plan for 2009–2015 further emphasized the importance of national wetland policies, calling for them "*. . .or equivalent instruments* [to be] *fully in place alongside and integrated with other strategic and planning processes by all Parties, including poverty eradication strategies, water resources management and water efficiency plans, coastal and marine resource management plans, national forest programmes, national strategies for sustainable development, and national policies or measures on agriculture*" (Ramsar Convention Conference of the Parties 2012).

Ramsar Resolution VII.6 (1999) offers guidance on developing and implementing national wetland policies. The policy-making process itself can be a mechanism for building consensus through the participation of stakeholders. Indeed, "*. . .the process used to develop the Policy is its greatest source of strength, particularly when dealing with broad issues and multiple stakeholder interests*" (Ramsar Convention Secretariat 2010).

Ramsar Contracting Parties must report each triennium the status of national wetland policies within their jurisdictions. The table below provides data on the extent to which such policies have been adopted, are in preparation, or are being considered.

	1993 Kushiro COP5	1996 Brisbane COP6	1999 San José COP7	2002 Valencia COP8	2005 Kampala COP9	2008 Changwon COP10	2012 Bucharest COP11
(a) Adopted	6%	7%	11%	32%	40%	44%	49%
(b) In preparation	12%	9%	5%	6%	28%	22%	17%
(c) Development, under consideration or proposed	12%	14%	23%	21%	9%	11%	11%
(d) No action yet reported	71%	71%	61%	41%	23%	23%	23%
Number of national reports tabled	**51 of 76**	**92 of 92**	**107 of 114**	**120 of 133**	**88 of 146**	**139 of 158**	**142 of 159**

Status of National Wetland Policies (Adapted from Ramsar Secretariat 2010 and updated from 2012 Ramsar Convention National Reports)

While there has been an increase in the adoption of national wetland policies, still only about half of Ramsar Contracting Parties report that they have done so. It is also important to note that wetland losses continue even when a national wetland policy is in place (Durigon et al. 2011).

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Marcela Bonells

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Abstract

The Ramsar Convention on Wetlands, formally known as the "Convention on Wetlands of International Importance especially as Waterfowl Habitat," is an intergovernmental treaty for the "wiseuse" and conservation of wetlands, signed in Ramsar, Iran in 1971. It encourages its Contracting Parties to adopt National Wetland Policies as tools to achieve the wise use of wetlands and to implement the Convention at the national level. The force of law, content, scope and format of National Wetland Policies will vary from country to country depending on their government, legal systems and national context. This entry provides an overview about National Wetland Policies under the framework of the Ramsar Convention, highlighting the benefits and obstacles associated with adopting such policies, as well as the trends in their development and adoption by Ramsar Contracting Parties and future challenges for their development.

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National Wetland Policies · Ramsar Convention on Wetlands · Wetland Conservation · Wise Use of Wetlands

Introduction

At the heart of the Ramsar Convention, an intergovernmental treaty for the "wise use" and conservation of wetlands, signed in Ramsar, Iran in 1971, is the concept of wise use (Ramsar Convention Secretariat 2008). The concept is implicit in the text of the Convention, which calls its Contracting Parties (or signatory member States) to conduct their planning in a way that promotes the conservation of wetlands in the List of Wetlands of International Importance (or Ramsar Sites List) and, "as far as possible the wise use of" all national wetlands (Ramsar Convention on Wetlands 2013a; Ramsar Convention Secretariat 2010). Wise use is defined as "the maintenance of [wetlands] ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development" (Ramsar Convention on Wetlands 2005; Ramsar Convention Secretariat 2010).

To help implement the wise use of concept, the Convention recommends to its Contracting Parties to develop National Wetland Policies, among other tools (idem). While the precise definition of the term "policy" and its force of law may vary within each Contracting Party, National Wetland Policies are, in a broad sense, instruments that guide the integrated management and wise use of wetlands at the national level (Ramsar Convention on Wetlands 2013b). The structure and scope of National Wetland Policies may vary according to the government and legal systems of each Contracting Party, as well as their needs. However, the overarching goal of the policies, wetland protection and wise use, remains the same.

This entry first discusses the basis for National Wetland Policies under the Ramsar Convention, providing a definition of the term. Next, it explains the purpose and importance of such Policies, including the benefits and obstacles related to their adoption. Furthermore, the author discusses trends in the development and adoption of such policies by Ramsar Contracting Parties. Finally, this entry outlines future challenges related to the development of National Wetland Polices.

Background

Basis for National Wetland Policies Under the Ramsar Convention

National Wetland Policies are important instruments to help implement the Convention's wise use concept, implicit in Article 3.1 of the Convention's text (Ramsar Convention Secretariat 2010). To assist parties with the development of National Wetland Policies, the Convention has developed *Guidelines for developing and implementing National Wetland Policies*, through a series of Recommendations and Resolutions (*see* Recommendations: IV.10 (1990) and VI.9 (1996) and Resolutions: V.6 (1993) and VII. 6 (1999)) (idem). These decisions and recommendations have been compiled in volume two of the Ramsar handbooks for the wise use of wetlands series: *National Wetland Policies: Developing and implementing National Wetland Policies*.

Moreover, Strategy 1.3 of the Convention's Strategic Plan (2009–2015) calls Contracting Parties to develop by 2015, with the assistance of the Secretariat, National Wetland Policies, simultaneously with, and integrate them into, existing planning and strategic processes, such as water resources management strategies, coastal management plans and poverty eradication strategies, among others (Ramsar Convention Secretariat 2008). In this light, the Convention's National Reports, submitted by Contracting Parties before each Meeting of Conference of the Parties (COP), serve as a benchmark to assess progress towards the attainment of this objective, indicating whether Parties have in place a wetland policy or an equivalent instrument.

It is important to note that there is no single formula or process for developing, adopting, or updating National Wetland Policies. Instead, each country will develop, adopt, or update them according to their national contexts.

What are National Wetland Policies? Definition and Overview

National Wetland Policies are tools (usually in the form of documents) to ensure the wise use and integrated management of wetlands at the national level (Ramsar Convention Secretariat 2013; Ramsar Convention Secretariat 2010). Contracting Parties may have different definitions of the term "policy," and some may use the term interchangeably with "plan" or "strategy" (Ramsar Convention Secretariat 2010). For clarity and consistency, the Convention's handbook refers to "policy" in general terms as "a clearly published statement by a national or sub-national government, often with measurable goals, timelines and commitments plus budgets for action" (idem). In contrast, plans or strategies may, in some cases, provide a more general vision of a government's goals with specific deadlines and budget lines to be defined in the future (idem). It is worth nothing, however, that plans and strategies are equally important to policies, and there is no suggestion that a particular term is more suitable than the others (idem).

National Wetland Policies then will act as blueprints, expressing a government's vision, goals, objectives, actions, and commitments, as well as guiding decisions regarding wetlands and clarifying roles and responsibilities (Ramsar Convention Secretariat 2010). Policies may also provide an overview about the status of wetlands, as well as programs or strategies undertaken (idem). Policies are often broad in scope covering key policy areas and issues affecting wetlands in different sectors and jurisdictions and should seek to involve a broad spectrum of stakeholders (idem).

The scope, level of government, process, and legal instruments for developing and adopting National Wetland Policies vary from country to country. For instance, in countries with nonfederal governments, such as Colombia and Uganda, National Wetland Policies may be nation-wide in scope, while in countries with federal government systems, such as the United States and Canada, such policies may operate only at the federal level with additional policies developed at the subnational level (Ramsar Convention on Wetlands 2012). In some countries, National Wetland Policies may be standalone documents, while in others, such as the United States, the United Kingdom and New Zealand, they may be a collection of statements, documents, and programs (idem).

Similarly, in some countries, such as Pakistan, Thailand, and Japan, a cabinet may adopt Policies or equivalent instruments, while in others, such as Chile and Belarus, ministers may adopt them (idem; Ramsar Convention Secretariat 2010). Additionally, some countries may adopt policies or equivalent instruments formally through decrees or regulations, such as Guatemala and India, while others, such as the United Kingdom, may do so through statements or white papers (Ramsar Convention on Wetlands 2012). Likewise, depending on the legal and government systems of a country, a National Wetland Policy may be legally binding (or have the same weight as other laws, decrees or regulations), while in others it may be a guiding document to government agencies or ministries (Ramsar Convention on Wetlands 2012; Ramsar Convention Secretariat 2010).

In some countries, National Wetland Policies are standalone documents specifically dealing with wetland management and conservation, while in others, such as Bangladesh, Brazil, Denmark, Egypt, El Salvador, Italy, Japan, Lesotho, and Mongolia, among others, policies may be contained in other national or regional strategies; laws or policies, such as National Biodiversity Strategies and Action Plans, under the Convention of Biological Diversity; the European Union's Water Framework Directive; or the *Central American Policy on the Conservation and Wise Use of Wetlands*, under the Central American Commission for Environment and Development of the Central American Integration System (Ramsar Convention on Wetlands 2012).

It is worth mentioning that while many Contracting Parties have adopted National Wetland Policies after acceding to the Convention (or becoming Parties), some countries, such as Thailand, adopted standalone wetland policies before joining the Convention (Ramsar Convention on Wetlands 2012; Ramsar Convention Secretariat 2010). Therefore, although the means for adopting National Wetland Policies varies, as well as their scope and form, they all have the common goal of ensuring the wise use and conservation of wetlands, which in turns helps deliver the Convention at the national level.

Need for National Wetland Policies: Benefits and Obstacles

Wetlands play vital ecological functions in the water and chemical cycles (Russi et al. 2013; Barbier et al. 1997). They provide a wealth of ecosystem services to society, including regulating services (e.g., water quality and flow regulation, coastal erosion protection, flood control), provisioning services (e.g., food, medicines, genetic and natural resources, fresh water), supporting services ("those necessary for the production of all other ecosystem services"), and cultural services (e.g., bird watching, eco-tourism, cultural heritage) (Russi et al. 2013; Millennium Ecosystem Assessment (MA) 2005). However, despite being "amongst the most productive ecosystems on earth," wetlands have and continue to be progressively degraded and destroyed for land conversion, urban development and agriculture, among other activities (Barbier et al. 1997; MA 2005; Russi et al. 2013).

Adopting a standalone National Wetland Policy can be a tool to help stem the loss and degradation of wetlands by, for example, helping raise awareness about the value and functions of wetlands, which require different management approaches (Ramsar Convention Secretariat 2010). Since wetland issues are often confined to a single government agency or ministry and seldom covered in other policies or strategies, such as water resources management or agriculture, an individual policy can help mainstream wetland issues into other sectors and develop targeted actions to deal with such issues (idem).

Additionally, National Wetland Policies may be adopted to fulfill obligations under the Ramsar Convention and foster its implementation at the national level. They may also be adopted in response to national legislation or to address wetland issues, before a country becomes a Contracting Party to the Convention (Ramsar Convention on Wetlands 2012).

Therefore, given the vital services that wetlands provide and their continued loss, National Wetland Policies can be a key instrument for their wise use and conservation.

Benefits and Obstacles

Development of a National Wetland Policy can offer multiple benefits to Contracting Parties and countries that are not yet Parties to the Convention, including:

- Setting accountability mechanisms for government agencies and ministries' actions regarding wetland management and conservation
- Enhancing coordination and communication between government agencies and ministries that have jurisdiction over wetlands or whose actions impact wetlands
- Fostering partnerships with other sectors and actors that depend on wetlands or whose actions affect wetlands
- Mainstreaming wetland issues into other sectors and involving a wide range of stakeholders
- Promoting implementation of new and improved incentives (economic and non-economic) and discouraging perverse incentives and disincentives that contribute to wetland loss and degradation
- Raising awareness about the economic benefits that landowners and wetland managers can derive from the wise use of wetlands and how these benefits can contribute to the long-term management of wetlands

- Establishing Communication, Education, Participation and Awareness programs to engage local communities and stakeholders in wetlands management and decision-making
- Helping set the priorities and tools to improve awareness of wetland resources
- Establishing information priorities and strategies to obtain the information necessary for effective wetland management (Ramsar Convention on Wetlands 2012; Ramsar Convention Secretariat 2010)

Notwithstanding the benefits that establishing National Wetland Policies can provide, there are obstacles to their adoption and implementation, which include (Ramsar Convention Secretariat 2010; Ramsar Convention on Wetlands 2012; Ramsar Convention Secretariat 2012a, b, c, d, e, f):

- Lack of financial and human resources to develop, adopt, and implement a wetland policy
- Lack of communication and coordination between government agencies and ministries with jurisdiction over wetlands or whose actions affect wetlands
- Lack of inclusion of wetland issues into other sectoral policies or strategies and lack of intersectoral cooperation
- Limited knowledge and understanding of wetland issues among policy and decision-makers
- Absence of a legal framework and a clear mandate regarding an agency's responsibility towards wetlands
- Perverse incentives or subsidies contributing to wetland loss and degradation
- Slow administrative processes to adopt effective policies
- Lack of political interest to address wetland issues

Drafting and Organizing National Wetland Policies

While there is no single process for drafting and adopting National Wetland Policies, the Convention's guidelines outline several steps and priorities that should be considered by Parties seeking to develop a policy, including:

- Understanding and prioritizing the issues that need to be addressed by the policy
- Identifying and designating a lead government agency for the drafting process
- Understanding the requirements, issues, and resources needed for passage of the policy
- Ensuring there are sufficient financial and human resources for the drafting process
- Considering time-tables, since the drafting process may involve considerable time
- Identifying other sectors and stakeholders that should be involved in the drafting process
- Determining how and who will write the policy
- Drafting a national issues statement and background paper

- Establishing a cross-sectoral National Ramsar Committee (or equivalent body) to advice and assist with the drafting process (or involving it in the process where it exists)
- Conducting national consultations for the development of the policy
- Conducting National and local wetland workshops (Ramsar Convention Secretariat 2010).

Additionally, although the contents and structure of National Wetland Policies may vary from country to country, they commonly provide a historical overview about actions taken, as well as a foreword for the policy and an overview of a country's wetland status. Policies may also indicate the roles and jurisdiction of government agencies or ministries and outline cross-sectoral and interagency (interministerial) cooperation. Finally, policies frequently define the overarching goal of the policy, guiding principles, general and specific objectives, policy implementation strategies, and actions (idem).

Equally important to developing a National Wetland Policy is ensuring its implementation, through, for example, designating a responsible lead agency/ministry or organization, developing implementation guidelines, determining what resources will be needed, and ensuring they are (or will be) available, prioritizing objectives and setting timetables, as well as developing an implementation plan (idem).

Trends

Analyses of National Reports submitted by Contracting since the Third Meeting of the Conference of the Parties (COP3) (Regina 1987) show an increase in the number of Parties who have a National Wetland Policy, Action Plan, or Strategy (or equivalent instrument) in place (idem). Currently, the Convention has 169 Contracting Parties (Ramsar Convention 2016). An analysis of National Reports submitted to COP11 (147 reports), both before and after the submission deadline (Bucharest 2012), indicates that 50% of Contracting Parties (73) have in place a National Wetland Policy, Strategy, or Action Plan (or equivalent instrument) (Ramsar Convention Secretariat 2012a). Out of that percentage, three Parties indicated having a draft policy in place, pending approval (idem). Additionally, 11% of Parties (16) indicated planning the development of such a Policy, Strategy, or Action Plan, and 16% of Parties (24) stated being in the process of developing it (Ramsar Convention on Wetlands 2012). In contrast, 22% of Parties indicated not having such Policies, Strategies, or Action Plans in place (idem). Therefore, most Parties have a National Wetland Policy, Strategy, Action Plan, or equivalent instrument in place or are in the process of developing or planning one (see also Ramsar Convention Secretariat 2012b, c, d, e, f, g).

Table 1 summarizes the status of National Wetland Policies from COP3 to COP10.

Table 2 summarizes the status of National Wetland Policies at COP11.

Table 1 Status of National Wetland Polices from COP3 to COP11

	1987 Regina COP3	1990 Montreux COP4	1993 Kushiro COP5	1996 Brisbane COP6	1999 San José COP7	2002 Valencia COP8	2005 Kampala COP9	2008 Changwon COP10
Status of National Wetland Policies								
(a) Adopted	0%	0%	6%	7%	11%	32%	40%	44%
(b) In [preparation]	0%	2%	12%	9%	5%	6%	28%	22%
(c) Development Under Consideration or Proposed	n.d.	2%	12%	14%	23%	21%	9%	11%
(d) No Action Yet Reported	100%	96%	71%	71%	61%	41%	23%	23%
Status of National Wetland Strategies/ National Wetland Action Plans								
(a) Adopted	24%	20%	18%	35.38%	44%	*	*	*
(b) In Draft Form	6%	2%	8%	12.13%	11%	*	*	*
(c) Development Under Consideration or Proposed	n.d.	n.d.	10%	8.8%	34%	*	*	*
(d) No Action Yet Reported	71%	78%	65%	37.40%	11%	*	*	*
Number of National Reports Tabled	**17 of 35**	**45 of 60**	**51 of 76**	**92 of 92**	**107 of 114**	**120 of 133**	**88 of 146**	**139 of 158**

Source: Ramsar Convention Secretariat 2010

Note: The data in this table are based on National Reports that were fully completed and submitted in time for the COPs. ["* Comparable data not available from the 2002, 2005, or 2008 National Reports: figures in the National Wetland Policies section of the table include "equivalent instruments" for these years"].

Table 2 Status of National Wetland Policies submitted to COP11 (2012 Rumania)

Status of National Wetland Polices 2012 Rumania COP11		
	Number of Parties	**Percentage of Parties**
(a) In place	73	50%
(b) In preparation	24	22%
(c) Planned	16	11%
(d) No answer	2	1%
Number of National Reports submitted	147	

Source: Ramsar Secretariat, National Reports Database 2012 (Ramsar Convention on Wetlands 2012)

Note: The data in this table are based on the total number of National Reports submitted to COP11

Future Challenges

National Wetland Policies are valuable tools to implement the Convention's wise use concept at the national level, helping raise awareness about wetland values, as well as coordination and management of wetland resources. While many Contracting Parties have adopted, or are in the process of adopting wetland policies, a number of challenges remain preventing effective implementation of such policies or their adoption by Parties who have not yet done so. Some of these challenges include: lack of resources (human and financial), political or administrative constraints, lack of cross-sectoral and inter-governmental cooperation, and lack of a legal framework.

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National Wetland Policy: Australia

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Contents

Abstract

In response to the requirements of the Ramsar Convention on Wetlands to ensure the wise use of wetlands, the Australian government developed a framework policy for the cooperative management of wetlands. This comprised a federal policy and an expectation that the state/territory jurisdictions would develop their own policies. The principal aim of the policy is to ensure that the Government's actions were consistent with those expected under the Convention, in particular the promotion of the wise use principles for wetland management. The policy was developed with and benefitted from consultation with wetland experts and key stakeholders and other government agencies.

Keywords

Wetland policy · Conservation · Ramsar Convention

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Introduction

In response to the requirements of the Ramsar Convention on Wetlands to ensure the wise use of wetlands, the Australian government developed a framework policy for the cooperative management of wetlands (Environment Australia 1997). This comprised a federal policy and an expectation that the state/territory jurisdictions would develop their own policies. The principal aim of the policy is to ensure that the Government's actions were consistent with those expected under the Convention, in particular the promotion of the wise use principles for wetland management. The policy was developed within the framework of the National Strategy for Ecologically Sustainable Development and benefitted from consultation with wetland experts and key stakeholders and other government agencies. It also benefitted from advice based on the Canadian federal policy for wetland conservation (Government of Canada 1991).

Federal Policy

The Australian federal policy contained a goal, objectives, and a number of guiding principles. As stated in the policy document, "*The Goal of the Wetland Policy of the Commonwealth Government of Australia is to conserve, repair and manage wetlands wisely.*" Specific objectives include to:

- "*Conserve Australia's wetlands particularly through the promotion of their ecological, cultural, economic and social values*
- *Manage wetlands in an ecologically sustainable way and within a framework of integrated catchment management*
- *Achieve informed community and private sector participation in the management of wetlands through appropriate mechanisms*
- *Raise community and visitor awareness of the values, benefits and range of types of wetlands*
- *Develop a shared vision between all spheres of Government and promote the application of best practice in relation to wetland management and conservation*
- *Ensure a sound scientific and technological basis for the conservation, repair and ecologically sustainable development of wetlands*
- *Meet Australia's commitments, as a signatory to relevant international treaties, in relation to management of wetlands."*

Collectively, these provided specific direction for Government actions that directly or indirectly affected wetlands. The objectives and principles recognized the national importance of wetlands and supported a consistent approach for wetland management by all federal government organizations with responsibilities for wetlands. It was not backed by legislation and did not directly apply to the actions of the states/territories.

The policy included strategies, with priority actions, covering the following six major areas:

1. Managing wetlands on federal lands and waters
2. Implementing federal government policies and legislation and delivering federal programs
3. Involving the Australian people in wetlands management
4. Working in partnership with state/territory and local governments
5. Ensuring a sound scientific basis for policy and management
6. International actions.

The policy was supported by a number of principles including:

- The precautionary principle
- Coordination and cooperation across government and with the community, including indigenous people
- Empowerment of local landholders as custodians of wetlands
- The wise use of wetlands
- Integrated natural resource and land-use management
- The repair of degraded wetlands
- Ongoing research

An implementation plan was later provided (Commonwealth of Australia 1997).

The policy was not supported by specific legislation, although the Environment Protection and Biodiversity Conservation Act 1999 provided a legal framework for supporting the implementation of the Ramsar Convention, including the adoption of the key principles of wise use and maintaining the ecological character of all wetlands. This Act has been used to support the reporting of adverse change in the ecological character of two Ramsar sites:

- The Macquarie Marshes
- The Coorong and Lake Albert and Lake Alexandrina.

Further legislative support for the Convention has been provided through the Water Act 2007. Pittock et al. (2010) provided an overview of the manner in which Australia has responded to its obligation to make wise use and maintain the ecological character of all wetlands.

The federal wetland policy is currently under review with a National Wetlands Policy Statement being developed. It is expected that a draft of the Statement will be circulated for public/stakeholder comment, although the timeline for doing this is not known.

State/Territory Policies

In response to the federal policy being agreed, the states/territories were expected to also produce wetland policies and provide a complete national coverage of wetlands. This recognized that the state/territory jurisdictions had more direct management responsibility for wetlands than the federal government. Policies were subsequently produced in some of the state/territory jurisdictions.

- New South Wales: The Wetlands Policy promotes the sustainable conservation, management, and wise use of wetlands across the state and the need for all stakeholders to work together to protect wetland ecosystems and their catchments (State of NSW and Department of Environment, Climate Change and Water NSW 2010). It is accessible from http://www.environment.nsw.gov.au/resources/water/10039wetlandspolicy.pdf (accessed 30 August 2016).
- Western Australia: The Wetland Conservation Policy commits the government to identifying, maintaining, and managing the State's wetland resource, including the full range of wetland values, for the long-term benefit of the people of Western Australia (Government of Western Australia 1997). It is accessible from https://www.dpaw.wa.gov.au/images/documents/about/policy/wetlandspolicy_text.pdf (accessed 30 August 2016).
- Queensland: The Strategy for the Conservation and Management of Wetlands (The State of Queensland, Environmental Protection Agency 1999) aims to have wetlands, many of which are of international importance, managed in accordance with the goal, core objectives, and guiding principles set out in the National Strategy for Ecologically Sustainable Development (Commonwealth of Australia 1992), which incorporates economic, social, and environmental considerations. It is accessible from http://wetlandinfo.ehp.qld.gov.au/resources/static/pdf/resources/reports/wetland-strategy.pdf (accessed 30 August 2016).
- South Australia: The Wetlands Strategy for South Australia provides a framework for the sustainable use of wetland ecosystems with the goal to see wetlands recognized and managed as ecological and community assets for the benefit of present and future generations (Minister for Environment and Conservation 2003). It is accessible from http://www.environment.sa.gov.au/managing-natural-resources/wetlands (accessed 30 August 2016).
- Northern Territory: A Strategy Wetlands had the goal of conserving the biodiversity of the wetlands of the Northern Territory (Parks and Wildlife Commission of the Northern Territory 2000).

The legislative base for these policies is largely dependent on environment or conservation acts rather than specific legislation for the policies themselves. The federal environment act can be invoked if issues of national environmental significance arise, including actions that may affect Ramsar-listed wetlands. The outcomes of such interactions can be contentious, as is the effectiveness of the Water Act for supporting the maintenance of the ecological character of wetlands (see Pittock et al. 2010).

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National Wetland Policy: Canada

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Mark Everard

Contents

Abstract

Canada hosts a major portion of the world's wetland resource base, encompassing more than 127 million ha of wetlands which represents up to one-quarter of the world's wetland area. *The Federal Policy on Wetland Conservation* was issued by the Government of Canada in 1991 following the development throughout 1986 and early 1987 of a national statement and fact sheet by the environmental regulator, Environment Canada. At the time of writing, Canada is considering revision of its Federal Wetland Policy.

Keywords

Government of Canada · Environment Canada · Land use · Wise use · Green Plan · Wetland loss

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Introduction

Canada hosts a major portion of the world's wetland resource base, encompassing more than 127 million ha of wetlands which represents up to one-quarter of the world's wetland area. *The Federal Policy on Wetland Conservation* was issued by the Government of Canada (1991), following the development throughout 1986 and early 1987 of a national statement and fact sheet by the environmental regulator, Environment Canada. The contents of this chapter derive substantially from descriptions in *The Federal Policy on Wetland Conservation* (Government of Canada 1991). However, Canada is considering revision of its Federal Wetland Policy at the time of writing.

Evolution of the Federal Policy

The statement and fact sheet developed by Environment Canada summarized management problems in Canada, identifying major obstacles to wetland conservation. From these sources, a Canadian Government-sponsored workshop to develop a *Wetland Conservation Policy* produced a series of recommendations relating to the need for wetland policy. Environment and Natural Resource Ministers from Provinces across Canada received copy of these outputs.

Consequently, the Federal-Provincial Committee on Land Use (FPCLU) in 1987 identified "wetlands management" as a significant land use issue. A Wetlands Subcommittee of the FPCLU reported on a framework for a Wetlands Policy predicated on the FPCLU's own "wise land use" provisions. This policy development activity was also spurred by Canada having been a signatory nation to the Ramsar Convention for some years, hosting the Third Conference of the Contracting Parties at Regina in 1987, and hence bearing an obligation to implement the principles of the Convention.

The Federal Government's intention to act on wetland policy in 1990 formed part of the national Green Plan. Following a series of consultations throughout the summer of 1990, identifying widespread public support for the conservation of Canadian wetlands, *The Federal Policy on Wetland Conservation* was formally adopted in 1991.

Canada's *Federal Policy on Wetland Conservation*

The Federal Policy on Wetland Conservation recognizes wetlands as important for a diversity of beneficial services – from water recharge, purification, and storage through to shoreline protection, flood regulation, habitat for rare species, and soil and water conservation, as a subset of examples – serving a range of socioeconomic functions cumulatively worth billions of dollars.

Notwithstanding recognition of the high importance of wetlands, the *Policy* also recognized the magnitude of wetland loss comprising an estimated 20 million ha (one seventh of Canada's total wetland base) drained or lost since 1800. This further justified the need for a Federal policy, sharing wetland conservation across federal, provincial, and territorial remits, including addressing transboundary resources such as water and wildlife. Policy review of the (then) over 900 Federal policies and programs across Canada was a priority, recognizing that many of these directly or indirectly affected wetlands.

The objective of the Federal Government with respect to wetland conservation is to, "*. . .promote the conservation of Canada's wetlands to sustain their ecological and socio-economic functions, now and in the future.*" Goals supporting these objectives include the Federal Government, in cooperation with the provinces and territories and the Canadian public, striving to achieve:

- *Maintenance* of the functions and values derived from wetlands throughout Canada
- *No net loss of wetland functions* on all federal lands and waters
- *Enhancement and rehabilitation* of wetlands in areas where the continuing loss or degradation of wetlands or their functions have reached critical levels
- *Recognition* of wetland functions in resource planning, management and economic decision-making with regard to all federal programs, policies, and activities
- *Securement* of wetlands of significance to Canadians
- *Recognition of sound, sustainable management practices* in sectors such as forestry and agriculture that make a positive contribution to wetland conservation while also achieving wise use of wetland resources
- *Utilization* of wetlands in a manner that enhances prospects for their sustained and productive use by future generations.

A range of guiding principles form part of the *Policy*. This addresses issues such as the contribution of wetlands and their functions to the health and wellbeing of Canadians, incorporating wetland conservation into the economic decision-making process, taking an integrated systems approach, using evolving science, taking a cooperative approach including with native institutions and representatives, as well as communication and education programs.

Strategies to Implement the *Policy*

The Federal Policy on Wetland Conservation also outlines seven strategies for the sustainable use and management of wetlands, spanning:

1. Developing public awareness
2. Managing wetlands on Federal lands and waters and in other Federal programs
3. Promoting wetland conservation in Federal Protected Areas

4. Enhancing cooperation
5. Conserving wetlands of significance to Canadians
6. Ensuring a sound scientific basis for policy
7. Promoting international actions

The definition of "Wetland" in the Policy varies somewhat from the Ramsar Definition, describing a wetland as, "*…land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment.*" The *Policy* recognizes that wetlands include bogs, fens, marshes, swamps, and shallow waters (usually 2 m deep or less) as defined in *The Canadian Wetland Classification System* published by the National Wetlands Working Group of the Canada Committee on Ecological Land Classification (1987). [This Classification System has subsequently been superseded by the *The Canadian Wetland Classification System, Second Edition* (Warner and Rubec, 1997), classifying a complex range of natural and constructed wetland types also taking account of chemical and hydrological conditions.]

Wetland policy is then continuing to evolve with improvements in science.

State-level Action

Most of the action required under Federal policy occurs at Province level. Consequently, below the federal level, a range of Canadian Provinces have evolved their own bespoke wetland policies.

As one example, the Alberta Government (2013) released the *Alberta Wetland Policy* in 2013 in order to provide "*…the strategic direction and tools required to make informed management decisions in the long-term interest of Albertians.*" The *Alberta Wetland Policy* includes minimization of loss and degradation of wetlands "*…while allowing for continued growth and economic development of the Province.*" The *Alberta Wetland Policy* is focused on four outcomes: (1) protecting wetlands of the highest value; (2) conserving and restoring wetlands and their benefits; (3) avoiding, minimizing, or replacing lost wetland value; and (4) considering regional context.

Conclusions

The Federal Policy on Wetland Conservation in Canada, formally adopted in 1991, has continued to evolve on the basis of its founding principles, objectives and guidance, informed by an improving science base. Management of wetlands entails linking scales from the Federal to the Provincial and the local, also taking account of transboundary issues. As most action occurs at the Provincial level, some Provinces have adopted their own, regionally tailored wetland policies.

At the time of writing, Canada's Federal Wetland Policy is under revision.

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National Wetland Policy: Chile

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Adriana Suárez-Delucchi

Contents

Abstract

Since the 1990s, Chile has experienced an accelerated pace of economic development that has been mainly based on the exploitation of its natural resources. The dependence on raw materials and an accelerated economic growth has increased the threats imposed to biodiversity, ecosystems and the livelihoods of indigenous and local communities that live upon them. This chapter outlines the geographic distribution of wetlands in Chile and the national and regional efforts to protect and manage these ecosystems. Important challenges remain in order to achieve effective wetland conservation, especially in terms of enabling an effective participation of local communities in natural resources management.

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Keywords
Wetland conservation · Wetland policy · Ramsar sites · Biodiversity conservation · Community participation · Natural resources management

Introduction

Since the 1990s, Chile has experienced an accelerated pace of economic development that has mainly been based on the exploitation of its natural resources. Mining activities accounts for 46% of national exports, while agriculture, fisheries, and forestry contribute to 40% (OECD and CEPAL 2005). Chile's dependence on raw materials has increased threats to biodiversity and ecosystems.

However, the return of democracy and democratic institutions in 1990 encouraged the promotion of environmental protection and strengthened Chile's international commitment to work towards sustainable development. In 1971, the Chilean Government became a signatory of the Ramsar Convention, which came into force in 1981.

Although Chile is not rich in terms of overall quantity of species when compared to tropical areas, it is nevertheless important in terms of the genetic variability existing between species and for the presence of an important number of endemic species, which represent a genetic patrimony unique in the world. For example, 85.5% of Chilean flora has its origin in Chile, from which 44.6% are endemic (Manzur and Lasen 2003).

Wetlands in Chile

A survey undertaken in 1999 concluded that Chile encompassed 4.5 million hectares of wetlands, representing 6% of the national territory. These wetlands are highly diverse in nature, largely due to the wide diversity of bioclimates found across the country.

Geographical Distribution of Wetlands in Chile

From the north of the country to as far south as Santiago, drainage basins tend to be arid or semiarid; consequently, wetlands are scarce in this part of the country, occurring only in exceptional situations. The term *bofedal* (wetlands located at more than 2800 m in the High Andes) is mainly used by the *Aymará* people in the Parinacota province, whereas the name *vega* is used by the Atacameños to identify the vegetation associated with wetlands. These indigenous cultures are closely linked to wetlands and have vernacular names for a high percentage of wetland species. They consume wetland resources such as fish and algae, divert water resources and use these areas for grazing (Centro de Ecología Aplicada and CONAMA 2006).

Coastal wetlands occur in the central region of Chile, located on bays and estuaries with both oceanic and continental water influences. These coastal wetlands are rich in biodiversity. Of a total of 63 species of flora located in the area of Coquimbo, 17 are considered native, 19 are endemic, and 5 have been classified as vulnerable. In terms of fauna, out of a total of 173 species, 15 are considered endemic to Chile, 8 are native, 4 are in danger of extinction, 11 are vulnerable, and 1 is considered rare (Centro de Ecología Aplicada and CONAMA 2006).

Towards the south of the country, wetlands are more frequent and they have also concentrated the activities of different cultural groups for centuries. *Hualves*, boggy wetlands dominated by arboreal species, are found in the Araucanía region. *Ñadis* (meaning 'seasonal swamps' in Mapudungun: the Mapuche language) are also present in the Araucanía and in the Lakes region. Flat areas with volcanic soil are also found here, constituting a waterproof horizon which is drenched for a period of 4–8 months a year. Further south, there are *turbas* (peatlands) and *mallines*. The latter are sunken, water-saturated areas with alluvial and aeolic sediments, with characteristics of gleysolic soils (Centro de Ecología Aplicada and CONAMA 2006).

Chilean Wetlands of International Importance

Chile has declared 12 Ramsar sites of international importance. These are located in the High Andes (7), the coast of central Chile (2), the mountainous area of central Chile (1), and at the coasts of the Lakes Region (1) and the Magallanes Region (1). In total, these Ramsar sites represent 205,876 ha. Ten of these sites are managed by public institutions, and two are under private management (CONAF 2010).

Chile participates in a range of regional initiatives to protect migratory species that depend on wetlands, including for example two Memoranda of Understanding (MoUs) signed with Argentina; one of them aims at protecting the *canquén de cabeza colorada* (the ruddy-headed goose, *Chloephaga rubidiceps*) and the other protects the *huemul* (the south Andean deer, *Hippocamelus bisulcus*). Another MoU has been signed with Perú and Bolivia for the conservation of flamingos. The AGCI (Chilean Agency for International Cooperation) project between Chile and Mexico promotes the standardisation of strategies and protocols on awareness-raising about the importance of wetlands in both countries (Informe Nacional Ramsar 2014).

Chile's National Wetlands Strategy

A document outlining Chile's National Wetlands Strategy (NWS) was developed in 2005. The Strategy has enabled the country to work towards a coordinated and efficient protection of priority wetlands in Chile, and to highlight their importance for sustainable development. The NWS complements the priorities promoted by the Chile's National Biodiversity Strategy (CONAMA 2003), which itself serves the vital function of representing a wider framework for conservation in the country. The NWS states that *wetlands are zones where biodiversity concentrates, and they determine the functioning of ecosystems and hence, of human life* (CONAMA 2005).

One of the objectives of the NWS is to *increase knowledge about wetlands*. In order to achieve this objective, Chile's Ministry for the Environment has created a system to classify Chilean wetlands in relation to their functions and structural characteristics, as a transparent basis for defining different ecotypes of wetlands. Each ecotype is associated with particular functions and threats, enabling development of generic management plans for each ecotype (MMA and Centro de Ecología Aplicada 2011).

A National Wetlands Inventory capturing this information is currently being developed. A wetland information system will become available to the general public, containing complete information on the wetlands that have already been surveyed. The General Water Commission (DGA, associated with the Ministry of Infrastructure) manages and shares information related to water quality and seasonal flow rates of the main rivers and lakes the Commission monitors (Informe Nacional Ramsar 2014).

Another objective of the National Wetland Strategy is to raise environmental, economic, social, and cultural awareness about wetlands. The NWS document also refers to the importance of implementing a legal and institutional framework for the conservation and sustainable use of wetlands. The participation of different actors – such as the private sector, indigenous peoples, universities, and NGOs – is specifically referenced, and the need to develop tools for participatory planning and management of priority wetlands is also stressed (CONAMA 2005).

National Wetlands Action Plan

In order to implement the NWS, a National Wetlands Action Plan was created in 2006. Different governmental organisations participated in its creation including the DGA, the Agriculture and Livestock Service (SAG), the National Natural History Museum (MNHN), the National Forestry Corporation (CONAF), and the National Commission for the Environment (CONAMA) which has been replaced by the Ministry for the Environment (MMA). The National Wetlands Action Plan is currently under review.

National Wetlands Committee

A National Wetlands Committee was also formed in 2005, directed and coordinated by the Ministry for the Environment. The Committee is formed by the Ministry of Foreign Affairs, the DGA, National Commission on Irrigation, SAG, CONAF, MNHN, SERNAPESCA (the National Fishery Service), SERNAGEOMIN (National Geological and Mining Service), and DIRECTEMAR (the Commission for Maritime Affairs).

The Committee meets monthly, and also for special activities if required by any of its members. Committee members analyze documents from the COPs of the Ramsar Convention, so that they can agree on a national view on a range of matters including, for example, the listing of new Ramsar sites and implementation of the NWS and its Action Plan.

Regional Conservation and Sustainable Use Under the High Andean Wetlands Strategy

The Ramsar Convention recognizes the importance of High Andean wetlands as strategic ecosystems, promoting international cooperation in order to preserve their valuable biodiversity, their role as water regulators and as vital spaces for a range of local communities in the Andes, including farmers and indigenous peoples (CONAF 2010).

Argentina, Bolivia, Chile, Perú, Ecuador, Venezuela, Colombia, and Costa Rica, with the support of the IOPs (International Organization Partners), the Flamingo Conservation Group, CREHO (Ramsar Regional Centre for the Western Hemisphere), and CONDESAN (Consortium for Sustainable Development of the Andean Ecoregion), developed in 2005 the High Andean Wetlands Strategy.

Challenges

- There is still much to do in terms of implementing effective planning, including Strategic Environmental Assessments, in those places where policies, programs, plans and development projects could impacts on wetlands and on the local and/or indigenous communities surrounding them.
- Limited funds are inhibiting effective wetland conservation, allied with a lack of applied and basic research on wetland ecosystems.
- There is a need for specific planning for each Ramsar Site and a higher level of commitment from the private sector.
- Monitoring and surveillance are especially important. It has been necessary to implement new environmental institutions, especially in the areas of biodiversity conservation and monitoring, for example with the replacement of CONAMA with the new Ministry for the Environment (MMA) (Informe Nacional Ramsar 2014).
- Governmental institutions are currently working on the national inventory on wetlands, which will support implementation of the Ramsar Convention and the National Wetland Strategy. This is considered a fundamental step towards achieving the sustainable management of Chile's wetlands (CONAMA 2005).
- There is a need for greater community participation in the management of wetland resources, through deliberate efforts for cooperation between public authorities and local communities in ways that promote sustainable and just outcomes.

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101 National Wetland Policy: China

Zhang Manyin

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Abstract
The diverse types of wetlands found in China range from those found in cold temperate to tropical zones, from plains to mountainous regions or plateaus, and from coastal to inland areas. The second wetland resource census of China indicates that the total area of Chinese wetlands is 53.6 million ha (excluding 30.06 million ha of paddy fields which are additional to this area), accounting for 5.58% of the national territorial area. Within this total area of wetland, the area of offshore and coastal wetland is 5.80 million ha, the area of river wetland is 10.55 million ha, the area of lake wetland is 21.73 million ha, and the area of constructed wetlands is 6.75 million ha. Significant change has occurred throughout history in Chinese wetland conservation and management policy. In the 1950s, due to the lack of food production, the Chinese government introduced a food production-oriented policy which led a large amount of wetland conversion into farmland. In the east-south region of China, the large-scale activity of reclaiming farmland from lakes resulted in rapid shrinkage of wetland extent while, in the coastal region of China, large amounts of mud flats have been

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converted into salt ponds, aquaculture ponds, and farmland. However, since 1992 when China joined the Ramsar Convention on Wetlands, the Chinese government has started to pay increasing attention to wetland conservation and has established strategic targets and supporting administrative departments and support mechanism, introduced a large number of action plans, planning processes, rules, regulations and laws, and operated national-level wetland conservation and restoration projects. By January 2014, China had 577 wetland nature reserves and 468 wetland parks, among which 46 had been listed as Ramsar wetlands of international importance.

Keywords
China · Conversion · Food production · Conservation · Intertidal · Mountain · Desert

Background

The diverse types of wetlands found in China range from those found in cold temperate to the wetlands of tropical zones, from plains to mountainous regions or plateaus, and from coastal to inland areas. The second wetland resource census of China indicates that the total area of Chinese wetlands is 53.6 million ha (excluding 30.06 million ha of paddy fields which are additional to this area), accounting for 5.58% of the national territorial area. Within this total area of wetland, the area of offshore and coastal wetland is 5.80 million ha, the area of river wetland is 10.55 million ha, the area of lake wetland is 21.73 million area, and the area of constructed wetlands is 6.75 million ha. There are 4220 types of wetland plants that belong to 438 biomes, and 2312 types of vertebrates that include 231 types of wetland birds.

Significant change has occurred throughout history in Chinese wetland conservation and management policy. In the 1950s, due to the lack of food production, the Chinese government introduced a food production-oriented policy which led a large amount of wetland conversion into farmland. A typical example is the Sanjiang plain wetland, the total area of which reduced by 60%. The Sanjiang plain wetland is now the most important grain production base in China. In the east-south region of China, the large-scale activity of reclaiming farmland from lakes resulted in rapid shrinkage of wetland extent while, in the coastal region of China, large amounts of mud flats have been converted into salt ponds, aquaculture ponds, and farmland. However, since 1992 when China joined the Ramsar Convention on Wetlands, the Chinese government has started to pay increasing attention to wetland conservation and has established strategic targets and supporting administrative departments and support mechanism, introduced a large number of action plans, planning processes, rules, regulations and laws, and operated national-level wetland conservation and restoration projects. By January 2014, China had 577 wetland nature reserves and 468 wetland parks, among which 46 had been listed as Ramsar wetlands of international importance.

The Strategic Targets of Chinese Wetland Conservation

Under its general targets of wetland conservation and wise use, the Chinese government gives priority to wetland conservation. In wetland conservation, priority is given to the conservation of natural wetland ecosystems (prevention first), supplemented by demonstration-scale ecological restoration and reconstruction of degraded wetlands, as well as the establishment of demonstration zones of wetland wise use.

As a matter of public interest, the Chinese government promotes policies to reduce the further degradation and loss of natural wetlands. The number of wetlands is expressly stated in the national ecological conservation system. The Chinese government has also strengthened supervision on the utilization of natural wetlands based on the principles of sustainable management of ecological systems and the practice of catchment management. Another important target is the operation of national key wetland protection projects (NWKCPs). Additionally, the Chinese government advocates the importance of public awareness of wetland conservation and the need for greater respect for the values and benefits provided by wetlands, in order to achieve the coadministration of wetland resource between the community and wetland managers. Key stakeholders in this coadministration include government departments, social organizations, private institutions, and international organizations.

In order to achieve the strategic targets of national wetland conservation, the NWKCPs place differential emphasis on different levels of governance: (1) at regional level, NWKCPs particularly support the coadministration aspects of community wetland conservation and wetland resource, including wetland restoration through changing local people's intensity of resource utilization; (2) at national level, NWKCPs particularly support wetland co-administration including the involvement of the provincial government level, including improvement of the capacity of provincial wetland conservation agencies and the establishment of regional integrated conservation mechanisms; and (3) at global level, NWKCPs particularly support the Ramsar Convention on Wetlands and the associated conservation needs of migratory water birds.

The broad objectives of the wetland conservation policy environment are to form a natural wetland conservation system through the protection of biodiversity, control of pollution, and adjustment of land use types, in order to maintain the natural characteristics and essential functions of wetland ecosystems, reversing the historic trend of shrinking wetland area and degrading wetland function. Implementing restoration and improved management of degraded wetlands in key ecological regions is a priority, serving systematically to restore natural wetlands and their ecological functions. So too is putting in place processes for assessing and consenting wise use of natural wetlands, with associated regimes appropriate for wetland monitoring and ecological risk assessment. It is also necessary to strengthen surveillance monitoring of the wetland resource and to improve capacity building for wetland advocacy, education, research, and technology promotion. This also necessitates the establishment of a robust wetland conservation management system, legal system, and research system. The ultimate objective is to improve the level of wetland conservation, management, and wise use in China; to form a complete natural wetland conservation network such that 90% of wetlands can be maintained in good status; to achieve the "virtuous circle" of wetland conservation and wise use; and to maintain and develop the various functions and benefits provided by wetland ecosystems, so enhancing their contribution to sustainable economic and social development.

The Management Agencies and System of Wetland Conservation

In order to fulfill the clause of the Ramsar Convention that "...all Parties have responsibilities and obligations to protect, manage and wisely use the wetland" since China joined the Ramsar Convention on Wetlands, the State Council has established a wetland management system in which State Forestry Bureaus take the lead. In this, they are assisted by related departments such as the Water Conservancy Bureau, Agricultural Departments, the Land Resource Management Bureau, Environmental Protection Departments, the Development and Reform Commission, the Oceanic Administration, and the Shipping Department. The State Forestry Bureau Wetland Conservation Management Centre (The Ramsar Convention Compliance Office of the People's Republic of China) was established in 2005. Subsequently, Forestry Departments in each province founded provincial management offices or centers for wetland conservation, and a proportion of counties (cities), wetland nature reserves, and wetland parks have also founded special wetland management agencies. In 2007, the State Council approved the National Ramsar Convention Compliance Council that was composed by 16 departments.

Currently, a wetland management system that incorporates wetland nature reserves, wetland parks, wetland protection areas, and multifunction zones has taken shape in China.

- A "wetland nature reserve" is an area delimited for special protection and management of the land, water, or ocean as it is representative of a particular natural ecological system; contains a significant concentration of rare and/or endangered wildlife species; or contains some other special natural feature designated under the law. Thus, a wetland nature reserve is the highest level of designation within the wetland management system.
- A "wetland protection area" is a region designated as a means to strengthen wetland resource protection. This is to ensure the health of the wetland ecosystem, to achieve comprehensive benefits for the wetland, and to contribute to overall regional ecosystem structure. Compared with a wetland nature reserve, a wetland protection area is a lower-level management unit, emphasizing its importance to local ecosystem structure.

- A “Wetland Park” is an area of wetland landscape, or a wider landscape of which wetlands are important constituent units, the protection of which demonstrates ecosystem services and functions, serves as an advocacy or educational resource, demonstrates the wise use of wetlands, contains specific cultural and/or aesthetic value, and may also provide a place for scientific research and ecotourism.

Laws and regulations for Wetland Protection

Before China joined the Ramsar Convention on Wetlands, concepts such as “mud flat,” “river,” “lake,” “ditch,” and “reservoir” were scattered in laws, regulations, policies, and plans. After China joined the Ramsar Convention in 1992, the concept of “wetland” was referenced in “the People’s Republic of China Natural Reserve Regulation” in 1994. Subsequently, “wetland” generally appeared in 15 laws and regulations such as the “Marine Environment Protection Law,” the “Forestry Law,” the “Wild Animal Protection Law,” the “Environment Impact Assessment Law,” the “Administration of the Use of Sea Areas,” the “Water Law,” and the “Law on Prevention and Control of Water Pollution”.

Since 1998, the National Forestry Bureau started the legislative work for wetland protection, and provincial governments started their work in formulating regulations for wetland conservation. The publication of “Heilongjiang Province Wetland Conservation Regulation” provided provincial legal safeguard for wetland management for the first time (Li 2000). Up to 2004, a total of 19 provinces (municipalities) had issued and implemented provincial wetland conservation regulations, including Heilongjiang, Gansu, Hunan, Shaanxi, Guangdong, Inner Mongolia, Liaoning, Ningxia, Jilin, Jiangxi, Sichuan, Tibet, Xinjiang, Qinghai, Yunnan, Zhejiang, Shandong, Beijing, and Hebei. Regulations from city and county levels have also appeared progressively. However, despite this plethora of legislation, there is at present no nation law specifically focused on wetland conservation.

Plans and Related Policies of Wetland Conservation in China

Since joining the Ramsar Convention in 1992, China has put in place a wide range of legal instruments and policy directions to protect and restore its wetland resource. Examples include:

- In 1993, The State Council approved the “Action Plan of China Environmental Protection (1991–2000)”, which set the establishment of oceans, wetlands, grasslands, deserts, and natural heritage wetland as important targets, confirming the content of wetland conservation in national plan.

- In 1994, China included wetland conservation and wise use in “21st Century Agenda – China’s population, environment and development white paper in the 21st century” and “China Biodiversity Conservation Action Plan.”
- In 1995, the Ministry of Forestry implemented the first national wetland resource census.
- In 1996, the State Council identified “wetland conservation” as one of the essential areas of natural conservation in “The ‘ninth five-year plan’ and 2010 vision of state environmental protection.”
- In 2000, the National Forestry Bureau and another 17 governmental departments jointly released the “Chinese Wetland Protection Motion Plan.”
- In 2003, the National Forestry Bureau drew up the “The national wetland protection engineering planning (2002–2030)” and implemented the “Grain for Wetland Project.”
- In 2004, the State Council published “the notice of General office of the state council on strengthening the management of the wetland protection.”
- In 2005, the State Council approved “the national wetland conservation project implementation plan (2005–2010),”, investing 3.1 billion Yuan for recurring conservation on important wetlands.
- In 2009, the State Council launched the second national wetland resource census and implemented the pilot of “the wetland ecological benefit compensation.”
- In 2010, the State Council published “the implementation opinions on wetland conservation subsidies in 2010” with an emphasis on “wetland conservation well” in the “On the twelfth five-year plan for national economic and social development program.”
- Between 2010 and 2011, the National Forestry Bureau issued the “National wetland park management approach (trial),” “National wetland park planning guide,” and “National wetland park pilot acceptance method (trial)” successively.
- In 2012, the State Council approved the “Twelfth five-year national wetland conservation project implementation plan” drawn up by the National Forestry Bureau.
- In 2012, the No. 32 regulation published by National Forestry Bureau in the “Wetland protection regulations” confirmed the contents of wetland conservation, planning, monitoring, restoration and wetland park construction, assessment, and management.

Over the past twenty years, the historic trend of wetland shrinkage and degradation has been revered through the concerted efforts of all levels of government and their relevant departments. This has encompassed a wide range of wetland conservation measures, pollution control, and land use adjustment, as well as major restoration projects, including “Grain for lake” and “Grain for wetland,” that have contributed significantly to the conservation and restoration of formerly degraded wetlands.

Challenges of Wetland Conservation

Despite these significant achievements, wetland-specific laws are still absent at national level to support wetland conservation work in China. Aside from the "Nature reserve ordinance," the majority of laws and regulations do not yet emphasize the integrated conservation of wetlands and biodiversity, which creates some difficulties for wetland resource conservation.

The conservation and management of Chinese wetlands requires the involvement of multiple government departments. However, the diversity of wetlands and the complexity of their functions span jurisdictional boundaries and can create conflicts. Where responsibilities remain unidentified, this can seriously restrict the performance of wetland conservation agencies. Wetland conservation in China mostly depends on the strength of government, lacking effective participation pathways for local residents, communities, and non-government organizations (NGOs). This deficit in public participation acts contrary to the efficient resolution of conflicts between environmental and economic benefit.

The conservation and management of wetlands in China also faces a shortfall of financial support necessary for wetland resource investigation, establishment of conservation areas and demonstration areas, and to put in place pollution control, law enforcement, and effective management teams. In the majority of situations, government is the single source of finance, with no other options for diversifying investment. In response to the challenges of wetland conservation and management, China should in future take the following measures:

- Based on the principles of sustainable development, publish a specific law on wetlands that incorporates ecological protection, resource development, and pollution control according to nationally consistent conditions
- Confirm the jurisdiction of different departments and types of wetland ownership
- Complete the coverage of wetlands across the legal system, and manage wetlands according to this more integrated legal framework
- Achieve "one district one law" in some important wetlands

In order to integrate wetland ecosystem protection, it is necessary to:

- Strengthen coordination both within and between departments to enforce the overall protection and efficient use of wetlands
- Reform responsibilities to bring about improved coordination of wetland resource management, confirming the core duties of wetland management agencies as a means to protect the integrity of wetland ecosystems
- Establish mechanisms of "community co-management" and "public participation," build information-sharing platforms for wetlands, and encourage public participation in wetland conservation

- Establish and complete ecocompensation systems for wetlands. This includes integrating wetland conservation into the national economy and social development plan, enforcing financial investment in wetland conservation, building a financial safeguard system that guides and encourages investment by enterprises, the public and NGOs, and promoting the conservation and sustainable development of wetlands; and
- Launch technological innovation to strengthen scientific understanding of wetlands and to better inform wetland conservation and restoration.

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National Wetland Policy: Ghana

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Mark Everard

Contents

Abstract
Wetland ecosystems in Ghana constitute about 10% of the country's total land surface, comprising marine/coastal, inland, and man-made systems. To protect and drive the sustainable use of these resources, Ghana instituted a National Wetland Strategy in 1999. This encyclopedia entry is summarized from the full Strategy document.

Keywords
Ghana · Wetland policy · Wise use

Managing Ghana's Wetlands: A National Wetlands Conservation Strategy

Ghana's national wetland strategy, contained in the Ghanaian Government publication *Managing Ghana's Wetlands: A National Wetlands Conservation Strategy* (Republic of Ghana 1999a), recognizes that, until relatively recently before the strategy was developed, "…wetlands were virtually considered as "waste lands"

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or areas that only served for breeding mosquitoes." This had formerly led to widespread dredging and drainage, wetland conversion for other uses, or the use of wetlands as dumping grounds for refuse. Neither were there effective controls or constraints on harvesting of wetland resources such as fish, reeds, mangroves, and thatch materials. The net consequence was a diminishing wetland resource.

Particularly since the Ramsar Convention came into force in 1971, wetlands have been increasingly recognized by the international community as ecosystems of considerable importance, comparable to forests, rangelands, and marine ecosystems. The Government of Ghana, as a signatory of the Convention since 1988, recognized the high degree of importance of the nation's wetlands as habitat for wildlife; maintenance of water tables; flood management and water purification; for wider socioeconomic benefits such as providing materials for construction and crafts; and for supporting beneficial uses such as fishing, hunting, grazing, crop production, and domestic water supply.

Recognizing this broad range of values, Ghana's National Land Policy in June 1999 (Republic of Ghana 1999b) precluded a range of practices directly damaging the wetland resource including:

- Physical draining of wetland water
- Draining of streams and watercourses feeding the wetlands
- Human settlements and their related infrastructural developments in wetlands
- Disposal of solid waste and effluents in wetlands
- Mining in wetlands

This was backed up by a program of research and public awareness-raising promoting the values of wetlands and the importance of their conservation and sustainable use to sustain a range of human benefits. This led in turn to publication of *Managing Ghana's Wetlands: A National Wetlands Conservation Strategy*, to promote participation of local communities and other stakeholders in the sound management and sustainable utilization of Ghana's wetlands and their resources.

Ghana's National Wetland Strategy categorized many of the nation's natural coastal and inland wetlands, as well as constructed wetland systems, recognizing their breadth of associated functional, productive, natural, and cultural/heritage values to society.

The Strategy also classified and described threats to wetlands including: water loss, loss of run-off control, infilling or other forms of conversion, hydrological disconnection, saltwater intrusion and soil salinization, soil degradation, pollution, overexploitation, deforestation, sediment and nutrient diversion, and disruption of stable climatic conditions.

The Strategy also listed a number of Ghanaian policies relevant to wetland use and conservation. It recognized that traditional and indigenous management practices are of value in protecting wetlands, incorporating as they do customary laws, informal agreements such as abstaining from fishing in certain fish nursery areas, or taboos. Many of these measures determine rights to land and resource use.

The Government also restated its commitment to restoration and protection of wetlands, for ecological and livelihood benefits.

The stated nine guiding principles in the Strategy include:

i. ***Wise use***, with an emphasis on managing wetlands within biological and physical constraints, to ensure that future generations have access to the same resources as the present generation.
ii. ***Interdependence*** between physical, biological, social, cultural, economic, technological, and environmental conditions of wetland ecosystems.
iii. The ***precautionary principle***, by avoiding activities which would affect the integrity of wetlands and subjecting all development activities to an environmental impact assessment process.
iv. The "***polluter pays principle***," applied to all development activities in wetland habitats.
v. ***Local knowledge and traditional management strategies*** play a role in the management of wetlands.
vi. The livelihood of local communities within the catchment area is ***interlinked*** with the ecological integrity of wetlands.
vii. ***Taking a participatory approach*** through involvement of traditional authorities, local communities, and all concerned people and organizations at levels of decision-making in the sustainable management of wetlands.
viii. Provision of ***incentives and disincentives*** as effective means for managing the use of wetlands and wetland resources.
ix. ***International cooperation*** as an essential measure for the conservation and management of shared wetland resources.

The published Aim of the Strategy is ". . .ensuring the wise use of wetlands for the benefit of the country and its people, present and future." This Aim is backed up by five Objectives:

- To promote the participation of local communities, traditional authorities, and other stakeholders in sound management and sustainable utilization of Ghana's wetland resources.
- To maintain the ecological, cultural, recreational, and aesthetic values of wetlands.
- To ensure that national policies, local knowledge, regulations, and activities contribute to the wise use and sound management of Ghana's wetland resources.
- To ensure that national capacity-building and appropriate legal and institutional frameworks are put in place for effective wetland conservation.

- To create awareness among the people of Ghana on the importance of wetlands and solicit their commitment to conservation and wise use.

Strategies for implementation cover a range of time horizons – short-term (1–2 years), medium-term (3–5 years), and long-term (more than 5 years) – and also recognize a number of priority cross-sectoral issues associated with wetland management in Ghana.

Challenges

At the time of publication (1999), the *National Wetlands Conservation Strategy* recognized the transition from recent conception of wetlands as "...'waste lands' or areas that only served for breeding mosquitoes" through to an integrated strategy with valuation, conservation, and wise use of wetlands at its core. This was also connected with Ghana's national vision for 2020, recognizing the scale of transformation.

While progress has been made with wetland conservation and aspects of wise use, full realization of the goals of the Strategy clearly remain long-term aspirations with their associated challenges.

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National Wetland Policy: New Zealand

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Mark Everard

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Abstract

In 1986, New Zealand became the first Contracting Party of the Ramsar Convention to publish a Wetland Management Policy. Interpretation and context have changed since publication of the New Zealand Wetland Management Policy, which the New Zealand Government is considering updating. The Policy recognizes that the wetlands of New Zealand have always been an important part of the nation's environment, from the earliest days when Maori settlers harvested shellfish and fin fish, collected material for weaving and farmed the extensive flat swamplands. Drainage of wetlands for economically productive farming has been a continuing trend, leaving few of New Zealand's native lowland wetlands intact and compromising their capacity to provide habitats for plants and other wildlife. The New Zealand Wetlands Management Policy recognizes the difficulty of reversing this trend towards drainage for short-term gain.

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Keywords
Maori · Wetland loss · Drainage · WERI · Tourism · Recreation · Conservation · Hydrological role

Introduction

In 1986, New Zealand became the first Contracting Party of the Ramsar Convention to publish a Wetland Management Policy. A scanned copy on the *New Zealand Wetlands Management Policy 1986,* can be found on the Ramsar Commission (n.d.) website. However, the New Zealand Department of Conservation has added a note stating that: "*As the first National Wetland Policy by a Contracting Party, this document is becoming outdated in terms of both the evolution of the Convention and the changes in legislation and governance structures in New Zealand during the past thirteen years. Accordingly, a stock-taking of implementation is currently under way, which may lead to a formal review of the policy*".

The contents of this chapter are derived from the wording of the *New Zealand Wetlands Management Policy*, though with the cautionary note that interpretation and context have changed since its publications and that the New Zealand Government is currently considering updating the Policy.

New Zealand's Wetlands

The *New Zealand Wetlands Management Policy* recognizes that the wetlands of New Zealand have always been an important part of the nation's environment. The earliest Maori settled and harvested shellfish and fin fish, including eels, from the coastal estuaries and lagoons, and collected material for weaving as well as snaring waterfowl from the flax swamps. Flax fibres from these flax swamps subsequently assumed economic importance, enabling establishment of the New Zealand settlement, and the extensive flat swamplands yielded fertile soil when drained, sustaining farmers and supporting sheep and dairy cows. Drainage became a major cultural activity, turning New Zealand into an economically productive land.

Consequently, few of New Zealand's native lowland wetlands remain intact, compromising their capacity to provide habitats for plants and other wildlife including rare species, natural landscapes, water storage systems and filtration plants for managing floods and water quality, and recreational pursuits like hunting waterfowl and fishing.

Development of the *New Zealand Wetlands Management Policy*

The *New Zealand Wetlands Management Policy* recognizes the difficulty of reversing this trend towards drainage for short-term gain. The situation at that time was exacerbated by little legislation for protecting wetlands yet a lot of policy and expertise facilitating their destruction, further compounded by the scattered nature of government agencies responsible for wetlands hampering coordinated policy.

Ratification of the *New Zealand Wetlands Policy* foreshadowed establishment of the Department of Conservation (http://doc.govt.nz/), which after vesting took a lead role in advocacy for wetland protection. Today, the Department for Conservation divides its roles with respect to New Zealand's natural environment, including its wetlands, into: protecting species; restoring places; monitoring; managing threats; and developing proposals (often in collaboration with others) for new parks, reserves and projects on conservation land. WERI, the national wetlands inventory, serves as the database for implementation of the *New Zealand Wetlands Policy*.

In approving the *Policy*, New Zealand's Cabinet Policy Committee noted "*. . .that the policy is intended to indicate that in broad terms the Government regards the protection of representative important wetlands as being desirable, rather than to bind the Government to any course of action or to justify restrictions on the actions of the private sector*". This is hardly the most ringing of endorsements of the need to preserve the broader values provided by the nation's dwindling wetland resource against a range of development pressures, though the statement has to be interpreted today in the context of the time in which it was made.

The *New Zealand Wetlands Management Policy*

The *New Zealand Wetlands Management Policy* defines wetlands as "*A collective term for permanently or intermittently wet land, shallow water and land-water margins. Wetlands may be fresh, brackish or saline, and are characterised in their natural state by plants or animals that are adapted to living in wet conditions*". It also recognizes the legacy of wetland loss, and with it the degradation of a range of beneficial hydrological, ecological, economic, recreational, educational and cultural services, noting that "*The need to preserve representative natural ecosystems already has public support and has been embodied in legislation*".

A number of key considerations are noted in the Policy, beyond the issues of public support and Government intent to offer wetlands – rare types of wetlands in particular – greater protection as noted above. These broader considerations include:

- Wetland management must consider causes and consequences beyond the wetland boundary;
- The long term benefits lost by modifying wetlands frequently do not justify the short term benefits gained; and
- Wetland modification may cause irreversible changes.

Objectives of the *New Zealand Wetlands Management Policy*

Three principal objectives are noted in the *New Zealand Wetlands Management Policy*, each supported by a range of sub-objectives. Quoting from the Policy document:

1. Preservation and Protection
   - To act urgently to protect by reservation additional wetlands that fulfil the criteria of the International Union for the Conservation of Nature and Natural Resources (IUCN) for Wetlands of International Importance.
   - To protect wetlands of national importance, and where appropriate, wetlands of regional and local importance.
   - To gain adequate permanent protection of representative examples of all types of wetland in private and public ownership. Priority will be given to preservation of the least modified and most ecologically viable examples of each kind.
   - To retain or re-establish wetlands significant for the protection or enhancement of aesthetic, scenic, recreational and tourism values.
   - To protect and manage habitats important for native flora and fauna, giving priority to rare and endangered species and habitats of importance to migratory bird species.
   - To protect, enhance, or re-establish wetlands and their access ways which are important for fish.
   - To promote the concept of managing all wetland catchments so that the complex relationships that exist within a wetland, and between a wetland and surrounding ecosystems, are taken into account.
   - To protect and manage wetlands that have an important hydrological role in such a way as to maintain or enhance that role.
2. Wetlands Inventory
   - To maintain an inventory of the most significant wetlands.
   - To link the national inventory for wetlands with other related government resource inventories to ensure optimal compatibility of the inventory.
3. Public Awareness
   - To promote public awareness of wetland values and encourage public participation in the planning and management of wetlands.
   - To preserve and enhance the opportunities afforded by wetlands for education, scientific study and recreation.
   - To promote the tourism and recreational potential of wetlands.

New Zealand's Department for Conservation today offers educational resources and other means for people to get involved with wetlands, including for example web pages dedicated to 'Wetlands Conservation' (Department for Conservation n.d.)

Next Steps

As noted in the *Introduction* of this chapter, the New Zealand Department of Conservation acknowledges that the current Policy has become outdated, and requires formal review. Nonetheless, interpretation of the *Policy* has evolved since its publication.

Conclusions

On publication, the *New Zealand Wetlands Management Policy 1986* constituted the first such national policy by a Contracting Party of the Ramsar Convention, recognizing long-standing losses to the nation's wetland resource and with them their substantial and diverse societal values. Measures put in place in the intervening years include institutional arrangements and use of emerging science. Today, the Department of Conservation recognizes the need to review the *Policy*.

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National Wetland Policy: South Africa

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John A. Dini and Mark Everard

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Abstract

Recognising the scarcity, value and threatened status of its wetlands, South Africa has focused much energy on wetlands at the level of national policy in the last four decades. At the international level, South Africa was a founding member of the Ramsar Convention on Wetlands. The country opted not to develop a stand-alone wetland policy, but to incorporate objectives relating to wetland conservation and wise use into relevant sectoral policies, including those covering environment, agriculture, biodiversity, and water. Some of the tools to support the implementation of wetland-related policy are discussed, together with ongoing challenges.

Keywords

National wetland policy · South Africa · Ramsar convention on wetlands · Mainstreaming

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Introduction

Wetlands mapped in South Africa to date cover a total area of 2.9 million ha or 2.4% of the country's surface area (Driver et al. 2012). This low extent of coverage is largely attributable to climatic conditions that are not conducive to the persistence of surface water. South Africa's average annual rainfall of 497 mm is well below the global average of 860 mm. Average annual potential evaporation over the majority of the country far exceeds rainfall.

Although wetlands make up such a small proportion of South Africa's surface area, they provide an array of valuable benefits to people. Many of these wetlands comprise natural infrastructure that assists in managing the country's limited water resources. Many also have biodiversity significance and are sites of spiritual, recreational, and scientific value. With climate change predicted to significantly alter South Africa's rainfall patterns (Knoesen et al. 2009), wetlands will play a more important role than ever before in mitigating the impacts of extreme events like floods and droughts. By providing highly productive opportunities for small-scale cultivation, as well as resources such as grazing, water, fish, fiber, and medicine, wetlands continue to underpin the health and livelihoods of many people in South Africa, especially poor rural communities.

Consistent with global trends, high levels of threat to the country's wetlands have been reported. The 2011National Biodiversity Assessment identified wetlands as the most threatened ecosystem type in South Africa (Driver et al. 2012). At least one third of indigenous freshwater fish are listed as threatened, and a recent study of the conservation status of fish, mollusks, dragonflies, crabs, and vascular plants in southern Africa reported far higher levels of threat in South Africa than in the rest of the region (Darwall et al. 2009). As a result of the limited extent of wetlands in South Africa, their loss and degradation will have more severe consequences than if the country was better endowed with larger and more extensive wetlands, particularly considering the context of South Africa as a semiarid country (Kotze et al. 1995).

Perhaps as a result of all these factors – scarcity, value, and threat – much energy has been focused on wetlands at the level of national policy in the last four decades. The current policy environment for wetlands is outlined further in the sections that follow, starting with relevant international agreements.

South Africa and the Ramsar Convention on Wetlands

At the international level, South Africa played a leading role in the formalization of the Ramsar Convention on Wetlands, becoming the fifth contracting party in 1975. The Convention entered into force the same year, following the accession of the seventh contracting party, and South Africa is thus considered a founding member.

The country's legal system is such that an international agreement like the Ramsar Convention must be transposed through the adoption at national and subnational levels of policy and legislative frameworks necessary to give effect to key elements of the Convention. For example, South Africa designates sites to the Ramsar List of Wetlands of International Importance, but these sites do not automatically receive protection under South African law simply by virtue of their Ramsar designation. Experience has shown that statutory protection is necessary if the country is to uphold its obligations under the convention in respect of these sites. For this reason, the Ramsar Administrative Authority in South Africa, the Department of Environmental Affairs, requires that sites proposed for Ramsar designation are proclaimed as a category of protected area under a legal instrument such as the National Environmental Management: Protected Areas Act.

Wetland-Related Policy in South Africa

The adoption and implementation of national wetland policies by contracting parties has emerged as a high priority for Ramsar, and much energy has been invested by the convention in encouraging the establishment of such policy frameworks at national and subnational levels (Bowman 2002). South Africa opted not to develop a standalone wetland policy, but to incorporate objectives relating to wetland conservation and wise use into relevant sectoral policies, including those covering environment, agriculture, biodiversity, and water. The "mainstreaming" of wetlands into those sectors with a high potential to impact on wetlands was considered to be the advantage of this approach. This made wetlands part of the business of these sectors, rather than creating a separate "wetlands sector" whose ownership was seen to lie with a traditional wetland champion like an environmental department.

As in other countries, agriculture has been one of the biggest drivers of wetland loss in South Africa (Kotze et al. 1995). It is therefore noteworthy that, for over a decade prior to the advent of the current suite of environmental and water legislation, the most powerful legal instrument to protect wetlands situated outside protected areas was the 1983 Conservation of Agricultural Resources Act. To this day, it remains a powerful legal tool for regulating the drainage and cultivation and utilization of vegetation in wetlands. The Act provides a clear mandate to the Department of Agriculture, Fisheries and Forestry (DAFF) with respect to wetland conservation and wise use, primarily from the perspective of ensuring the sustainable use of agricultural natural resources.

More recently, a range of other sectoral policy frameworks have incorporated wetlands. Government's environmental management policy (DEAT 1998) was formulated to give effect to constitutionally enshrined rights to an environment that is not harmful to health or well-being and that is protected for the benefit of present and future generations.

Framework legislation giving effect to these rights and principles was enacted in the form of the 1998 National Environmental Management Act. The Act places particular emphasis on "sensitive, vulnerable, highly dynamic or stressed ecosystems, such as coastal shores, estuaries, wetlands and similar systems," providing for specific attention to be given to these in management and planning procedures. Environmental impact assessment regulations published under the Act include a list of activities that have potentially significantly detrimental effects on the environment (DEA 2010). A number of activities pertaining directly or indirectly to wetlands are covered, including construction of aquaculture facilities, dams, canals, and infrastructure for water transfer; dredging, filling, and excavating in wetlands; and the extraction of peat.

Similarly, policy on the conservation and sustainable use of biodiversity (DEAT 1997) has been expressed in law through the enactment of the Biodiversity Act (2004) and Protected Areas Act (2003). The National Biodiversity Framework (DEAT 2009), published under the Biodiversity Act, provides a vehicle for coordinating and aligning efforts to conserve and manage South Africa's biodiversity. The establishment of a focused program to conserve freshwater biodiversity is listed as an urgent priority, together with other actions focused on implementing cross-sector policy objectives for conservation of freshwater biodiversity, incorporating biodiversity conservation objectives into the work of catchment management agencies, and placing emphasis on freshwater ecosystems in programs for ecosystem-based adaptation to climate change.

Perhaps most significantly from a policy perspective is national water policy (DWAF 1997) and one of its statutory expressions, the 1998 National Water Act. These were globally hailed as innovative and progressive for their explicit recognition that water resources are more than just water, that aquatic ecosystems comprise the resource base on which all other uses depend, and that healthy ecosystems underpin the sustainability of water use (Rowlston 2011). The Act contains three sets of Resource Directed Measures to protect aquatic ecosystems in order to secure ecologically sustainable development and use of water resources:

1. A system for classifying water resources (rivers, springs, wetlands, lakes, surface water, estuaries, and aquifers) into management classes, which in turn determine the balance between the use and protection of individual water resources. The system uses social, economic, and environmental criteria to identify which resources need protecting and which will be heavily used to accommodate social and economic needs.
2. The Ecological Reserve, which prescribes the minimum amount of water required to maintain the functioning of aquatic ecosystems, will vary according to the management class of the resource.
3. Associated with the management class of a water resource are a set of Resource Quality Objectives, which set out the quantity (including variability over time) and quality of water, assurance of instream flow, and character and condition of instream and riparian habitat and biota.

Supporting the Implementation of Policy

As is clear from the preceding discussion on wetland-related policies, the multifaceted and cross-cutting nature of wetlands requires what the South African Constitution terms "cooperative government" involving a wide range of sectors at multiple levels of governance. In order to deal with the reality of overlapping and sometimes conflicting sectoral policy mandates, the holders of these mandates jointly developed a set of cross-sector policy objectives for conserving South Africa's freshwater biodiversity. Under a common national goal for conserving freshwater biodiversity, a set of policy objectives has been articulated that collectively establish quantitative conservation targets for freshwater ecosystems, plan for representation and persistence of freshwater ecosystems, establish a portfolio of potential freshwater conservation areas, and support practical implementation (Roux et al. 2006).

A subsequent multipartner initiative elaborated on some of these objectives by producing maps of National Freshwater Ecosystem Priority Areas (NFEPAs). These maps constitute a single, nationally consistent information source of strategic spatial priorities for conserving freshwater ecosystems (Driver et al. 2011; Nel et al. 2011). The maps provide guidance on which rivers, wetlands, and estuaries should remain in a natural or near-natural condition to support the water resource protection goals of the National Water Act. They are also directly relevant to the listing of threatened freshwater ecosystems and bioregional planning processes provided for by the Biodiversity Act, and to the expansion of the protected area network through the Protected Areas Act.

Future Challenges

After two decades under a postapartheid constitutional dispensation, some hard realities are making themselves felt. The first is that South Africa is not short of policy. The problem lies primarily with implementation, rooted largely in insufficient resources and capacity, as well as weaknesses in the structures and practices of cooperative government.

This is particularly apparent with the National Water Act which, despite being hailed as a global forerunner, has proven difficult to implement. Resource Directed Measures have yet to be implemented across most of the country (Pienaar and King 2011). While much learning and preparatory work has been done since the Act's inception in 1998, the principal challenge ahead remains translating the Act's vision into action on the ground.

Overcoming these challenges will also require the employment of freshwater and wetland ecologists in the relevant government agencies (especially newly formed catchment management agencies) and municipalities to promote sustainable water development decisions.

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National Wetland Policy: Taiwan

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Terence Lee

Contents

Abstract

Taiwan is an island located to the west of the Pacific Ocean and the southeast edge of Mainland China. Taiwan has a total area of 36,192 km^2. However, as much of the land comprises hills and mountains, only less than 40% of this land area is suitable for farming, industrial, urban, and rural development. Nevertheless, Taiwan is host to a diversity of types of wetlands including high-mountain lake and other natural ponds, fishponds and irrigation systems, creeks, and rivers, in addition to coastal mudflats, marshes, lagoons, and mangroves. These different types of wetlands form a network of ecological corridors across Taiwan. Politically, a complexity arises in that sovereignty of the islands is contested, though it is generally recognized as being part of the Republic of China. The Wetland Conservation Act (WCA), announced by the President Order on July 3, 2013, is the foundation of wetland management in Taiwan. Beside the regulations, the

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policies and actions of conserving wetland follow the guidance of "Taiwan's Wetlands of Importance Conservation Program (2011–2016)" that are based on the principle of wise use under the Ramsar Convention.

Keywords
Taiwan · Typhoons · Paddies · Wetland Conservation Act · WCA · Development · Pollution · Green Belt

Introduction

Taiwan is an island located to the west of Pacific Ocean and the southeast edge of Mainland China. Taiwan has a total area is 36,192 km^2. However, as much of the land comprises hills and mountains, only less than 40% of this land area is suitable for farming, industrial, urban, and rural development. Nevertheless, Taiwan is host to a diversity of types of wetland including high-mountain lake and other natural ponds, fishponds and irrigation systems, creeks, and rivers, in addition to coastal mudflats, marshes, lagoons, and mangroves. These different types of wetlands form a network of ecological corridors across Taiwan. Politically, a complexity arises in that sovereignty of the islands is contested, though it is generally recognized as being part of the Republic of China.

The Wetland Conservation Act (WCA), announced by the President Order on July 3, 2013, is the foundation of wetland management in Taiwan. Beside the regulations, the policies and actions of conserving wetland follow the guidance of "Taiwan's Wetlands of Importance Conservation Program (2011–2016)" that are based on the principle of wise use under the Ramsar Convention.

Background

Taiwan is situated on the earthquake belt where the Euro-Asia crust meets Pacific crust. It is the conflict of these crustal plates that creates the high mountain ranges of the small island. Every summer, typhoons bring in torrential rain, which washes soil and rocks into the island's creeks and rivers, its passage down to the ocean creates plains, forms ponds and lakes, irrigates paddies and farms, and deposits sediment on mudflats, lagoons, and mangrove stands along the coast. These diverse wetlands form a network that is closely connected with scrubland and forests, forming important ecological corridors across the island.

Due to the limited availability of land for human uses, population density and urban sprawl place significant pressure on environmentally sensitive areas including wetlands. (Census data from 2011 place the human population of Taiwan at 23,293,593, representing a population density on available land of 1609 per km^2.)

The first wave of wetland designations in 2007 in Taiwan recognized 75 wetlands of importance, with a further seven sites of wetland of importance recognized by January 2011. By the end of 2013, the number of wetlands of importance recognized in Taiwan numbered 83 sites. These include two sites considered to the of international importance: the Zengwun Estuary wetland and Sihcao wetland, within the Taijang National Park, which are the winter habitats of black-faced spoonbill (*Platalea minor*). There are an additional 40 wetland sites of national and 43 sites of regional importance, each of these sites performing different functions and with different targets for conservation such as management as wildlife refugia, for aquaculture, irrigation, urban recreation, sewage treatment, or/and flood management. The total area of these wetlands of importance comprises 56,860 ha.

Issues

In addition to the implications of climate change, Taiwan's wetlands are facing two principal threats: development and pollution.

Most of the wetlands in Taiwan are closely connected with people's daily life and lie close to rural or urban areas. The demands for land by industry and housing have caused many wetlands to have been filled and transformed for construction and development.

The second threat is pollution, especially for coastal wetlands. Formerly, wetlands were regarded and consequently treated as low value and marginal land and were commonly used as dumping grounds and sites for illegal waste management. This situation is common to many countries, not just in Taiwan, and has profound impacts not merely on the landscape but also for damage to coastal fisheries, food webs, ecology, and a host beneficial services.

Policy

Based on Article 18 of Taiwan's Basic Environment Act, "Government entities at all levels shall actively protect wildlife, ensure biodiversity, protect forests, estuaries and wetland environments, and maintain a diverse natural environment . . .." This is backed up by scientific support and environmental education. Taiwan's wetland policy includes three main aspects:

Building up the Regulation and Management System for Wetland

The Wetland Conservation Act (WCA), which was promulgated and announced by Presidential Order No. Hua-Zong-Yi-Yi 10200127201 on July 3, 2013, contains two core values: biodiversity and wise use. The Act comprises four parts:

(a) Assessment and designation of wetlands of importance
(b) Conservation and utilization plans for wetlands of importance
(c) Wise use management
(d) Development mitigation and ecological compensation

The Act is therefore the principal instrument for achieving wetland conservation and management in Taiwan.

Beside the WCA, there are connections with and cooperation in implementation of the Environmental Impact Assessment Act (EIA), Regional Plan Act (RPA), Urban Plan Act (UPA), and related land use management regulations are also important and necessary in supporting wetland management. The key challenge is to incorporate the spirit of wise use into these regulations.

Developing Financial Tools and Creating Market Mechanisms for Wetlands

Wetlands support the provision of food, water resources, recreation, and many other dimensions of well-being for people's daily lives in Taiwan. Thus, using market mechanisms to support wetland conservation has significant potential in terms of internalizing these often formerly overlooked values. There may be potential for creation of internal markets, for example where a proportion of profits from products derived from wetlands, especially related to agricultural production, could be recirculated to promote wetland conservation and hence resource sustainability.

Central government budgets also recognize the importance of wetland conservation, with a three-year program of support and cooperation with local government bodies, colleges, NGOs, and communities for wetland conservation programs and activities. This strategy has proven effective for promoting private sector and public awareness and interest in addressing wetland issues and conservation.

Given the global financial situation, and specifically its tightening of public finances, Taiwan is seeking funds from these novel market mechanisms as a partial replacement for central government support.

Using Environmental and Land Use Planning Tools to Preserve Wetlands and Form Ecological Corridors

Wetlands are a key element of the ecological corridors across Taiwan, linking with forest, bush, and urban open space habitats. The concept of a "green belt" is worth progressing, and may in time form a key tool is the spatial planning system.

Taiwan's strategy for development of a green belt is to preserve forest, bush, and natural wetland resources, including designation of wetlands of importance, as well as promoting more environmentally sensitive paddy field management. There is also interest in investment in constructed wetlands, which may help address flood risk management, urban recreation, pollution control, and a range of other environmental values. Wetland systems play important roles in supporting a range of beneficial services and so, as part of a network of ecological corridors, need to become incorporated into regional planning processes. The spirit of wise use will be fulfilled when wetland site management is integrated within a wider approach to urban land use and wider spatial planning.

Future Challenges

Although the WCA was announced in 2013, its development will face continuing resistance from a range of stakeholders including challenges from developers. For example, concerted representations from construction, real estate, fishery, agriculture, and other interests could limited the potential of the Act to protect wetlands of importance and the efficacy of related regulations. Possible solutions to this potential outcome include implementing appropriate mechanisms, such as agreed methods of management or building codes, that are not only science-based but also embody the spirit and principle of wise use.

Incorporating ecosystem management into market mechanisms also represents a significant opportunity for natural resource management. Although wetlands have many associated use and nonuse services, offering direct and indirect benefits for human society, a few, except some physical products, have current market values assigned to them. Other wetland values may be amenable to assigning market prices, including for example the links between wetland fauna and cultural creativity which may offer inducements for wetland conservation. Extension of market mechanisms to nontraditional wetland services needs considerably more research.

Finally, it is important to educate and influence wider sectors of the public about the importance of wetlands in supporting their well-being, in addition to the inherent and ethical values of protecting habitats and wildlife for future generations. Understanding how best to connect these values with those held by people of different groups and ages requires more fundamental research if Taiwan's wetlands and their services are to become mainstream societal concerns.

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National Wetland Policy: Uganda

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Abstract

Uganda's wetlands are widespread and complex. Wetlands cover an estimated 13% of Uganda's land surface, constituting the most widespread ecosystem type in the country. Uganda's wetlands contribute immensely to people's livelihoods and to the health of the environment, through processes such as water storage and purification, products obtained from ecosystems, benefits from regulation of ecosystem services, nonmaterial benefits obtained from ecosystems, and services necessary for the production of all other ecosystem services.

Keywords

Uganda · Sustainable management · No further drainage · National Wetlands Conservation and Management Programme · NWP · Uganda National Wetlands Policy · Ramsar Centre for Eastern Africa

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Introduction

Uganda's wetlands are widespread and complex. Wetlands cover an estimated 13% of Uganda's land surface, constituting the most widespread ecosystem type in the country. Uganda's wetlands contribute immensely to people's livelihoods and to the health of the environment, through processes such as water storage and purification, products obtained from ecosystems, benefits from regulation of ecosystem services, nonmaterial benefits obtained from ecosystems, and services necessary for the production of all other ecosystem services.

Derivation of the Uganda National Wetlands Policy

An officially recognized National Wetlands Policy is an essential step towards the sustainable management of wetlands in Uganda and was the main thrust during the first four years of existence of Uganda's National Wetlands Programme. With hindsight, it is clear that the time and human resource requirements necessary for the process of developing the National Wetlands Policy were severely underestimated.

In Uganda, wetlands policy development started with very little political support, awareness, and information. Rushing policy development under such circumstances runs the risk of creating a document that has little support beyond its initiators, and which may not be realistic in terms of its objectives and implementation mechanisms. However, postponing publication of a wetlands policy until the whole country is behind the idea, and when all relevant information has been collected and digested, would result in unhelpful delays, potentially including postponing policy development indefinitely.

From the outset, wetland management efforts were directed towards policy formulation and development, and development of a specific institutional framework for wetland management. The need to establish a specific institution and legislation for wetland management came with the Policy on *No Further Drainage,* declared in 1986 by His Excellency the President of Uganda. This pronouncement came against a backdrop of rampant degradation, which was proceeding unabated at the time.

In 1989, the Uganda Government established the National Wetlands Conservation and Management Programme (NWP) to develop policies and implementation strategies for the sustainable management of Uganda's wetlands. This came in the wake of a slowly emerging realization in Government circles and civil society that wetlands are extremely important ecosystems that contribute considerably to the national economy and rural livelihoods.

The Uganda National Wetlands Policy was adopted in 1994, making Uganda the first country in Africa to adopt a national wetlands policy.

The Aims, Principles and Strategies of Uganda's National Wetlands Policy

The Uganda National Wetlands Policy aims at promoting the conservation of Uganda's wetlands in order to sustain their ecological and socioeconomic values for the present and future wellbeing of the people. In support of this aim, the National Wetlands Policy sets five goals:

- To establish the principles by which wetland resources can be optimally used now and in the future
- To end practices which reduce wetland productivity
- To maintain the biological diversity of natural or seminatural wetlands
- To maintain wetland functions and values
- To integrate wetland concerns into the planning and decision making of other sectors

Three principles apply in pursuit of these goals:

- Wetland resources form an integral part of the environment and their management must be pursued in the context of interaction between conservation and the national development strategies and activities.
- Wetland conservation can best be achieved through a coordinated and cooperative approach involving all stakeholders.
- It is of vital importance for wetland conservation and management that current attitudes and perception of Ugandans regarding wetlands are changed.

The National Wetlands Policy lays out several strategies to achieve the policy goal. These include, but are not limited to, the following:

- Ensuring that there is no drainage of wetlands unless more urgent environment requirements supersede
- Ensuring that only non-destructive activities are carried out in and around wetlands
- Ensuring that wetland developments are subject to environment impact assessment and environment audits
- Maintaining an optimum diversity of uses and consideration of other stakeholders when using a wetland

Successes Under the National Wetlands Policy

More than 20 years of work in wetlands management has greatly increased understanding of how wetland ecosystems work, and what they do for local users and the country as a whole. This has lead to the conclusion that wetland ecosystems in

Uganda must be recognized as a vital natural resources sector, very similar to forests or agriculture.

Success has been registered in wetland conservation and management. There is increased awareness of wetlands, and wetlands are now high on the political and public agenda as exemplified by the debates in cabinet and parliament as well as the coverage in both print and electronic media. Wetlands of Uganda have also been mapped and an inventory has been completed, and wetland issues have been incorporated into key legislation such as the Constitution of 1995, the National Environment Act of 1995, and the Land Act of 1998. Community participation remains a key tool used by the department to promote best use guidelines and designation of Ramsar sites. There is also increased reporting of wetland abuse cases, and more people are seeking guidance on how to manage wetlands.

However, success notwithstanding, challenges still exist due to the complex tenure and ownership of land, weak enforcement of the law, a narrow knowledge base, conflicting interests, the effects of climate change, development and population pressure, and disregard of environmental laws with impunity. To address these challenges, a new strategic direction 2011–2020 has been developed and is intended to focus on enhancing the productivity and service provision of wetlands, strengthening of regulatory frameworks and equity in stewardship, strengthening institutional and technical capacity, enhancing public and stakeholder awareness, and encouraging participation in wetland management and resource mobilization.

So has the policy worked? An official Wetland Policy is a declaration of intent to develop legislation, but of itself carries no legal weight as far as enforcement is concerned. Uganda was fortunate in that, between 1995 and 1998, an array of important laws were produced, which created the opportunity to insert wetland clauses into key legislation. Between 1995 and 1998, wetland clauses were incorporated into the National Environment Statute (1995), the Constitution (1995), the Local Government Act (1997), and the Land Act (1998).

This quick succession of laws and attendant wetland clauses gave the National Wetlands Policy statutory "muscle." It also created, however, a certain level of confusion, notably on the status of wetlands as a natural resource. This first started when the Constitution (1995) vested "*...the ownership of land in the people of Uganda,*" but also stated that "*...wetlands...shall be held in trust by the government for the common good of all people.*" This trustee doctrine has put wetlands in the same class of natural resources as forest reserves, rivers, lakes, and national parks. At first glance, this would seem to be a tremendous opportunity for strict wetland management. However, Uganda is today still grappling with the full implications of this situation. Large tracts of Uganda's seasonal wetlands are managed under customary management regimes, either as common or individual property. The new Constitution takes the management responsibility away from the original "owners" and vests it in a trustee, which as yet remains unidentified.

Important Lessons Learnt

A number of lessons have been generated and learnt during the policy development process. Key among these is the fact that wetland development is a long-term process. The initial plan was to have a 2-year project to develop the National Wetlands Policy. As it turned out, the policy was adopted 4 years and 20 years on, and implementation continues. Development of the scientific and socioeconomic knowledge base takes time, money, and effort, but is important in informing decision-making processes, and requires continued targeted, specific, and general awareness-raising. Engagement of all relevant stakeholders is also key to successful interventions. Effective implementation of laws and the functioning of associated institutions require adequate human and financial resources. Policy and institutional harmonization is also key to strengthening operations, political will and support is vital to successful program, and land ownership issues must be clarified from the outset if effective enforcement is to be realized.

Furthermore, communities cannot conserve wetlands without economically viable livelihood options, incentives, and benefits. In other words, there is a need to have "linked enterprises": enterprises which will make people love and value the natural resources. These linked enterprises, for example ecotourism and measures such as catchment-related activities which contribute to soil and water conservation and the introduction of piggeries and goat-keeping in the catchment, need to be economically viable and also deliberately to reduce pressure on natural resources, specifically including wetlands. Cross-sectoral collaboration and increased engagement with relevant legal and management institutions and capacity development are key to institutional sustainability. It is also important to create political and public interest by using the right arguments, leading to the acceptance of modified user systems and co-opting as many organizations as possible.

In conclusion, wetlands in Uganda are vast, complex, and extremely valuable ecosystems. Their value is derived not so much from the products they deliver to the resource user, but from the hydrological and ecological services they provide almost unnoticed. However, planners, economists, and politicians usually have no idea what wetlands do for the economy, public health, or the prosperity of their countries. They often see wetlands only as valuable land for conversion for agriculture, industrial estates, or urban development, and believe that they can do away with these wetlands through infilling or channeling water. Local users, on the other hand, may have a better understanding of the value of wetland products, but that understanding does not necessarily lead to sustainable wetland use since maximizing outputs of a few products may result in the collapse of the whole system, and notably its broader set of ecosystem services. Misconceptions about wetlands are so deep and widespread that turning people away from wetland degradation is not easy and requires a long-term outlook, including considerable persistence, creativity, and resources. However, the task is far from impossible.

The tremendous work accomplished in the area of wetland conservation and management in Uganda has been a motivating and evolving process, which has made Uganda a frontrunner, not only in Africa but the world over. Currently, many countries across the world are keen on benefiting from our rich and informative experience. For this purpose, the Ramsar Centre for Eastern Africa (RAMCEA) has been established to build capacity of wetland managers and practitioners.

Certainly, as Uganda continues this learning journey, which started with a political pronouncement on “No further drainage of Wetlands,” we are now more confident that almost two decades of wetland conservation provide the benchmark for future success. In addition, the upgrading of what started out as a National Wetlands Programme, now constituting a fully fledged Wetlands Management Department, makes the task ahead much easier in the knowledge that more stakeholders are informed and willing to join the battle to jealously guard and protect our wetlands through the promotion of their wise use.

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National Wetland Policy: USA

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Abstract

United States wetland policy has developed over the past four decades. Prior to the 1970s, government policy largely reflected the view of wetlands as unproductive swampland that posed a nuisance to human health, promoting drainage and repurposing for agriculture. Concern about wetland loss emerged only after its negative impacts to fish and waterfowl became apparent. Lack of restriction on wetland conversion leads to the loss of more than 50% of total wetland acreage in the contiguous United States over a 200-year period ending in 1980. In 1972, the US Congress passed the Clean Water Act, amending an existing law to create what remains the lynchpin of federal wetlands policy. As understanding of the importance of wetlands grew, a broad range of other federal legislation and programs advancing wetlands protection emerged. By 1987, the

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NGO Conservation Foundation convened a cross-sectoral forum leading to recommendation that national policy be guided by the goal of "no net loss" that was partially successful in reducing the rate of wetland loss but which did not account for regional differences and impacts on wetland functionality.

Keywords
US · Jurisdiction · Supreme court · Federal authority · No net loss · Mitigation · Army Corps of Engineers

Introduction

United States wetland policy has developed over the past four decades. Prior to the 1970s, government policy largely reflected the view of wetlands as unproductive swampland that posed a nuisance to human health. The Swamp Lands Acts of 1849, 1850, and 1860, for example, provided states with control over large wetland areas in order to promote their drainage and repurposing for use in agriculture. Concern about wetland loss emerged only after its negative impacts to fish and waterfowl became apparent, in part due to federal laws such as the Fish and Wildlife Coordination Act of 1934 and the US Fish and Wildlife Wetlands Inventories of 1954 and 1973 (National Research Council 1992). This policy of promoting or not restricting wetland conversion lead to the loss of more than 50% of total wetland acreage in the contiguous United States over a 200-year period ending in 1980 (Dahl 1990).

In 1972, the US Congress passed the Clean Water Act, amending an existing law to create what remains the lynchpin of federal wetlands policy. Under the Clean Water Act, two federal agencies, the Environmental Protection Agency (the EPA) and the Army Corps of Engineers, are jointly charged with administering a permitting program that prohibits the conversion of protected wetlands unless certain conditions are met. As understanding of the importance of wetlands grew, a broad range of other federal legislation and programs advancing wetlands protection emerged. By 1987, as the rate of wetland loss had slowed but not halted, the EPA requested a nongovernment organization, Conservation Foundation, to convene a forum of government, business, academic, and NGO stakeholders to address how the nation should protect and manage its wetlands. The forum issued as its key recommendation that national policy be guided by the goal of "no net loss" of remaining wetlands, and this policy has become the defining feature of US wetland management. President George HW Bush adopted the no net loss policy in 1989, and subsequent administrations have each endorsed the goal.

Despite adoption of a clear "no net loss" objective, US wetlands management policy has had mixed results. From an average loss of 458,000 acres of wetland per year during the 1950s–1970s, annual losses dropped to 290,000 acres from the mid-1970s to mid-1980s, and more dramatically to 58,500 acres from 1986 to 1997 (Dahl 2006). From 1998 to 2004, wetland acreage *increased* an average 32,000 acres per year. This net growth, however, reversed from 2004 to 2009, with wetland conversion again slightly outpacing wetland creation and restoration for a net loss of an estimated 62,300 acres during the 5-year period. These figures do not capture regional trends in wetland loss, changes in wetlands quality or functionality, or shifts in wetland types, which are also important indicators of the success of wetland policy (Dahl 2011).

Key Features of U.S. Wetland Policy

Limits on Federal Jurisdiction

Federal authority to regulate wetlands is limited because the US Constitution provides only certain enumerated powers to the federal government and retains the balance of authority to State governments. Federal authority to manage wetlands stems from its power to regulate interstate commerce. While the law is somewhat in flux, in general, the federal government may assert jurisdiction over wetlands if those wetlands, alone or in combination with similarly situated wetlands, significantly affect a waterway traditionally subject to federal jurisdiction (i.e., navigable waters that historically served as conduits for interstate commerce). The US Supreme Court established the operable test for federal jurisdiction over wetlands in a 2006 decision (Rapanos v. United States). Because the current test does not provide clear, categorical guidance as to which wetlands are jurisdictional, dispute has arisen surrounding federal agencies' assertion of regulatory authority over isolated or ephemeral hydrological features like ponds, intermittent streams and remote wetlands. The EPA and the Army Corps of Engineers recently proposed a regulation that is expected to clarify the limits of federal jurisdiction.

Under the US federal system of government, states retain an important role in wetlands management. States often exercise regulatory authorities that can critically impact wetlands, including potentially authority over isolated wetlands outside of federal jurisdiction. State agencies frequently implement wetlands management programs, while federal programs provide support for such activities through grants or information-sharing. States also play a direct role in the federal permitting program, by certifying a proposed permit will comply with state water quality standards and establishing coastal zone management programs with which federal actions, including permits, must be consistent.

Federal Agencies

There is no single federal agency responsible for US wetland policy. Instead, a large variety of agencies have regulatory authority over or implement programs affecting different aspects of wetland policy.

The Clean Water Act

The Clean Water Act is the most important federal tool to advance national wetland policy. The Act's purpose is broadly "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters", Section 404 of the Act establishes a permitting regime to regulate the filling of US waters, including wetlands under federal jurisdiction. In order to receive a permit, an applicant must demonstrate to the Army Corps of Engineers that its proposed discharge (i.e., the activity resulting from the drainage or filling of a wetland) complies with what are known as the "404(b)(1) Guidelines". Assuming these guidelines are met, the Army Corps of Engineers will grant the permit unless it finds the permit to be contrary to public interest.

The 404(b)(1) guidelines aim at prohibiting a discharge unless it is demonstrated that it will not have unacceptable impacts on the aquatic ecosystem. A permit applicant must show that its proposed discharge is the least environmentally damaging practicable alternative to achieve the purpose of a project. (For example, a highway scheme may be required to result in the destruction of the smallest number of wetlands to provide desired public transit, given economic and technical constraints.) If a project does not depend on being sited within wetlands or in proximity to water, the Army Corps of Engineers assumes that such a practicable alternative is available unless a permit applicant clearly demonstrates otherwise. A permit will not be granted if the discharge has certain adverse effects, such as contributing to a violation of water quality standards or threatening an endangered species. Even if these conditions are met, a permit will not be granted unless the project takes practicable steps to minimize the harm it causes to the aquatic environment. In addition to the direct impacts of the discharge on wetlands, the agency must consider the cumulative impacts of other discharges on the ecosystem (for example, whether filling only a small quantity of wetland nonetheless impacts the quality of the water environment because it has already been severely impacted) and the secondary impacts of the project (such as the indirect impacts caused by building a road due to increased access to the environment) before issuing a permit.

Once all practicable steps to avoid adverse impact have been incorporated into the proposed activity, the Army Corps of Engineers requires mitigation for unavoidable impacts. Such compensatory mitigation may include restoration, creation, enhancement, and preservation of aquatic resources through a number of means. In 2008, the Army Corps of Engineers adopted new regulation intended to improve the success of compensatory mitigation by promoting a focus on the watershed and encouraging the use of ecological performance measures.

Federal Funding

In addition to direct regulation, a large number of federal laws provide funds or withhold federal benefits in order to promote the protection or restoration of wetlands. Other federal laws designate wetlands for particular protection.

National Measures to Protect and Restore Wetlands. Source: EPA 2005		
Estuary Protection Act (P.L. 90-454) (1968)	Department of the Interior (DOI)	Authorized the study and inventory of estuaries and the Great Lakes, and provided for management of designated estuaries between the DOI and the states
Estuary Restoration Act of 2000 (P.L. 106-457) (2000)	EPA, NOAA, USACE, FWS, USDA	Promotes the restoration of estuary habitat, develops a national estuary habitat restoration strategy, provides federal assistance and promotes efficient financing of such projects, and enhances monitoring and research capabilities
Executive Order 11990, Protection of Wetlands (1977)	Federal agencies	Requires federal agencies to minimize impacts of federal activities on wetlands
Executive Order 11988, Protection of Floodplains (1977)	Federal agencies	Requires federal agencies to minimize impacts of federal activities on floodplains
Federal Aid in Wildlife Coordination Act of 1956	DOI	Authorizes the development and distribution of fish and wildlife information and the development of policies and procedures relating to fish and wildlife
Food, Agriculture, Conservation, and Trade Act of 1990 (P.L. 101-624)	USDA National Resource Conservation Service	Water Resources Development Act of Wetland Reserve Program purchases perpetual nondevelopment easements on farmed wetlands. Subsidizes restoration of croplands to wetlands
Food Security Act of 1985 (Swampbuster) (P.L. 99-198)	USDA Farm Service Agency, Fish and Wildlife Service (FWS)	"Swampbuster" program suspends agricultural subsidies for farmers who convert wetlands to agriculture. Conservation Easements program allows FmHA FSA to eliminate some farm debts in exchange for long-term easements that protect wetlands and other areas
Migratory Bird Hunting and Conservation Stamps (1934) (Ch. 71, 48 Stat. 452)	FWS	Acquires wetland easements using revenues from fees paid by hunters for duck stamps
National Environmental Policy Act of 1969 (P.L. 91-190)	Federal Agencies	Requires the preparation of an environmental impact statement for all major federal actions significantly affecting the environment
North American Waterfowl Management Plan (1986)	FWS	Establishes a plan for managing waterfowl resources by various methods, such as acquiring wetlands

(*continued*)

National Measures to Protect and Restore Wetlands. Source: EPA 2005		
North American Wetlands Conservation Act (1989) (P.L. 101-233)	FWS	Encourages public/private partnerships by providing matching grants to organizations for protecting, restoring, or enhancing wetlands
Rivers and Harbors Act of 1938 (52 Stat. 802)	US Army Corps of Engineers (USACE)	Provides that "due regard" be given to wildlife conservation in planning federal water projects
U.S. Tax Code Tax Reform Act of 1986 (P.L. 99-514)	Internal Revenue Service	Provides deductions for donors of wetlands and to some nonprofit organizations
Water Bank Act (1970) (P.L. 91-559)	USDA Farm Service Agency	Leases wetlands and adjacent uplands from farmers for waterfowl habitat for 10-year periods.
Water Resources Development Act of 2000 (P.L. 106-541)	USACE	States that future mitigation plans for federal water projects should include "in kind" mitigation for bottomland hardwood forests
Wetlands Loan Act (1961) (P.L. 87-383	FWS	Provides interest-free loans for wetland acquisition and easements.
Wild and Scenic Rivers Act (P.L. 90-542) (1968)	DOI, USDA	Protects designated river segments from alterations without a permit

Future Challenges

The proposed EPA/Army Corps of Engineers rule may improve clarity over the scope of federal jurisdiction; however its effects are uncertain until the rule is finalized. The process to finalize the rule and then observe whether it survives legal challenge in court is likely to span several years, during which the legal uncertainty surrounding the limits of federal authority will remain a barrier to federal enforcement efforts.

Due to the limits on federal authority to regulate certain wetlands, the patchwork and divergent approach of state regulators remains a continued challenge to wetlands protection. Many states lack strong legislation to protect wetlands or fail to enforce laws already in place.

Controls over stormwater runoff, which may adversely impact wetlands due to pollutant loads, remain an evolving area of the law. Certain areas of the USA. are exploring novel methods to control runoff, but attaining widespread implementation of such controls remains a challenge.

Finally, failed mitigation represents a serious challenge to attainment of national wetlands policy. Despite federal efforts to reform wetland mitigation requirements to ensure that there is no net loss of wetland functionality due to approved projects, there remains concern that mitigation requirements are imperfectly designed and inadequately monitored, and that full compliance is not being attained.

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No Net Loss: Overview

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Mark Everard

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Abstract

The concept of No Net Loss relates to the principle under which counties, agencies, and governments strive to balance unavoidable habitat, environmental and resource losses, with replacement of those items on a project-by-project basis so that further reductions to resources may be prevented. No Net Loss is a bold and attractive proposition, which is gaining traction in addressing wetland and other environmental management challenges globally. However, its practical implementation is fraught with difficulties and the potential for derailment by vested interests. More robust and operationally applicable methods are required to assess the diversity of functions and socially valuable services provided by wetlands if their net contributions to society are to be fully assessed and factored into decision-making about the avoidance, mitigation, or compensation of wetland loss.

Keywords

No net loss · Natura 2000 · Environmental Liability · Partnership · Biodiversity loss · EU Biodiversity strategy · Swampbuster · Avoid-mitigate-compensate

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Introduction

The concept of No Net Loss relates to the principle under which counties, agencies, and governments strive to balance unavoidable habitat, environmental and resource losses, with replacement of those items on a project-by-project basis so that further reductions to resources may be prevented (EU 2014a).

No Net Loss policies have been applied to a wide range of environmental issues. One example is its application to water balance in the Lower Colorado River Basin in Texas under the House Bill (HB) 1437 Agriculture Water Conservation Program, offering an innovative way to conserve water, meet rising municipal demands, and maintain agricultural productivity (LCRA n.d.) Requirements to avoid damage to Natura 2000 sites across the European Union, designated under the EU Birds Directive, the EU Habitats Directive, and the EU Environmental Liability Directive, also require habitat replacement where losses are "unavoidable" constituting a broader No Net Loss approach to wildlife habitat.

However, the No Net Loss policy is most widely known in the field of wetland protection, safeguarding both the environment and public access to it. Across the USA, No Net Loss has been the key policy in wetlands protection at both the federal and state level, instigated during the presidency of George H.W. Bush during which time each State administration adopted the "no net loss of wetlands" policy (White House Office on Environmental Policy 1993).

The US No Net Loss Policy Concerning Wetlands

Since the eighteenth century, wetland area has decreased from nearly 220 million acres (890,000 km^2) in the lower 48 states of the USA to 107,700,000 acres (436,000 km^2) in 2004, leaving approximately 100 million acres (400,000 km^2) of wetlands (Dahl and Allord 1997). Since the 1950s, over 50% of this loss has come from wetlands being transitioned to agricultural lands (Hansen 2006). Wetland loss is also driven by other factors including development and forestry.

No Net Loss is the United States government's overarching policy goal regarding wetland preservation. The goal of the policy is to balance wetland loss due to economic development with wetlands reclamation, mitigation, and restoration such that the total acreage of wetlands in the country does not, at worst, decrease but rather remains constant or increases (Sibbing n.d.)

Barriers to implementation are numerous. In part, these stem from the fact that some 70% of wetlands are on private landholdings, requiring a partnership-based approach. Politically, influential groups can lobby or otherwise use their leverage to receive exemptions. Economically, the magnitude of opportunity costs associated with foregone profit from agriculture and development can and are used to justify wetland conversion, though the extent to which this actually takes account of losses of the broad beneficial hydrological, cultural, ecological, and other ecosystem services provided by a wetland is moot. In addition, the technical difficulty of wetland recreation is generally underestimated, while the functional equivalence of compensatory wetlands is rarely addressed.

No Net Loss in the EU

Recognizing that the target of halting biodiversity loss by 2010 had not been achieved across the European Union, with many species and habitats continuing to decline significantly, the EU (EU 2014a, b) has accepted the value of a No Net Loss policy. However, there is at present no requirement for compensation of unavoidable residual impacts outside species and habitats covered by nature legislation, regardless of their loss.

There is therefore a need for further action to promote a wider No Net Loss approach to biodiversity and ecosystem services to achieve the overall objective of the *EU Biodiversity Strategy to 2020* (EU 2012) to halt biodiversity and ecosystem service loss by 2020. Action 7 under target 2 of the *EU Biodiversity Strategy to 2020* seeks to "...ensure no net loss of biodiversity and ecosystem services." Action 7 comprises two complementary subactions:

- Action 7a foresees that "...in collaboration with the Member States, the Commission will develop a methodology for assessing the impacts of EU funded projects, plans and programmes on biodiversity by 2014." This recognizes the need for "biodiversity proofing" the EU budget to ensure that spending under the EU budget has no negative impacts on biodiversity and that it is overall supportive of achieving the biodiversity targets.
- Action 7b specifies that "...the Commission will carry out further work with a view to proposing by 2015 an initiative to ensure there is no net loss of ecosystems and their services (e.g. through compensation or offsetting schemes)."

As in most jurisdictions today, these bold statements in Europe are subsumed within an economic growth agenda. This is both a threat and an opportunity. For example, the EU's (2011) *A Roadmap to a Resource Efficient Europe* explicitly refers to the "...innovation potential of Green Infrastructure and the 'restoration economy'...," which may provide a means for increasing recognition of the multiple ecosystem service benefits flowing from wetlands and other habitats to be internalized into economic decision-making. Following an ongoing (at the time of writing) consultation exercise on *Halting biodiversity loss – the EU no net loss initiative* (EU 2014a,b), an EU No Net Loss initiative is expected in 2015.

How Does the No Net Loss of Wetlands Policy Work?

As a minimum, a No Net Loss strategy aims to maintain a minimum area of wetland throughout a state or other political land unit. However, because of the increasing demands for population and development, this wetland resource is under increasing pressure. In the USA, various instruments are applied at federal, state, and local levels to achieve the objective of No Net Loss. These include command-and-control regulation, principally under the Clean Water Act, market-based instruments (including, for example, the Swampbuster programme), catchment management strategies, stakeholder involvement, and wetlands mitigation banking.

The "avoid-mitigate-compensate" sequence is one of the ways in which this is being addressed. As its name suggests, the first approach should be to avoid damage to a wetland, for example, by siting a proposed development elsewhere or changing its design fundamentally. The second tier of approach is then to seek to mitigate harm from the development proposal. The third is provision of compensatory habitat, whether locally or more remotely, and may include wetland restoration, creation, enhancement, and reallocation.

There are major challenges in the construction and management of compensatory wetland habitat. For example, while creation of a wetted area of roughly the same size is less challenging, producing a wetland of the same character and function is far more difficult. This difficulty increases incrementally as more of the ecosystem services produced by wetlands are taken into account. For example, does the new constructed habitat have the same ecological characteristics as that which it is assumed to replace, playing host to the same biodiversity and natural processes? Is it sited such that it will still serve a valuable role in migratory flyways and other means by which species translocate across landscapes? Will it perform the same range of valuable hydrological, geochemical, educational, recreational, aesthetic, and other ecosystem services? And how do we measure these diverse parameters pragmatically, intuitively, and unequivocally in an operational context? The ► Chap. 110, "No Net Loss Case Study: Wetland Banking in Chicago (USA)" chapter highlights how ineffective wetland compensation is in general across the USA. The same observations apply equally to wetland mitigation.

Conclusions and Challenges Remaining

No Net Loss is a bold and attractive proposition, which is gaining traction in addressing wetland and other environmental management challenges globally. However, its practical implementation is fraught with difficulties and the potential for derailment by vested interests. More robust and operationally applicable methods are required to assess the diversity of functions and socially valuable services provided by wetlands if their net contributions to society are to be fully assessed and factored into decision-making about the avoidance, mitigation, or compensation of wetland loss. There are also substantial gaps in political and business understanding of how these functions and services support economic and other facets of societal progress, the dominant perception still evidently remaining that wetland conservation is a constraint on progress rather than a sound investment in underpinning capital vital for future wellbeing including economic prospects.

For a No Net Loss policy to be fully effective, realistic and economically relevant metrics (including persuasive valuation of wider nonmonetary contributions to society) need to be developed to demonstrate the multiple wider values provided by wetlands and how these factor into economic progress. At present, No Net Loss appears not to be being achieved, and so the progressive decline of natural capital and future security continues to the detriment of future prospects and security.

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109 No Net Loss Case Study: Structural and Functional Equivalence of Mitigation Wetlands

M. Siobhan Fennessy and Abby Rokosch Dresser

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Abstract

Sections 401 and 404 of the Clean Water Act require that wetlands in the United States must be created or restored as "replacement" or "mitigation" wetlands in order to compensate for wetland areas that are lost due to development. This is in order to meet the US policy mandating a "no net loss" of wetland extent or function by balancing the need for economic development with wetland preservation. A question related to this policy is whether or not mitigation wetlands are structurally and functionally equivalent, to those they are designed to replace. In other words, is wetland mitigation a fair trade? The uncertainty in whether mitigation wetlands can succeed in replacing natural sites has led them to be called a "large-scale ecological experiment."

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Clean Water Act · United States · Replacement wetlands · Mitigation · Wetland loss · No net loss · Economic development · Wetland preservation · Structurally equivalent · Functionally equivalent

Introduction

Sections 401 and 404 of the Clean Water Act require that wetlands in the United States must be created or restored as "replacement" or "mitigation" wetlands in order to compensate for wetland areas that are lost due to development. This is in order to meet the US policy mandating a "no net loss" of wetland extent or function by balancing the need for economic development with wetland preservation (Zedler 1996; NRC 2001; Burgin 2010). A question related to this policy is whether or not mitigation wetlands are structurally and functionally equivalent, to those they are designed to replace (Matthews and Endress 2008; Moreno-Mateos et al. 2012; Stefanik and Mitsch 2012). In other words, is wetland mitigation a fair trade? The uncertainty in whether mitigation wetlands can succeed in replacing natural sites has led them to be called a "large-scale ecological experiment."

Mitigation projects are classified as creation, restoration, or enhancement according to the approach used in project design. Wetland creation is defined as the conversion of uplands to wetlands, while restoration is the reestablishment of a wetland where one had existed in the past. Enhancement refers to an increase in the ecological condition of an existing wetland. In wetland mitigation, restoration projects are often found to be more successful. This is largely due to the higher probability that hydric soils, a seed bank, and appropriate hydrological conditions will be present at the site.

The goal of the case study presented here was to do a comprehensive investigation of a set of natural ($n = 9$) and mitigation wetlands ($n = 10$) located in Ohio, USA, in order to determine whether the mitigation sites were similar to natural wetlands in terms of their hydrology, soils, and vegetation (a full description of all aspects of this study can be found in the original report, Fennessy et al. 2004). As part of our analysis, we employed vegetation-based biological assessment tools to quantify the differences in the ecological condition of the sites. We assumed that if the hydrology, soil characteristics, and plant community structure of the mitigation sites were similar to natural sites, then they would also function similarly; thus they would have equivalent ecosystem structure and function, a tenet of the no net loss policy.

When selecting the natural sites we considered the hydrogeomorphic class (Brinson 1993), dominant vegetation type, relative degree of disturbance, and site access. All natural wetlands were mixed emergent marshes that spanned a gradient of disturbance from those highly impacted by human activities to those least impacted. Nine of the created wetlands were mitigation projects, created due to individual CWA Section 401/404 permits, and were selected to represent sites aged between 1 and 10 years. The 10th was created under the Natural Resources Conservation Service Wetlands Reserve Program.

Methods

We used an ecosystem level approach to assess the components and processes that are most likely to: (1) illustrate differences between the two types of wetlands and assess whether mitigation was meeting the net loss goal in terms of wetland structure (e.g., vegetation and soil characteristics) and function (primary productivity, hydrology exchanges); (2) make recommendations that might improve the success of mitigation projects; and (3) establish indicators suitable for use as performance standards (Fig. 1). Data were collected during one growing season (May–September). Complete descriptions of field and laboratory methods are described in Fennessy et al. 2004.

Results and Discussion

Hydrology

Natural and created wetlands had significant hydrological differences (Fig. 2). Created wetlands had deeper surface water levels (data not shown), greater mean depth to groundwater, and a shorter duration of water in the root zone (top 30 cm of soil profile). Surface water depths (measured at the deepest part of the wetland) averaged 29.5 cm in the natural and 43.7 cm in the mitigation wetlands. Groundwater levels were found in the root zone 55% of the time in natural sites and only 22% of the time in mitigation sites.

The high bulk density of the mitigation site soils is a likely cause of the hydrologic differences between the wetland types (see Table 1). High bulk density limits water movement through the soil, and may have contributed to the hydrological differences noted, including the relatively short duration of water in the root zone and the lack of a draw down period in the mitigation sites.

The combination of deeper surface water and lower groundwater levels in created sites has implications for plant growth, community development (water available for root uptake; successional processes), and biogeochemical processes, for example, denitrification. In this case, the relative lack of water movement limits the delivery of compounds such as nitrate and dissolved organic carbon needed by denitrifying microbes. Ultimately, improper hydrology seems to limit the development and function of the mitigation sites towards a more natural wetland condition.

Soils

Overall, soil characteristics were significantly different in the mitigation sites (Table 1). Soil organic carbon (%) was nearly five times greater ($p = 0.007$) in natural than mitigation sites, averaging 15.1+3.2 and 3.1+0.64%, respectively. Concentrations of percent soil nitrogen ($p = 0.008$) and plant available phosphorus (μgP/g soil; $p = 0.08$) were also significantly higher in the natural sites. The soil composition of

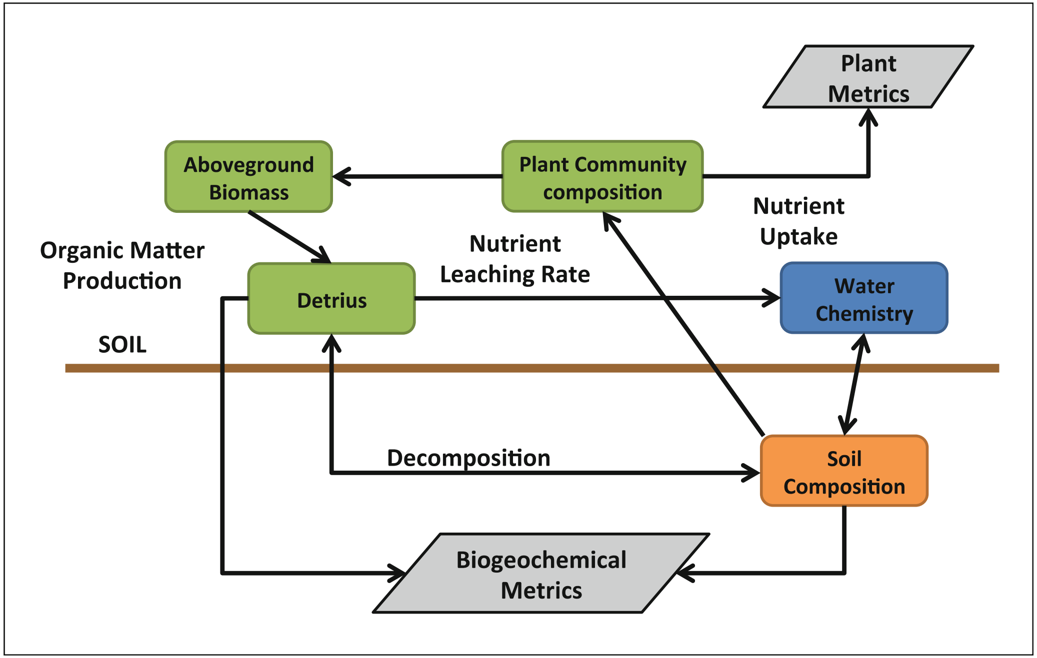

Fig. 1 Conceptual model of the ecosystem components included in this study including structural components, processes, and metrics that are empirically derived to indicate ecosystem condition (Adapted from Fennessy et al. 2004; Note: not all of the variables depicted here are detailed in this chapter)

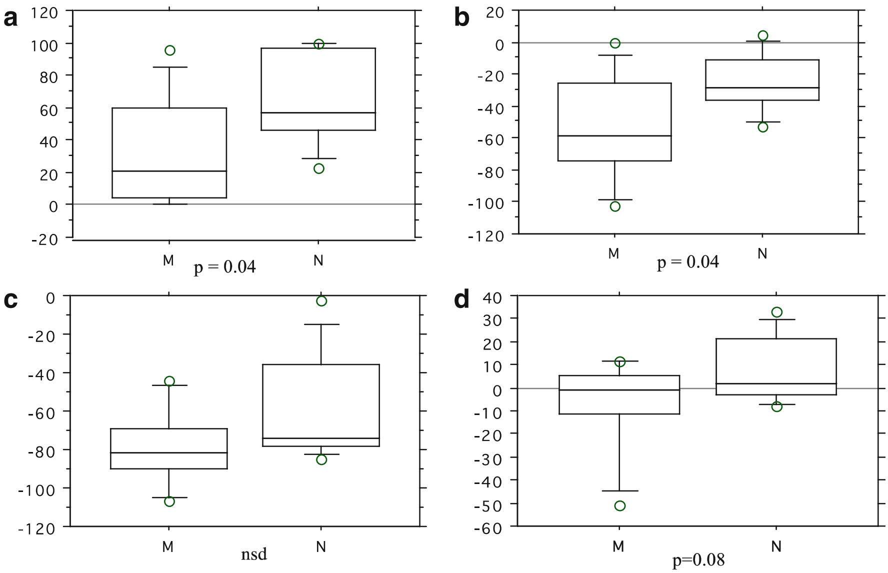

Fig. 2 Box plots for four hydrological parameters for mitigation versus natural wetlands during the growing season (May–September). Parameters are: (**a**) percent time groundwater was in the root zone (top 30 cm of soil profile), (**b**) mean depth to groundwater (cm), (**c**) mean maximum depth (cm), and (**d**) mean minimum depth to groundwater (cm). Positive values in **b**–**d** indicate water levels were above the soil surface. Means were evaluated using unpaired t-tests (p values shown)

Table 1 Summary of mean soil parameters for natural and mitigation wetlands. Values recoded are an average of five replicates in each of the nine natural sites and 10 mitigation sites (mean ± SE). P-values show results of a two-sample T-test to determine if means are significantly different (From Fennessy et al. 2004)

Soil parameter	Natural	Mitigation	p-value
pH	5.42 ± 0.17	6.29 ± 0.22	0.006
% Organic carbon	15.1 ± 3.2	3.1 ± 0.64	0.007
% Nitrogen	1.13 ± 0.24	0.26 ± 0.03	0.008
Plant available P (μg/g)	12.2 ± 1.1	7.4 ± 2.3	0.08
Bulk density (g/cm^3)	0.62 ± 0.13	1.75 ± 0.15	0.001
% Solids	42.5 ± 6.0	73.6 ± 1.5	0.001
Ammonia (ug/g)	62.4 ± 15.3	20.5 ± 3.3	0.01
P total (ug/g)	1156 ± 252	669 ± 91	0.07
Exchangeable Ca (ug Ca^{2+}/g)	10897 ± 3585	8705 ± 3700	NS
Exchangeable Mg (ug Mg^{2+}/g)	3600 ± 700	5401 ± 1602	NS
Exchangeable K (ug K^+/g)	5501 ± 1384	5155 ± 733	NS
Exchangeable Na (ug Na^+/g)	2612 ± 581	1242 ± 23	0.02

the created wetlands confirms the results of other studies that found low organic carbon content and nutrient availability likely limits the development of wetland creation projects (Bishel-Machung et al. 1996; Shaffer and Ernst 1999; Stolt et al. 2000). Mean bulk density was also significantly higher in mitigation sites (Table 1), which may reduce root growth and limit the accumulation of soil carbon. The addition of soil organic matter amendments, for example, from high organic matter compost or donor wetland soils could accelerate the development of high-quality soil, promoting the colonization of certain plant communities (Brown and Bedford 1997).

Vegetation

Plant species vary in their sensitivity to environmental stressors, making vegetation one of the most commonly used biotic assemblages for the assessment of a site's ecological condition, and as the basis to evaluate the temporal development of mitigation projects and set potential performance targets. Macrophytes respond predictably to environmental stressors such as hydrological changes, agricultural or urban land uses surrounding a site, high nutrient or sediment loads, or herbivory. The support of specific plant species provides an indication of its ability to support specific ecosystem functions and processes (Cronk and Fennessy, 2001; Fennessy et al. 2002). This is the basis for vegetation-based indicators like the Floristic Quality Assessment Index (FQAI) and the Vegetation Index of Biotic Integrity (VIBI; Mack and Micacchion 2000) that were used in this study (Fennessy et al. 2004). We also investigated plant community measures of species richness and aboveground biomass (Table 2).

The composition of the plant communities differed significantly. Mitigation sites had a higher relative abundance of tolerant (or generalist) species, fewer sensitive species, and fewer species of sedge genera such as *Carex* (Cyperaceae). FQAI scores were significantly lower ($p = 0.004$) in the mitigation (14.2) compared to the natural sites (21.6). This reflects the fact that generalist species tend to dominate in early successional or disturbed communities. The distribution of scores differed dramatically, for instance, 90% of the mitigation sites had FQAI scores of 15 or lower, while all of the natural sites scored above 15. VIBI scores ranged from 9 to 82 in the natural sites, confirming that they spanned the gradient of human disturbance. The range of scores for created wetlands was narrow, ranging from 16 to 50. This "compression"

Table 2 Summary of mean vegetation attributes for natural and mitigation wetlands. Values recoded are an average of the 9 natural sites and 10 mitigation sites (mean ± SE). P-values show results of a two-sample T-test to determine if means are significantly different (From Fennessy et al. 2004)

Vegetation attribute	Natural	Mitigation	p-value
Species richness	31.0 ± 4.4	25.6 ± 2.8	0.22
Vegetation index of biotic integrity	57.6 ± 8.1	27.2 ± 4.6	0.005
Floristic quality assessment index	21.6 ± 1.7	14.2 ± 0.7	0.004
Biomass production (g/m^2)	347 ± 35	209 ± 57	0.04

in scores is due, at least in part, to the high degree of similarity in plant community composition in the created sites where common, tolerant plant species dominated. While species composition varied, there was not a difference in species richness ($p = 0.22$).

Natural sites had significantly higher ($p = 0.04$) aboveground biomass (a measure of wetland primary productivity) with a mean of 347 g m^{-2} compared to 209 g m^{-2} in created sites. These biomass estimates are low compared to other wetland studies. For example, in a literature summary, Mitsch and Gosselink (2007) report biomass production ranging from 500 to 5500 g m^{-2}. Low productivity in the mitigation sites could be a result of environmental factors such as water depth or limited nutrient availability. Many of the mitigation wetlands had relatively deep and stable surface water levels, while many of the natural sites experienced a dry-down period in which very little or no surface water was present. Without a dry-down period during the growing season, seed germination is inhibited, potentially limiting both species diversity and biomass production (Cronk and Fennessy 2001).

Ecosystem Analysis

Experimental data from this case study shows that the mitigation wetlands differed from natural sites in terms of hydrology, soil characteristics, plant community composition, and the production of biomass. To synthesize some of these data and address the question of whether the mitigation sites are replacing the characteristics of lost wetlands, a cluster analysis was conducted to take an integrated view of these ecosystems by combining key measures of vegetation (FQAI scores) and percent soil C and N (Fig. 3). The analysis shows a clear division between natural and mitigation sites;

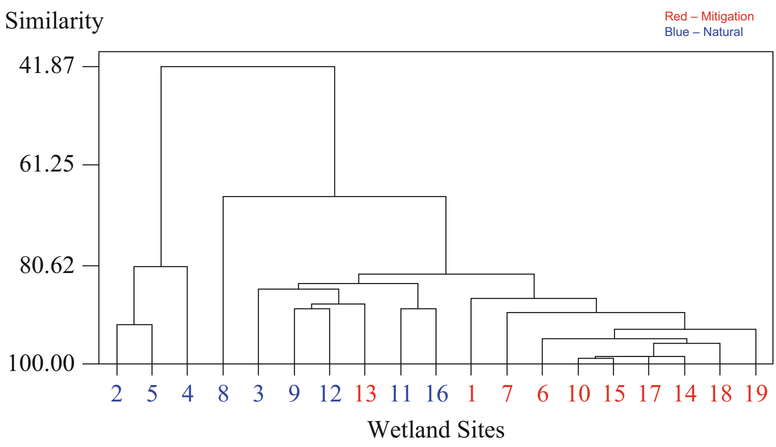

Fig. 3 Cluster analysis using the FQAI vegetation scores and soil characteristics (% total carbon and nitrogen). Natural wetlands are shown numbered in *blue* ($n = 9$) and created wetlands in *red* ($n = 10$)

mitigation wetlands differ in terms of individual components (e.g., soil, vegetation), as well as their overall ecosystem structure. Thus, the no net loss goal to replace the structure (and hence function) of wetlands has not been met by these mitigation sites.

Conclusions and Future Challenges

This case study, as well as the research of others, indicates that creating wetlands similar to natural sites (in both structure and function) takes more time than just the few years required for mitigation monitoring (typically five) and an understanding of the complexity of ecosystem components and their interactions. Based on these results, recommendations that might increase the likelihood of success in meeting the no net loss goal include:

1. Treatment of Soil During Construction: Hydric soils are the physical foundation for wetland ecosystems. They influence key ecosystem processes including microbially driven nutrient cycles, plant growth, water storage, and the composition of the biological communities (Stolt et al. 2000). Some mitigation project construction techniques negatively influence the composition and structure of wetland soils. For instance, created wetlands are often constructed by scraping off the top layer (A-horizon) of soil with heavy machinery. The A-horizon is typically where soil organic matter and plant nutrient concentrations are highest. Removing and compacting surface soils leaves them high in silts and clay materials, which can alter ecosystem development; for instance, Stolt et al. (2000) showed that excavating soil reduced soil nutrients, microrelief, and ultimately plant diversity. Many of the sites in this study were created by removing top soil and exposing the B-horizon, which was lacking in organic matter. We also found that soils did not improve with age. To overcome this problem, mitigation wetlands need more ecologically robust design and construction methods, for example, by requiring that replacement wetlands are true restoration projects (i.e., located where there are hydric soils present).
2. Wetland Hydrology: Hydrology is a key variable of wetland ecosystems, driving the development of wetland soils and the biotic communities. It plays a role in determining plant species composition, the spatial array of species, their productivity, and capacity for nutrient uptake (Cronk and Fennessy 2001). Groundwater exchange and movement through the root zone is an important component of natural systems. Recreating suitable wetland hydrology is essential to the development of mitigation wetlands.
3. Soil Nutrients: Where soil nutrient availability is low, steps should be taken to amend the soils during construction. Increasing organic matter (carbon) content has been shown to help vegetation establishment in mitigation projects. Stauffer and Brooks (1997) tested the application of leaf litter and soil organic matter to increase the performance of newly created wetlands. After 2 years, they found that leaf litter plots held an average of 8.2% organic matter, which more closely resembled the levels found in natural wetlands. While these results are

encouraging, more research needs to be done in this area. Studies investigating the effects of carbon and nitrogen amendments will aid in our understanding of the mechanisms behind C and N dynamics in mitigation wetlands.

4. The placement of a site within the landscape is a key consideration; we found the majority of the mitigation wetlands were atypical in terms of their hydrology or topographic position. A recommendation to address this is to use the HGM guidelines proposed, for example, by Brinson and Rheinhardt (1996) and Bedford (1996) when determining the location of a mitigation project.

The findings of this study have implications for the sustainability of our water resources, because the data help diagnose problems in mitigation wetland design which can then be used to set ecologically defensible performance targets. Mitigation project monitoring can be conducted using both previously developed biological indicators (e.g., FQAI, VIBI) and those based on the biogeochemical conditions (e.g., levels of soil organic carbon). The use of indicators to measure wetland ecological condition can improve monitoring of the project outcomes and help ensure their success in meeting the no net loss goal.

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No Net Loss Case Study: Wetland Banking in Chicago (USA)

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Morgan Robertson

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Abstract

Compensation for permitted losses of wetlands and streams has been approved under the Clean Water Act in the USA since the late 1970s. A form of compensation known as wetland and stream banking was developed in the 1980s and has come to dominate compensation practices in the US. While most banks were initially developed by government agencies, the large majority of banks are now private and entrepreneurial. Although banking solves many of the problems identified with other forms of compensation, it does not do away with issues of temporal and spatial losses of wetland acreage and function. Data are presented from the use of compensation banking for wetland impacts permitted between 1994 and 2002 in the Chicago District of the US Army Corps of Engineers. While "no net loss" of wetland acreage is achieved by the end of the period, wetlands are not always present at bank sites by the time of the impact for which they compensate. Furthermore, whether "no net loss" is achieved depends on the scale at which the question is posed. Because of the spatial clustering of bank sites, banking results in losses being spread across many small watersheds while gains are concentrated in a few small watersheds.

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Wetland banking · Corps of Engineers · Clean Water Act · Compensation · Mitigation

Wetland Compensation and the Emergence of Wetland Banking

Compensation for permitted losses of wetlands and streams has been approved under the Clean Water Act in the USA since the late 1970s, but until the early 1990s, almost all types of compensation took the form of individual wetland restoration sites that were developed and managed by the party seeking the wetland permit. By 2000, a litany of problems with this kind of *permittee-responsible compensation* had been documented by researchers (see NRC 2001) and was familiar to Clean Water Act regulators at the US Army Corps of Engineers and the US Environmental Protection Agency (EPA). The most serious problems included the lack of arrangements for long-term stewardship (often these small sites, once built, were deeded to neighborhood associations or to public parks districts, without monies attached for management), the physical dispersal of these sites across large areas (Corps District staff lacked the time to visit each site and confirm its existence), and the vague or absent performance standards to which such sites would be held. Kevin Erwin's landmark study of all Florida compensation sites, published in 1991, documented that only 33% of such sites were ever even constructed and only 6% were meeting the (often unsatisfactory) performance standards specified in the permit for the impact. Clearly, this situation posed a major threat to meeting the mission of "no net loss of wetlands" (NNL) that had been a campaign slogan for President Bush in 1988, and was adopted by the Corps following the 1990 Water Resources Development Act.

The development of wetland banking addressed many of these key deficiencies with permittee-responsible compensation. Banking emerged slowly through the 1980s as large state or private organizations (usually Departments of Transportation or energy-sector firms) received permission to create large areas of advance compensation for planned future impacts. By 1990, the EPA and Corps had expressed their strong enthusiasm and support for the concept as a way of meeting the NNL requirement, although it was assumed that state or local governments would be the "bankers" of wetland credits. However, by the time of the 2001 National Research Council report that outlined the many failures and concerns with wetland compensation, banking was largely a private entrepreneurial activity, and the NRC singled banking out as a potentially important way to solve the problems of compensation. The advantages of banks in this respect are that they: (a) make it easy for regulators to visit and observe the compensation for many different permitted impacts with a single trip, (b) are sponsored by individuals or organizations who must pledge firm and well-funded arrangements for long-term stewardship, and (c) are usually held to detailed and ecologically based performance standards.

The Case of Chicago

In the Chicago District of the US Army Corps of Engineers (see Fig. 1), the first wetland banking credit was transacted in April of 1994. (The data presented here were first reported in Robertson (2006) and Robertson and Hayden (2008), and summarized in Robertson (2008).)

Between 1994 and 2002, 15 banks were approved, operated by 8 different private banking firms. 199 transactions were made at these banks in compensation for 241 individual impacts. A total of 309.321 credits were purchased to compensate

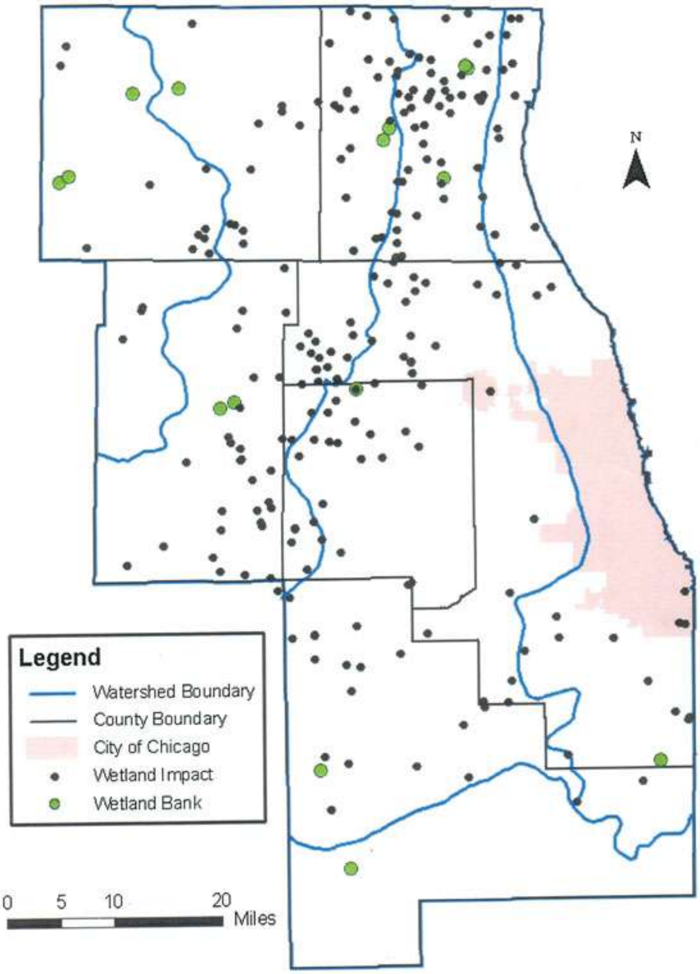

Fig. 1 The Chicago District of the US Army Corps of Engineers, showing the location of impact sites (*black*) that were compensated at wetland banks (*green*), and the watersheds within which the compensation was encouraged to stay (*blue*) (From Robertson 2006)

for 269.071 acres of impacts to wetlands; a credit was almost always defined as an acre of wetland, so this meets the basic definition of NNL: the overall replacement ratio during the time period was therefore 1.15:1. These transactions represented 31.8% of the compensation actions required by the Chicago Corps District and other regulatory agencies, but only provided compensation for 9.3% of the area impacted (Robertson 2008). Wetland bank credits were often used to compensate for smaller impacts, with an average purchase of 1.6 credits (where one credit usually equaled one acre of wetland). Did this practice assure "no net loss" of wetlands in the Chicago District? Yes and no.

While all of the credits sold at Chicago District banks eventually represented actual wetland restoration acreage, they did not always do so at the time of sale. Wetland banks in Chicago are allowed to sell a tranche of wetland credits after they have secured the site, have an approved restoration plan, have secured all necessary permits, and arranged for long-term funding of stewardship. They can sell 30% of their eventual credit total before the site has begun to be restored to a wetland condition. To require a site to meet all ecological performance standards prior to the first release of credits for sale would have forced private credit developers to wait 5 or more years before seeing any return on their investment, and would thus have ensured that only state agencies produced wetland credits. The arrangement for early sales meant that 60.3% of all transactions in Chicago occurred at banks that had not done any work on site development (see Fig. 2). Only 8.4% of transactions occurred at banks that had met all ecological performance standards. This undoubtedly represents an improvement over prior practices, as nearly all permittee-responsible compensation sites were built after the impact had been carried out and few were held to performance standards in any meaningful way. On the other hand, it represents an undeniable *temporal loss* of wetland area and functions.

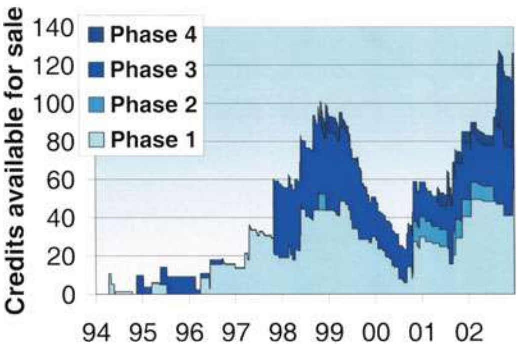

Fig. 2 The number of wetland bank credits available for sale in the Chicago District of the US Army Corps of Engineers, subdivided by the level of performance standard met: banks in Phase 1 had satisfied all administrative requirements but had not started development of the site; banks in Phase 2 had met hydrologic criteria; banks in Phase 3 had successfully implemented a vegetation planting plan; and banks in Phase 4 had met all performance criteria (Source: Robertson and Hayden 2008)

The Geography of Wetland Banking

Furthermore, there is always *geographic displacement* with wetland banking, because bank sites can be distant from the site of impact, while most nonbank compensation in the Chicago District was on the same property parcel as the impact. This creates the potential for other kinds of functional losses. The average distance between impact site and compensation site, where compensation was purchased at a bank, was 25.8 km (16.1 miles) between 1994 and 2002, but showed an increasing trend. Since compensation must be in the same Corps District as the impact, and the Chicago District is the smallest Corps District (most are much larger), this may be assumed to be a much smaller distance than the impact-compensation distance in other Districts.

Another aspect of geographic displacement involves the concentration of compensation at bank sites, resulting in a spatial clustering whereby some areas lose wetland area through banking and some gain it. The Chicago District of the Corps divided the District into five watersheds and required that compensation must occur in the same watershed as the impact (see Fig. 1). It was possible to compensate out of watershed, but the permittee would have to purchase twice as many credits: 79.7% of impacts were compensated within the defined watersheds, and each watershed (which were defined at the scale of 8-digit US Geological Survey codes) showed a clear net gain in wetland area as a result of banked compensation. However, when considered at other watershed scales, the pattern of loss and gain is different (see Fig. 3). At the scale of USGS 10-digit watersheds, 7 gained wetland acreage, 18 lost wetland acreage, and 15 were unaffected. Considering the even smaller USGS 12-digit watersheds, 10 gained wetland acreage, 63 lost wetland acreage, and 90 were unaffected. Wetland banking resulted in “no net loss” at the scale of

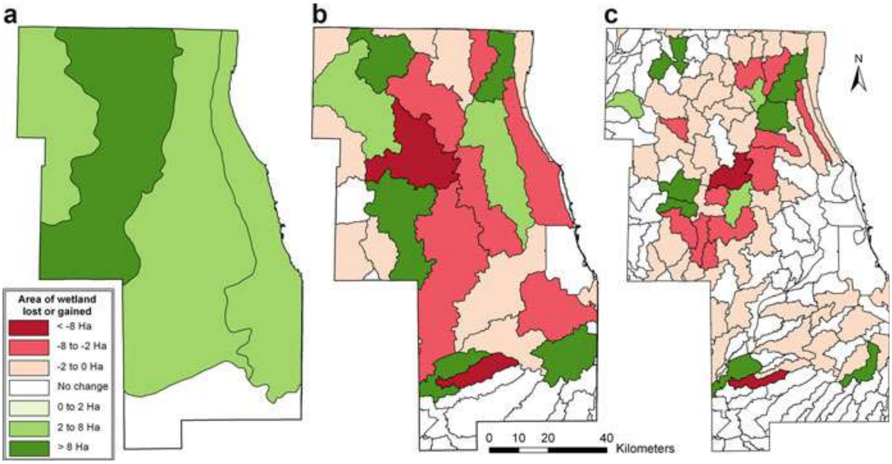

Fig. 3 Wetland acreage gain and loss in three scales of watershed in the Chicago District; (**a**) Corps-defined watersheds, (**b**) USGS 10-digit watershed classifications, (**c**) USGS 12-digit watershed classifications (From Robertson and Hayden 2008)

observation prioritized by regulators, but it created a clear geography of loss and gain when viewed at a finer geographic scale.

Challenges

Wetland banking in Chicago successfully addressed many of the obstacles to the implementation of a "no net loss" policy but did not fully address questions about temporal loss and the geography of loss and gain. It is clear that wetland banking involves the spatial and temporal displacement of wetlands; while these displacements may be more effectively monitored and managed than they are in other types of compensation, they are not well understood.

Other challenges (common to all types of compensation) include whether the assessments used to approve bank sites are appropriate to ensure that wetland functions are not lost, even as area increases. Many wetland assessment techniques measure structural features of hydrology and vegetation as imperfect proxies for the valued wetland functions that ensure a site's long-term viability.

Wetland banking has continued to grow rapidly as a compensation practice and by 2005 accounted for at least one third of all types of compensation. The release of the 2008 Corps and EPA "Compensation Rule" appears to have stimulated the wetland banking industry, and its preamble strongly highlighted the advantages of wetland banking in achieving no net loss. One consequence of this is the presence of a relatively vocal private-sector industry with unusual positions on environmental policy. The banking industry tends to lobby the US government to promote *increased* enforcement of the Clean Water Act (because increased enforcement drives increased business for them) and *higher* standards for compensation (because high standards prevent low-cost wetlands credits from flooding the market and sullying the reputation of their business). On the other hand, the continued viability of their business also relies on the continued issuance of permits to impact wetlands: permits which many environmentalists believe should be denied. The advantages and deficiencies observed in the Chicago case study remain with banking today.

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Regulation of Activities for Wetland Conservation and Management: Overview

111

Mark Everard

Contents

Abstract

However strong the scientific and ethical case for the conservation of wetlands and their associated biodiversity and capacities to support human livelihoods, practical conservation and management depends upon a number of drivers. In this regard, "regulation" has a far broader meaning than purely the enactment of legislation. Indeed, statutory obligations may be poorly observed or poorly enforced, or may be practically unenforceable, in many territories, particularly in the developing world. The wider formal and informal policy environment, referred to collectively as societal levers, comprises statutory legislation, common/civil law, markets, market-based instruments, and both formal and informal protocols evolved as a fragmented policy environment of incentives and constraints influencing the freedoms of and rewards to resource owners and managers. Harmonisation of these societal levers with higher level sustainable aspirations is necessary to influence landowner or land manager decision-making at a local scale.

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Introduction

However strong the scientific and ethical case for the conservation of wetlands and their associated biodiversity and capacities to support human livelihoods, practical conservation and management depends upon a number of drivers. In this regard, "regulation" has a far broader meaning than purely the enactment of legislation. Indeed, statutory obligations may be poorly observed or poorly enforced, or may be practically unenforceable, in many territories, particularly in the developing world.

It is for this reason that this overview takes a broader view of "regulation" to encompass the diversity of "societal levers" identified by Everard et al. (2014) to connect higher level sustainable aspirations with practical influences on landowner or land manager decision-making at a local scale. These "societal levers" include statutory legislation, common/civil law, markets, market-based instruments, and both formal and informal protocols, which have evolved as a fragmented policy environment of incentives and constraints influencing the freedoms of and rewards to resource owners and managers.

The following overview considers these broad categories of "societal levers" in turn. Far more detail on individual "levers" is provided in other chapters in this Wetland Book.

Statutory Legislation

Statutory legislation and associated regulatory obligations represent formalized "rules" agreed by society. Some protect the rights of resource owners, but many act to constrain actions that infringe the freedoms of other sectors of society. These agreements may be supranational (such as EU Directives), national (Acts of Parliament and subsidiary Regulations), or local (for example, by-laws).

Some statutory protections have yielded significant successes for ecosystems and selected services, for example, through various wildlife, water resource, air quality, and landscape protection legislation. Specific pieces of legislation directly affecting wetlands in a European context include the EU Habitats Directive, Birds Directive and, where wetlands are explicitly recognized as constituting at least part of "Water Bodies," the Water Framework Directive. (For more details see ▶ Chap. 59, "Transnational and Regional Legal Frameworks.")

However, the primary focus of this overview is on regulation of activities affecting wetland conservation and management, rather than those centered on the wetlands themselves. As wetlands are part of mosaic natural and human landscapes, regulation of development activities – urban, infrastructure, industrial, and land use – have a significant potential to affect wetlands both directly and more remotely. These potential impacts need therefore to be addressed in proposals for development before impacts occur, enabling their avoidance, reduction, or mitigation (see ▶ Chap. 115, "Avoid-Mitigate-Compensate Sequence: Wetland Conservation").

For this purpose, and associated with various strands of legislation, various regulatory tools and obligations are imposed on development proponents. (Examples include ▶ Chaps. 112, "Environmental Impact Assessments," ▶ 113, "Strategic Environmental Assessments," ▶ 114, "Permit Schemes," ▶ 117, "Compensation in Wetlands," ▶ 118, "Mitigation Banking for Wetlands," ▶ 130, "In-Lieu Fees in Wetlands," ▶ 119, "Enforcement: Wetlands," and ▶ 135, "Property Rights.")

Common Law

English common law is founded on the protection of rights, evolving since Roman times through a less formalized body of case law to uphold the rights of individuals or communities potentially infringed by the actions of others. The system of English common law is replicated in many countries across the world, although it cannot be assumed to operate identically in all jurisdictions. For example, Scotland has a civil law, rather than a common law. Nevertheless, the broad principle of evolving case law based on judgments about the rights of different people lies at its core.

Case law relating to rights to air, water, soil, and enjoyment of a natural environment of undiminished quality as well as sporting or access rights demonstrates common law protection progressively extending to some of what we now term ecosystem services, including those provided by wetlands, such as protection of the quality and enjoyment of recreational fisheries (Carty and Payne 1998). Representing as they do the multiple ways in which different sectors of society benefit from ecosystems, ecosystem services represent a wider framework of public rights, many of them historically omitted from management considerations, to which common law protection may be extended by case law (Everard and Appleby 2008; Everard et al. 2012).

As such, beneficiaries of the ecosystem services provided by wetlands – be they provisioning, regulatory, cultural, and/or supporting services – may have a case in law to seek an injunction on activities likely to damage a wetland resource and its beneficial services or else to sue for compensation for any damages that may have occurred. Sporting rights that can be tied down to damage to "property" have been the source of rich case law over centuries (Carty and Payne 1998). English manorial court records from the postmedieval period also provide a rich source of case law relating to resource conflicts, particularly with respect to water with the spread of the once-dominant water meadow system in the catchments of Wessex rivers which spurred conflicts between water meadow operators, mills, navigation and fishery interests, and other users of river flows (Bettey 1999). Property rights associated with wetlands can thus constitute levers for their protection as well as for their exploitation (see ► Chap. 135, "Property Rights").

Markets

Markets exert a significant, indeed sometimes an overwhelmingly dominant, influence over choices about the management of natural resources. Markets most commonly favor provisioning services but generally fail to recognize that their production is heavily dependent on the underpinning support of a wide range of additional ecosystem services (Power 2010).

Some market failures damaging water, wetland, and other environmental resources and associated human wellbeing are beginning to be addressed, for example, the recent evolution of carbon markets and the institution in the UK of an Aggregates Levy on mined substances. However, most ecosystem services remain external to current markets, and their value to society is therefore inadequately incorporated into policy and business calculations. Agriculture has been the foremost pressure leading to the degradation of wetlands habitat worldwide, largely driven by consumer pressure (Millennium Ecosystem Assessment 2005) reinforced by governments through a food security agenda (Everard 2011) as well as favoring short-term economic growth over long-term consequences. Securing adequate food and maintaining economic growth for a growing population are pressing and legitimate priorities for governments and individuals, but exploitation of ecosystems at the expense of longer-term and wider societal needs not only conflicts with stated commitments to sustainable development but also represents short-termism, a substantial market failure, and the consequent creation of multiple vulnerabilities.

Were markets to evolve to recognize more of the multiple values provided by wetland and other natural systems, these foundational resources would be greater valued and protected. Examples of this occurring include the shift in flood control globally from a paradigm of local "flood defense" of assets at risk to a more systemic consideration of flood risk management by considering flows of water through whole connected catchment wetland systems and increasing use of "natural flood management" techniques (Everard et al. 2009).

Market-Based Instruments

Various market-based instruments have also been developed to promote aspects of societal wellbeing supported by environmental resources and processes. These are commonly related to statutory agreements, including both inducements (such as agri-environment subsidies) and levies to constrain exploitation or generate environmentally compensating projects (such as the UK Landfill Tax and Aggregates Levy).

Successive iterations of the EU Common Agricultural Policy (CAP) represent market interventions that have shifted from a primary focus on output support, from a time when food scarcity drove policy, through to broader social and environmental goals. The EU CAP has had a dominant influence on management of much of the European rural landscape and hence on the balance of ecosystem services that are either favored or eroded. Given the dominant influence of agriculture on the global wetland resource to date (Millennium Ecosystem Assessment 2005), future agriculture and food policy and internalization of their potential harm to wetlands and the human interests that stem from them will inevitably have profound implications for the balance of ecosystem service production.

Voluntary payment mechanisms such as "payments for ecosystem services" (PES) schemes can also specifically secure the supply of valued ecosystem services wherein payments from "consumers," or "buyers," of ecosystem services are accepted by those "producing" those services (Wunder 2005). A water utility, for example, benefitting from higher quality river water requiring less "clean up" costs, may pay farmers, typically via an intermediary body, for undertaking water-sensitive farming practices beyond those required by statute (Smith et al. 2013). The OECD estimated that there were already more than 300 PES or "PES-like" schemes in operation globally by 2010, addressing a diversity of services ranging from water supply to carbon sequestration, conservation of biodiversity, amenity and recreational opportunity, and ranging from global to local scales (OECD 2010). Interest in PES has since increased substantially, Schomers and Matzdorf (2013) identifying 457 published peer-reviewed papers on PES addressing a small subset of many more schemes around the world. (See Economic Incentives to Conserve/Manage Wetlands [Non-regulatory]).

Formal and Informal Protocols

In addition to legal and market mechanisms, a range of protocols to safeguard selected desirable services has been established. Among the wide variety of protocols influencing societal choices, the 1971 Convention on Wetlands (the "Ramsar Convention": www.ramsar.org) represents a pioneer of the Ecosystem Approach, explicitly acknowledging the global resource of wetland ecosystems not merely as ecologically important but also as central to the livelihoods and future socioeconomic prospects of people, therefore requiring "Wise Use" (since acknowledged as synonymous with "sustainable development" and mirroring the Ecosystem Approach).

Other relevant international protocols affecting management activities that may impinge upon wetlands include the United Nations Framework Convention on Climate Change (UNFCCC: www.unfccc.int) and Convention on Trade in Endangered Species of Wild Fauna and Flora (CITES: www.cites.org).

At national scale, the target to achieve 25% forest cover in the Scotland Land Use Strategy (Scottish Government 2011) is a national-scale aspiration that is not statutory and is at present supported only by a fragmented set of national economic incentives delivered under the Rural Development Program (Everard et al. 2014). Policies established by major landowning or otherwise influential organizations, such as the National Trust in England, in Wales and in Scotland, the Royal Society for the Protection of Birds (RSPB) or the National Farmers' Union, may also be regarded as forms of nonstatutory but nevertheless significant protocols establishing governing policies influencing the management of wetlands and other natural resources and the production of ecosystem services.

Boundaries Between "Societal Levers"

The boundaries between statutory, common law, market and market-based, and both formal and informal protocol levers are not fixed, with one measure potentially transforming into another. For example, issues of concern addressed by common case law may evolve into statutory legislation as in the case of many contemporary laws relating to the air and atmosphere as well as water systems (Everard et al. 2012), which may in turn instigate market preferences, inducements, or disincentives.

Conclusions and Challenges

One of the strengths of the often organically evolving nexus of "societal levers" influencing resource owners and managers is that it produces the heterogeneous mosaic landscapes familiar around the world, reflecting both natural conditions and landowner aspirations. Collectively, these societal levers safeguard or otherwise influence some socially valuable ecosystem services, including those produced by and affecting wetlands.

However, Everard et al. (2014) also conclude that the present policy environment is neither sufficient nor sufficiently integrated to achieve coherence between the choices of resource owners and wider societal aspirations for ecosystem service provision. The fragmented manner and service-by-service basis on which these various levers have evolved, often as a reaction to localized and acute issues of concern, does not automatically moderate the freedom of choice of landowners, often in localized geographical contexts, to favor the production of service outcomes of optimal benefit to all in society at catchment and other landscape scales (Everard et al. 2014). Indeed, the plethora of competing incentives, from CAP to habitat-enhancing support, may make it difficult for landowners to identify what is best for the common good, however defined, and to allow them to choose practices serving this common good while at the same time reaping individual benefits. The net consequence is that the integrity and vitality of wetland systems that may be affected adversely both directly and indirectly may not be seen as a priority.

Further evolution of "societal levers" – the diversity of statutory and informal influences on practical management decision-making – is therefore a necessity to bring about greater coherence from high-level (international and national) down to local-scale protection of wetland systems and their associated ecosystem services.

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Environmental Impact Assessments 112

Mark Everard

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Abstract

Environmental impact assessment (EIA) is a formal process used to predict environmental consequences, which may be positive or negative, of a plan, policy, program, or project prior to approval. The International Association for Impact Assessment (IAIA) defines an environmental impact assessment as "*. . .the process of identifying, predicting, evaluating and mitigating the biophysical, social, and other relevant effects of development proposals prior to major decisions being taken and commitments made.*" EIAs should contain proposals for scheme design or operation to reduce these impacts to acceptable levels or to address different technical solutions. "Acceptable levels" is a contested and political concept, though the purpose of an EIA is to inform better decision-making by bringing wider environmental concerns into consideration alongside economic, social, and political concerns.

Keywords

Environmental impact statement · Environmental assessment · Biodiversity · Vulnerabilities · Cumulative imact · Buffer zone

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Introduction

Environmental impact assessment (EIA) is a formal process used to predict environmental consequences, which may be positive or negative, of a plan, policy, programme, or project prior to approval. The International Association for Impact Assessment (IAIA) defines an environmental impact assessment as "*. . .the process of identifying, predicting, evaluating and mitigating the biophysical, social, and other relevant effects of development proposals prior to major decisions being taken and commitments made*" (International Association for Impact Assessment 1999).

EIAs should contain proposals for scheme design or operation to reduce these impacts to acceptable levels or to address different technical solutions. Clearly, "acceptable levels" is a contested and political concept, though the purpose of an EIA is to inform better decision-making by bringing wider environmental concerns into consideration alongside economic, social, and political concerns. Each EIA is unique, not requiring adherence to a predetermined environmental outcome but rather requiring decision-makers to take account of environmental values in their decisions. This then, at least in theory, drives a more sustainable and transparent process of development.

The Origins and Development of EIAs

Development of the EIA process commenced in the 1960s, spurred on by growing environmental awareness. In the USA, environmental impact assessments obtained formal status with enactment of the National Environmental Policy Act (NEPA) in 1969 (Department of Environment n.d.; Clark and Canter 1997). EIAs are effectively a more streamlined version of the more detailed environmental impact statements (EISs), also brought into law in the USA under the NEPA, to provide more detail about actions "*. . .significantly affecting the quality of the human environment.*" EIAs are more commonly referred to in the USA as environmental assessment (EA), a public document comprising an environmental analysis prepared to determine whether a federal action would significantly affect the environment. If the ES finds that it might have significant impact, a more detailed environmental impact statement (EIS) is required. If not, a *Finding of No Significant Impact* (FONSI) is issued. This process is administered in the USA by the Council on Environmental Quality (CEQ).

EIAs have since been increasingly deployed throughout the world as a rapid means to assess the scale of potential impacts of schemes before development occurs. For example, in Australia, EIA procedures were introduced at a state level prior to that of the Commonwealth (Federal). The pioneering Australian state was New South Wales, in which the State Pollution Control Commission issued EIA guidelines in 1974. This was followed by the passing in 1974 at Commonwealth (Federal) level of the Environment Protection (Impact of Proposals) Act, superseded by the Environment Protection and Biodiversity Conservation Act 1999 (EPBC), each promoting the EIA process.

In Canada, the EIA process entered the mainstream of environmental management through common law challenges, ultimately formalized in the Canadian Environmental Assessment Act 2012 which describes EA as a planning tool to identify, understand, assess, and mitigate, where possible, the environmental effects of a project.

In China, the Environmental Impact Assessment Law (EIA Law) requires that an environmental impact assessment be completed prior to project construction. However, the only penalty in the event that a developer ignores this requirement before implementing a project without first submitting an environmental impact statement is that the Environmental Protection Bureau (EPB) may require the developer to produce a retrospective environmental assessment. Only if the developer fails to do this within a designated time is the EPB then authorized to fine the developer up to a relatively low capped amount. This lack of stringent enforcement mechanisms results in a significant proportion of projects disregarding the EIA requirement (Wang 2007).

In the EU, the Environmental Impact Assessment (EIA) Directive has been in force for 25 years and is currently being revised (EU 2014). Other nations and jurisdictions requiring EIAs include Egypt, Hong Kong, India, Malaysia, Nepal, New Zealand, the Russian Federation, and Sri Lanka.

Application of EIA

A wide range of EIA methods is available across these territories to implement the EIA process to a range of applications. On project completion, an audit is generally required to evaluate the accuracy of the EIA by comparing actual to predicted impacts, including both scientific assessment of predictions and errors as well as the success of mitigation measures.

However, the EIA process is not without its critics. For example, EIA is often regarded as a decision-aiding tool rather than a decision-making tool (Jay et al. 2007). Conversely, EIAs are often seen by development proponents as being costly and hampering development. There is also generally a lack of clear definition of the spatial bounding of impacts, resulting in almost all EIAs addressing only direct and immediate on-site effects rather than systemic impacts (Lenzen 2003).

EIA and Wetlands

The Ramsar Convention Secretariat produced a Handbook on *Impact assessment* (Ramsar Convention Secretariat 2010) in response to recognition that the concepts of environmental impact assessment (EIA) and strategic environmental assessment (SEA) were increasingly seen as necessary components of international environmental policy and law. Despite a clear role for impact assessment being spelt out in the texts of a range of conventions, detailed implications for wetlands within both EIA and SEA were lacking despite their evident importance.

Consequently, the Convention on Biological Diversity's *voluntary guidelines on biodiversity-inclusive environmental impact assessment* and also the CBD's *guidance on biodiversity-inclusive strategic environmental assessment* were adopted by the Ramsar Commission in 2008. This reaffirmed the role of impact assessment (environmental, strategic, and social) and economic valuation as key tools for assisting the contracting parties in their efforts to achieve the objectives of the convention, calling upon contracting parties to the Ramsar Convention "*...to reinforce and strengthen their efforts to ensure that any projects, plans, programmes and policies with the potential to alter the ecological character of wetlands in the Ramsar List, or impact negatively on other wetlands within their territories, are subjected to rigorous impact assessment procedures and to formalise such procedures under policy, legal, institutional and organizational arrangements.*" The ensuing guidelines for incorporating biodiversity-related issues into EIA and SEA are essentially voluntary in nature.

The CBD voluntary guidelines on biodiversity-inclusive environmental impact assessment, outlined in the Ramsar Convention Secretariat (2010) handbook, specify stages in the EIA process and how to address biodiversity issues at each of them. These are:

- *Screening* determines which projects or developments require a full or partial impact assessment study. Screening criteria have to include biodiversity measures if proposals with potentially significant impacts on biodiversity are to be screened out. However, legal requirements for EIA may not guarantee that biodiversity will be taken into account, so consideration should therefore be given to incorporating biodiversity criteria into existing, or developing new, screening criteria. The outcome of the screening process is a *screening decision*.
- *Scoping* to identify which potential impacts are relevant to assess (based on legislative requirements, international conventions, expert knowledge, and public involvement), to identify alternative solutions that avoid, mitigate, or compensate adverse impacts on biodiversity (including the option of not proceeding with the development, finding alternative designs or sites which avoid the impacts, incorporating safeguards in the design of the project, or providing compensation for adverse impacts), and finally to derive terms of reference for the impact assessment.
- *Assessment and evaluation of impacts and development of alternatives*, to predict and identify the likely environmental impacts of a proposed project or development, including the detailed elaboration of alternatives.
- *Reporting*: the environmental impact statement (EIS) or EIA report, including an environmental management plan (EMP) and a nontechnical summary for the general audience.
- *Review* of the environmental impact statement, based on the terms of reference (scoping) and public (including authority) participation.
- *Decision-making* on whether to approve the project or not and under what conditions.

- *Monitoring*, *compliance*, *enforcement*, *and environmental auditing*, comprising monitoring of whether the predicted impacts and proposed mitigation measures occur as defined in the EMP, verification of compliance by the proponent with the EMP, and post-project auditing to ensure that unpredicted impacts or failed mitigation measures are identified and addressed in a timely fashion.

Pertinent questions from a biodiversity perspective should be raised at all stages of EIA development. These should include whether the intended activity may:

- Affect the biophysical environment directly or indirectly in such a manner or cause such biological changes that it will increase risks of extinction of genotypes, cultivars, varieties, populations of species, or the chance of loss of habitats or ecosystems
- Surpass the maximum sustainable yield, the carrying capacity of a habitat/ecosystem, or the maximum allowable disturbance level of a resource, population, or ecosystem, taking into account the full spectrum of values of that it provides
- Result in changes to the access to and/or rights over biological resources

To facilitate the development of screening criteria, the Ramsar Convention Secretariat (2010) handbook highlights the importance of exploring biodiversity at the levels of ecosystem diversity, species diversity, and genetic diversity.

As developments can affect both the structure of wetlands and the ways in which they function, incorporation of wetland impacts in EIA processes is very important. All likely impacts arising from the type and scale of development will impact need to be assessing, whether within the wetland or in the catchment area surrounding it. Examples of the kinds of wider consideration of impacts to be considered in an EIA include:

- Does the physical structure of the development affect the wetland directly, for example by drainage or permanent alteration, or indirectly by impinging on the wetland or changing the hydrology of the catchment that serves it?
- Might fertilisers, other nutrients, or pollutants in run-off from land cause ecological change in the wetland?
- Can management inputs, including for example pesticides or herbicides, run into the wetland with likely negative impacts for flora, fauna, and water quality?
- Can disturbance from increased human activity in or close to the wetland affect the ecosystem and its functioning?

The EIA should therefore:

- Delineate the wetland.
- Assess wetland biodiversity and its value and vulnerabilities.
- Address the broader potential direct and indirect impacts of the proposed development on the wetland, together with any requisite mitigation measures.

- Consider cumulative impact on the broader catchment or the effects on the entire wetland environment.
- Identify buffer areas around wetlands to protect them from impacts from the proposed development.

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Strategic Environmental Assessments

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Mark Everard

Contents

Abstract

Strategic impact assessment (SEA) is a systematic decision support process, aimed generally at the programmatic level rather than at project of individual scheme level. (The *Environmental impact assessment* process is generally applied at more localized scheme level.) The purpose of SEA is to ensure that environmental and possibly other sustainability aspects are considered effectively in policy, plan, and program making.

Keywords

Strategic environmental assessment · Environmental impact assessment · OECD · New Zealand · Environmental impact statement · environmental impact plan

Introduction

Strategic Impact Assessment (SEA) is a systematic decision support process, aimed generally at the programmatic level rather than at project of individual scheme level (The Environmental Impact Assessment process is generally applied at more

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localized scheme level.). The purpose of SEA is to ensure that environmental and possibly other sustainability aspects are considered effectively in policy, plan, and program making.

Background to SEA

Although the Environmental Impact Assessment (EIA) process assesses scheme level proposals for development, for example as specified in the European Union EIA Directive (85/337/EEC) currently under revision (EU 2014), EIA only applies to certain projects and specific effects at the local level. EIA alone is therefore deficient as deficient. This is due to the fact that many environmentally damaging decisions will have already been made at a more strategic level. As one example, an EIA on a local road scheme may overlook the fact that new transport infrastructure may generate an increased demand for travel. Consequently, the concept of Strategic Impact Assessment (SEA) originated in regional development and land use planning decision-making in the developed world.

The *U.S. Housing and Urban Development Department* published in 1981 an *Area-wide Impact Assessment Guidebook* which contained many of the principles now enshrined in the SEA process. Meanwhile, in Europe, the UNECE (1991) *Convention on Environmental Impact Assessment in a Transboundary Context* (the Espoo Convention) established foundations for the introduction of SEA in 1991. Subsequently, the European SEA Directive 2001/42/EC (EU 2001) required that all Member States of the European Union should have ratified and transposed the Directive into national legislation by 2004. SEA has subsequently been implemented in EU Member States since that time.

SEA is also applied in other countries around the world. For example, in New Zealand, SEA constitutes part of an integrated planning and assessment process under the Resource Management Act, which has the principal objective of sustainable management. The OECD (Organization for Economic Cooperation and Development) is developing guidance on SEA, which may result in its more widespread and consistent adoption.

How SEA Works

Fischer (2007) describes SEA as a process of:

- A structured, rigorous, participative, open, and transparent environmental impact assessment (EIA)-based process, applied particularly to plans and programs, prepared by public planning authorities and at times private bodies
- A participative, open and transparent, possibly non-EIA-based process, applied in a more flexible manner to policies, prepared by public planning authorities and at times private bodies

- A flexible non-EIA-based process applied to legislative proposals and other policies, plans, and programs (PPP) in political/cabinet decision-making

Effective SEA operates as a tiered decision framework. Under the EU SEA Directive, SEA is based on the following phases:

- Screening: investigation of whether the plan or program falls under the SEA legislation
- Scoping: defining the boundaries of investigation, assessment, and assumptions required
- Documentation of the state of the environment: constituting a baseline on which to base judgments
- Determination of the likely environmental impacts of development usually in terms of Direction of Change rather than firm figures
- Informing and consulting the public
- Influencing "decision taking" based on the assessment
- Monitoring of the effects of plans and programs after their implementation

The EU SEA Directive also includes other impacts besides environmental considerations, such as material assets and archaeological sites. In most western European states, this has been broadened further to include economic and social aspects of sustainability.

SEA and Wetlands

The Ramsar Convention Secretariat produced a Handbook on *Impact assessment* (Ramsar Convention Secretariat 2010) in response to recognition that the concepts of environmental impact assessment (EIA) and strategic environmental assessment (SEA) were increasingly seen as necessary components of international environmental policy and law. Despite a clear role for impact assessment being spelt out in the texts of a range of conventions, detailed implications for wetlands within both EIA and SEA were lacking despite their evident importance.

Consequently, the Convention on Biological Diversity's *voluntary guidelines on biodiversity-inclusive environmental impact assessment* and also the CBD's *guidance on biodiversity-inclusive strategic environmental assessment* were adopted by the Ramsar Commission (2008). This reaffirmed the role of impact assessment (environmental, strategic, and social) and economic valuation as key tools for assisting the Contracting Parties in their efforts to achieve the objectives of the Convention, calling upon Contracting Parties to the Ramsar Convention "…to reinforce and strengthen their efforts to ensure that any projects, plans, programmes and policies with the potential to alter the ecological character of wetlands in the Ramsar List, or impact negatively on other wetlands within their territories, are subjected to rigorous impact assessment procedures and to formalise such procedures under policy, legal, institutional and organizational arrangements." The ensuing guidelines for incorporating biodiversity-related issues into EIA and SEA are essentially voluntary in nature.

The CBD voluntary guidelines on biodiversity-inclusive environmental impact assessment, outlined in the Ramsar Convention Secretariat (2010) handbook, specify stages in both SEA and EIA processes and how to address biodiversity issues at each of them. These are:

- *Screening* determines which projects or developments require a full or partial impact assessment study. Screening criteria have to include biodiversity measures if proposals with potentially significant impacts on biodiversity are to be screened out. However, legal requirements for EIA may not guarantee that biodiversity will be taken into account, so consideration should therefore be given to incorporating biodiversity criteria into existing, or developing new, screening criteria. The outcome of the screening process is a *screening decision*.
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Pertinent questions from a biodiversity perspective should be raised at all stages of SEA development. These should include whether the intended activity may:

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- Surpass the maximum sustainable yield, the carrying capacity of a habitat/ecosystem, or the maximum allowable disturbance level of a resource, population, or ecosystem, taking into account the full spectrum of values that it provides
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Permit Schemes

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Marla Nelson

Contents

Abstract

Despite international recognition through various treaties, wetlands remain one of the most undervalued and least understood of all global natural resources. One reason for this is the inevitable conflict between wetland preservation measures and the landowner's freedom of choice in developing property. Permit-based regulatory systems that are designed to protect wetlands generally require permission from the government before conducting activities such as filling or otherwise altering the wetland. Permit-based legal mechanisms throughout the world designed to facilitate wetland management and the wise use of wetlands are reviewed. This includes an overview of international law calling for regulation and conservation of wetlands, then focuses on the United States' permit program and regulatory programs in several other countries. Opportunities for market-based incentives through mitigation banking are considered.

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Keywords
United States · Freedom of choice · Legal mechanisms · Market-based incentives · Mitigation banking · No net loss · Dredge and fill · Clean Water Act

Introduction

Despite international recognition through various treaties, wetlands remain one of the most undervalued and least understood of all global natural resources (Fiore 1993). In the United States, 82% of the nation's assessed wetlands acreage are impaired in at least one respect (Adler 2013). One reason is the inevitable conflict between wetland preservation measures and the landowner's freedom of choice in developing property. Permit-based regulatory systems that are designed to protect wetlands generally require permission from the government before conducting activities such as filling or otherwise altering the wetland.

This entry reviews the current permit-based legal mechanisms throughout the world designed to facilitate wetland management and the wise use of wetlands. The entry first explores international law calling for regulation and conservation of wetlands. It then outlines the United States' permit program and highlights regulatory programs in several other countries. Finally, the entry identifies opportunities for market-based incentives through mitigation banking.

International Laws

Numerous international laws seek to protect wetlands directly or indirectly as habitat for imperiled species. For example, the Convention on Wetlands of International Importance Especially as Waterfowl Habitat, known as the Ramsar Convention, creates an international legal framework for the protection and wise use of wetlands (Kim 2011). Each party to the Ramsar Convention must designate at least one site to include on the List of Wetlands of International Importance and promote conservation of listed wetlands. Even broader than the Ramsar Convention, the Convention on Biological Diversity takes an ecosystem perspective to encourage the conservation of biological diversity and sustainable use (Gardner 2003). The Convention on the Conservation of Migratory Species of Wild Animals, known as the Bonn Convention, not only focuses on conserving migratory species but also protects wetland function or value where the wetlands provide habitat for migratory species (Gardner 2003). Canada, the United States, and Mexico are parties to the North American Waterfowl Management Plan, which also promotes wetland conservation and restoration due to its focus on protecting wetland-dependent species (Gardner 2003). Despite these agreements, a few contain binding legal obligations and thus the extent of implementation varies considerably.

Domestic Permit Schemes

Many countries, including leading wetland conservation countries, do not have legislation or policies that exist specifically to protect wetland (Kim 2011). This includes Denmark, Germany, and the Netherlands. The United States applies a policy goal of "no net loss" of wetland functions and values. Countries with a similar "no net loss" policy include Canada, the European Union, and Trinidad and Tobago (Gardner 2003). Countries like Australia, Colombia, Costa Rica, Ghana, New Zealand, and Uganda focus instead on wetlands restoration (Gardner 2003).

United States

The United States has one of the most extensive permit schemes, regulating both public and privately owned wetlands (Gardner 2003). As development of land in the United States continues, less desirable land has become the focus of construction projects (Braddock 2007). It is the official US policy to allow wetlands to be filled pursuant to the conditions of a dredge and fill permit (Adler 2013).

Section 404 of the Clean Water Act is the most significant piece of federal legislation that directly regulates activities affecting wetlands (Fiore 1993). Pursuant to its authority under the Commerce Clause of the United States Constitution, Congress enacted the Clean Water Act with a goal "to restore and maintain the chemical, physical, and biological integrity of waters of the United States" (33 U.S.C. § 1251 et seq.)

The Clean Water Act prohibits the discharge of a pollutant from a point source to waters of the United States, unless pursuant to a permit. Most activity that places solid matter in a wetland will require a dredge and fill permit, also known as a "section 404" permit (Braddock 2007). Each element is necessary to trigger the permit requirement: a discharge, a pollutant, a point source, and a water of the United States. Not surprisingly, many of these elements have been the subject of controversy and litigation (Braddock 2007). Defining the term "waters of the United States" has been especially troublesome (Adler 2013). Notwithstanding the triggering elements, certain activities are exempt, including agriculture and silviculture activities (Braddock 2007).

The United States Army Corps of Engineers is the lead agency under section 404, and may authorize dredge and fill activities pursuant to a General Permit, Nationwide Permit, or individual permit (Braddock 2007). The Environmental Protection Agency retains oversight by writing the guidelines for issuance of a section 404 permit and retaining the authority to veto any permit issued by the Corps (Adler 2013). The Corps may issue General and Nationwide Permits on a state, regional, or nationwide basis for activities that are similar in nature and impact, and cause only minimal cumulative adverse effects. General and Nationwide Permits provide flexibility so that many routine activities can occur with little delay or paperwork (Strand and Rothschild 2009). Critics have raised concerns about the total wetland loss attributable this streamlined permit process (Strand and Rothschild 2009).

Individual permits provide a more comprehensive review of a specific proposal to fill or dredge a wetland (Strand and Rothschild 2009). The Corps may issue an individual section 404 permit if it determines, pursuant to US Environmental Protection Agency's (EPA's) guidelines, that the proposed activity is the least environmentally damaging practicable alternative and the discharge will not have significant adverse environmental consequences. Other federal statutes may also apply to the activity, including the Coastal Zone Management Act, the Endangered Species Act, and the National Environmental Policy Act.

Noncompliance can result in civil and criminal liability (Braddock 2007). The Corps, EPA, or the United States Department of Justice may enforce the Clean Water Act (Braddock 2007). Citizens may also enforce unpermitted discharges. Judicial review is available for permit denials or penalties assessed (Fowler 2003).

States play an important role in the permit process. States may displace the Corps' authority if the state's permit program meets statutory requirements, although only Michigan and New Jersey have done so (Fiore 1993). The Clean Water Act also allows states to maintain concurrent jurisdiction under their own state laws. Finally, the Corps may not issue a section 404 permit if the affected state denies water quality certification (Fiore 1993).

Other Permit Schemes

Permit schemes in other countries tend to be narrower in application. For example, the European Union requires participating countries protect of ecosystems, including wetlands, but only at designated Natura 2000 sites (Gardner 2003).

Federal regulation in Canada typically occurs as a function of the protections provided by other statutes protecting wildlife (Fiore 1993). For many jurisdictions, wetland policy is absent or in the early stages of development (Clare et al. 2011). One exception is Alberta, Canada, where the wetland policy resembles that of the United States (Clare et al. 2011). The 1999 Water Act requires approval for any activity that may cause an effect on the aquatic environment (Clare et al. 2011). Limiting this approach is the purpose of the act, which requires conservation goals to be balanced with Alberta's need for economic growth and prosperity. Alberta's regulations also do not require an alternatives analysis or a formal process for identifying the basic purpose of a project (Clare et al. 2011).

In the Republic of Korea, the Public Waters Reclamation Act grants private property rights in reclaimed lands to those with a reclamation permit (Kim 2011). Reclamation permits are issued to accommodate and grant consent to those who seek to use public waters or drain or reclaim privately owned public waters (Kim 2011). Recent amendments to the act require reclamation activities be environmentally friendly and disclose any environmental impacts on the site (Kim 2011). Yet environmental provisions remain rhetorical and subordinate to economic interests, the laws contain a narrow definition of wetlands and contemplate relatively weak penalty provisions (Kim 2011).

Uganda's regulations require a wetland resource use permit for certain activities in wetlands, such as brickmaking, drainage, aquaculture, and commercial exploitation of wetland resources (Gardner 2003). The regulations exempt other activities such as harvesting papyrus and collecting water for domestic use (Gardner 2003).

Australia's national government has relatively weak authority over purely domestic environmental matters (Gardner 2003). It has been able to assert more authority pursuant to its international obligations, such as the Ramsar Convention. Government approval is required for any action likely to have a significant impact on the ecological character of a Ramsar wetland, and management plans for these wetlands must outline restoration or rehabilitation. These national permit, however, apply to only the limited set of designated Ramsar wetlands. Local regulations fill in some of the gaps from the national level.

Environmental Trading Markets

Many permit-based regulatory systems require the permittee to mitigate wetland impacts resulting from the permitted activities. A pitfall of compensatory mitigation for wetland loss is that it often requires the assumption that the restoration is effective (Gardner 2003). As an alternative, mitigation banking provides front restoration, creation, enhancement, or preservation of wetlands to provide compensatory mitigation in advance of the impacts authorized by a permit (Gardner 2003). An incentives-based market emerges as permittees create their own mitigation banks and entrepreneurial banks, or commercial banks, and sell mitigation credits to permittees seeking to satisfy a mitigation condition (Gardner 2003).

Compensatory mitigation is generally disfavored. The United States and Canada implement a mitigation hierarchy that considers avoidance and minimization before compensatory mitigation. The European Union uses a sequential order where compensatory measures are considered only after defining the specific adverse impact to a Natura 2000 site. A disadvantage to mitigation banking is that promised mitigation projects may fail to replace the functions of the original wetlands (Gardner 2003). Other pitfalls include lack of clear performance standards, lack of monitoring or enforcement, and little long-term maintenance (Gardner 2003). Benefits of mitigation banking include more consistent management of wetland restoration or remediation efforts at designated, pre-approved sites.

Future Challenges

A major challenge for effective permit schemes is overcoming the general failure to avoid wetland impacts despite contrary stated policy preferences, which has resulted in an overall decline in the number and quality of natural wetlands (Clare et al. 2011). Successful wetland regulatory permit programs will clearly state the permit scheme's goals and objectives, include an educational and public participation component, and provide for monitoring and enforcement to ensure compliance (Gardner 2003). Creating permit schemes that encompass these goals is a major challenge, even in countries that already implement a wetlands permit program. Studies on the effectiveness of the US permit program indicate that despite the broad goals of the Clean Water Act, the regulatory framework does not adequately recognize wetland values (Adler 2013).

The international agreements described above promote education by informing governments and their citizens about the importance and values of wetlands. Still, more understanding is needed, as evidenced by the continued destruction of wetlands worldwide.

Major challenges for developing effective permit schemes also include limited government resources and authority. For developing countries and countries in economic transition, the Ramsar Convention's "Small Grants Program" provides funds to create financial incentives for wetland restoration (Gardner 2003). The 2014 recipients include the Hawizeh Marsh (part of the Mesopotamian Marshes), Kenya (to adopt a nationwide wetland policy), Darlen Province (Punta Patino), and the Polesle region (Shatsk Lakes, Perebrody Peatlands, Polissia Mires, and Desna River Floodplains). Governments also may lack the necessary authority to implement a permit scheme. Debate continues in the United States as to the scope of federal jurisdiction under the Clean Water Act to require a permit for dredging or filling activity, especially when it concerns private land. Creative solutions are necessary to overcome these obstacles.

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Avoid-Mitigate-Compensate Sequence: Wetland Conservation

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Royal C. Gardner

Contents

Abstract

The avoid-mitigate-compensate sequence is a common approach to wetland conservation. It has been recognized at the international level in various Ramsar Convention documents and in other multilateral fora. The avoid-mitigate-compensate sequence is also a regular feature of many countries' environmental laws and policies, often using interchangeable correlated terms that are essentially consistent. In the wetland context, the avoid-mitigate-compensate sequence can be an important tool for maintaining the ecological character of wetlands. Such an approach may sometimes be related to a "no net loss" objective.

Keywords

Avoidance · Mitigation · Compensation · Offset · "no net loss"

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Introduction

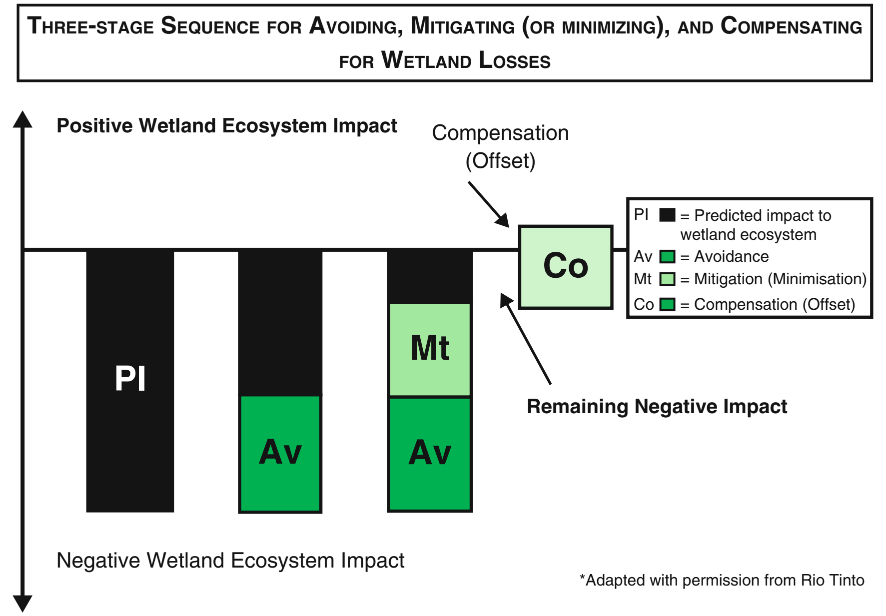
The avoid-mitigate-compensate sequence is a common approach to wetland conservation. It has been recognized at the international level in various Ramsar documents and in other multilateral fora (Ramsar Convention Conference of the Parties 2012; International Finance Corporation 2012). The avoid-mitigate-compensate sequence is also a regular feature of many countries' environmental laws and policies (Gardner et al. 2012). While not all of these countries use the precise avoid-mitigate-compensate formulation, they use interchangeable correlated terms that are consistent in essence. In the wetland context, the avoid-mitigate-compensate sequence can be an important tool for maintaining the ecological character of wetlands. Such an approach may sometimes be related to a "no net loss" objective.

The Avoid-Mitigate-Compensate Sequence

The Ramsar Convention Conference of the Parties (2012) has generally defined the term "avoidance" to mean "*. . .proactive measures to prevent adverse change in a wetland's ecological character through appropriate regulation, planning or activity design decisions. Examples would include choosing a non-damaging location for a development project, or choosing a 'no-project' option when the risks to the maintenance of ecological character are assessed as being too high*."

If impacts cannot be avoided then, under this sequence, mitigation should occur. "Mitigation" as defined by the Ramsar Convention Conference of the Parties (2012) involves "*. . .reactive practical actions that minimize or reduce* in situ *wetland impacts*." Ramsar Convention Resolution X.17 (2008) provides examples such as "*. . .changes to the scale*, *design*, *location*, *siting*, *process*, *sequencing*, *phasing*, *management and/or monitoring of the proposed activity*, *as well as restoration or rehabilitation of sites*." Mitigation actions can take place in situ or ex situ, as long as their effect is realized in the site where change in ecological character is likely. The Ramsar Convention Conference of the Parties (2012) has noted that "*. . .*[*i*]*n many cases it may not be appropriate to regard restoration as mitigation*, *since doing so represents an acknowledgement that impact has already occurred*: *in such cases the term* "*compensation*" *may be a truer reflection of this kind of response*."

If unavoidable impacts cannot be minimized, the next step in the sequence is compensation. The Ramsar Convention Conference of the Parties (2012) views "compensation" as "*. . .actions that are intended to offset the residual impacts on wetland ecological character that resite wetland restoration or creation project*, *provided it adds value beyond what would have happened otherwise* (i.e., *relying on an already-planned benefit would not constitute compensation*)." From a wise use perspective, it is preferable to compensate for wetland loss with wetlands of a similar type and in the same local water catchment (Ramsar Convention ResolutionVII.24, 1999). It is also preferable if the compensation is in place prior to impacts to reduce temporal losses of ecosystem functions and services (Ramsar Convention Conference of the Parties 2012).



Three-stage Mitigation Sequence Chart: Rio Tinto, 2008. Rio Tinto and biodiversity: Achieving results on the ground. Rio Tinto's Biodiversity Strategy. Available at: http://www.riotinto.com/documents/ReportsPublications/ RTBidoversitystrat-egyfinal.pdf.

Fig. 1 Avoid-mitigate-compensate sequence (Adapted from McKenney and Wilkinson 2015; reproduced with permission of The Nature Conservancy)

Thus, as illustrated below in the figure (Fig. 1), reproduced from Gardner et al. (2012), the avoid-mitigate-compensate sequence lays out a stepwise approach. It emphasizes that negative wetland impacts should be avoided if at all possible. If such negative impacts cannot be avoided or prevented, actions should be taken to mitigate (minimize or reduce) this wetland loss or degradation. Finally, if wetland loss or degradation remains after such mitigation, actions should be taken to compensate for (i.e., offset) these residual impacts.

For further information and examples in national laws and policies, refer to Ramsar Scientific and Technical Review Panel Briefing Note Briefing Note 3 (Gardner et al. 2012), from which this entry is adapted.

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116 Avoiding Loss of Agricultural Subsidies: Swampbuster

L. Leon Geyer and Dan Lawler

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Abstract

Throughout past centuries, wetland loss has developed into a serious issue that carries profound environmental, economic, and political implications in the United States. The lower 48 states have lost an estimated 53% of their original wetlands over the last 200 years, largely because the United States' agricultural sector has grown and developed, a profound amount of wetlands have been converted into cropland in order to capitalize on their fertile, nutrient-rich soils. This diminishes the capacity of ecosystems to generate a wealth of beneficial services. Historically, economic incentives embodied in agricultural statutes have influenced wetland decline by increasing short-term profitability. To reverse this trend, efforts to protect wetlands were implemented through the Highly Erodible Land Conservation and Wetland Conservation Compliance provisions – widely known as "Swampbuster" – as a part of the 1985 US Food Security Act. Along with key amendments in the Food, Agriculture, Conservation, and Trade Act of 1990, the statute encourages crop producers to conserve wetlands through finan-

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cial incentives, substantially changing the working relationship between farmers and their lands. In order to effectively display how the loss of agricultural subsidies relates to wetlands, this chapter will detail specific Swampbuster provisions, identify their strengths and weaknesses, and discuss future challenges that affect Swampbuster's role in environment, agriculture, and politics.

Keywords
Wetlands · Swampbuster · Agricultural program benefits · Agricultural subsidies

Introduction

Throughout past centuries, wetland loss has developed into a serious issue that carries profound environmental, economic, and political implications in the United States. According to the Natural Resources Conservation Service (NRCS), the lower 48 states have lost an estimated 53% of their original wetlands over the last 200 years (NRCS 2013), largely because as the United States' agricultural sector has grown and developed, a profound amount of wetlands have been converted into cropland in order to capitalize on their fertile, nutrient-rich soils. This deprives ecosystems of the "environmental functions of wetlands, such as flood control, sediment control, groundwater recharge, water quality, wildlife habitat, recreation, and esthetics" (NRCS 2013) and occurs because "historically, the economic incentives embodied in agricultural statutes have influenced wetland decline by increasing the profitability while reducing the risks of agriculture" (DOI 2013). In order to reverse this trend, efforts to protect wetlands were implemented through the Highly Erodible Land Conservation and Wetland Conservation Compliance provisions – widely known as "Swampbuster" – as a part of the 1985 Food Security Act. Along with key amendments in the Food, Agriculture, Conservation, and Trade Act of 1990, the statute encourages crop producers to conserve wetlands through financial incentives, substantially changing the working relationship between farmers and their lands. In order to effectively display how the loss of agricultural subsidies relates to wetlands, this article will detail specific Swampbuster provisions, identify their strengths and weaknesses, and discuss future challenges that affect Swampbuster's role in environment, agriculture, and politics.

Swampbuster Provisions

In the United States' agricultural sector, price volatility for commodities presents a significant threat to farmers' profitability, exposing them to a high degree of risk. In order "to make effective risk management decisions, crop producers must integrate farm programs and crop insurance alternatives in a comprehensive risk management decision-making framework" (Lubben et al. 2013), stressing the importance of agricultural subsidies and federal farm benefits in maintaining revenue stability. Swampbuster provides a financial disincentive against the conversion of wetlands to agricultural use by making farmers who do convert wetlands “ineligible for all or a portion of certain USDA program benefits, including loans, subsidies, crop insurance, and price support programs” (USFWS 2003). According to the US Fish and Wildlife Service, conversion of a wetland specifically refers to the “draining, dredging, filling, leveling, removing woody vegetation, or other means for the purpose, or to have the effect of, making possible the production of an agricultural commodity” (USFWS 2003). It is important to note, however, that farmers who converted wetlands prior to December 23, 1985 are exempted from Swampbuster regulations and that farmers can regain eligibility for program benefits if they restore their wetland or effectively mitigate the functions and values of the former wetland (USFWS 2003). Thus, farmers are motivated to conserve wetlands by avoiding the loss of agricultural subsidies.

In order to remedy some of the weaknesses of the original 1985 Swampbuster provisions, the Food, Agriculture, Conservation, and Trade Act (FACTA) was passed in 1990, making some important changes to the statute. According to the DOI, “under FACTA, altering a wetland triggers the Swampbuster sanctions whether or not an agricultural commodity is planted,” increasing the amount of wetlands conserved on agricultural lands (DOI 1994). In addition, FACTA scales penalties according to the severity of the violation, making the statute more accommodating to the high degree of circumstantial variability from case to case (DOI 1994).

Effectiveness

The NRCS estimates that about 87% of wetland losses from the mid-1950s to mid-1970s were due to agricultural conversion (NRCS 2013). Swampbuster has been very effective in limiting this trend, as an article published in *Ecological Applications* argues that “declining rates of net loss have been attributed to the introduction of wetland regulation in the mid-1980s” (De Steven and Lowrance 2011). Statistics from the NRCS support this contention, as “the wetland conservation provisions have sharply reduced wetland conversions for agricultural uses, from 235,000 acres per year before 1985 to 27,000 acres per year from 1992 through 1997” (NRCS 2013).

Although Swampbuster has significantly limited wetland-to-cropland conversions, it does have deficiencies that weaken its efficacy. The first, according to the Department of the Interior (DOI), is that the statute provides an exemption for the conversion of artificial wetlands (DOI 2013). This allows crop producers to convert man-made wetlands into cropland without fear of farm program ineligibility, leaving areas like California’s Central Valley vulnerable to environmental degradation due to their high proportion of artificial wetlands (DOI 1994). Another deficiency that weakens the environmental conservation compliance provisions is that “the particular subsidies extended to growers of sugarcane were not among the Act’s sanctions”

(DOI 2013). This allows farmers to convert wetlands into sugarcane fields without losing farm program eligibility, although this exemption has had a limited effect because sugarcane made up only 0.29% of the total area of cropland harvested in the US in 2012 and is only produced in Florida, Hawaii, Louisiana, and Texas (National Agriculture Statistics Service 2013).

Future Challenges

Despite its overall effectiveness in preventing wetland loss, there are some future challenges that could pose problems for Swampbuster. The first concerns the reduction or elimination of direct payments from the federal farm commodity program. According to the USDA's Economic Research Service, if direct payments are eliminated or reduced, "compliance incentives would be reduced on many farms, potentially increasing environmental quality problems" (Claassen 2012). This suggests that less crop producers would conserve their wetlands because the financial reward for doing so would be significantly lowered, although cutting direct payments would likely only happen if Congress attempts to lower the federal deficit or if a government shutdown prevented the passing of a farm bill.

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Compensation in Wetlands

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Royal C. Gardner

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Abstract

Compensation in the wetland context generally refers to actions taken to offset residual impacts of activities that adversely affect wetlands. Compensation will typically occur as the third stage of the avoid-mitigate-compensate sequence; that is, compensation is utilized when a development project cannot avoid wetland impacts, and steps to mitigate or minimize those unavoidable impacts do not reduce the impact level to zero or some minimal threshold. Depending on the jurisdiction, compensation may also be called an offset or compensatory mitigation or similar terms. A compensation requirement can be part of a policy that seeks to achieve an overall "no net loss" of wetland area and/or function. For example, regulations promulgated by the US Environmental Protection Agency and US Army Corps of Engineers (2008), the wetland regulatory agencies under the US Clean Water Act offer working definitions.

Keywords

Compensation · Offset · Mitigation · "no net loss"

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Introduction

Compensation in the wetland context generally refers to actions taken to offset residual impacts of activities that adversely affect wetlands. Compensation will typically occur as the third stage of the avoid-mitigate-compensate sequence; that is, compensation is utilized when a development project cannot avoid wetland impacts and steps to mitigate or minimize those unavoidable impacts do not reduce the impact level to zero or some minimal threshold. Depending on the jurisdiction, compensation may also be called an offset or compensatory mitigation or similar terms (Gardner et al. 2012).

A compensation requirement can be part of a policy that seeks to achieve an overall "no net loss" of wetland area and/or function. For example, regulations promulgated by the US Environmental Protection Agency and US Army Corps of Engineers (2008), the wetland regulatory agencies under the US Clean Water Act offer this definition:

> "Compensatory mitigation means the restoration (re-establishment or rehabilitation), establishment (creation), enhancement, and/or in certain circumstances preservation of aquatic resources for the purposes of offsetting unavoidable adverse impacts which remain after all appropriate and practicable avoidance and minimization has been achieved."

The Figure below (US National Research Council 2001) illustrates the placement and relationship of compensation in the avoid-mitigate-compensate sequence required by the US Clean Water Act (Fig. 1).

Fig. 1 Avoid-Mitigate (Minimize)-Compensate sequence required by US Clean Water Act (Image credit: M Everard © copyright remains with the author)

The Ramsar Convention Conference of the Parties (2012) has adopted a similar definition of compensation:

> "*Compensating for wetland impacts refers to actions that are intended to offset the residual impacts on wetland ecological character that remain after any mitigation has been achieved. An example of compensation would be an on-site or off-site wetland restoration or creation project, provided it adds value beyond what would have happened otherwise (i.e., relying on an already-planned benefit would not constitute compensation). Contracting Parties have emphasized the fact that it is preferable to compensate for wetland loss with wetlands of a similar type and in the same local water catchment (Resolution VII.24, 1999), and priority should be given to on-site compensation*."

Other entities contemplate that compensation may in some cases result in a net gain of biodiversity. The Business and Biodiversity Offsets Program (BBOP), which has applicability beyond wetlands, is a diverse collaborative effort that consists of more than 75 companies, financial institutions, governments, civil society organizations, and individuals. BBOP (2012) defines offsets as:

> "*...measures taken to compensate for any residual significant, adverse impacts that cannot be avoided, minimised and / or rehabilitated or restored, in order to achieve no net loss or a net gain of biodiversity. Offsets can take the form of positive management interventions such as restoration of degraded habitat, arrested degradation or averted risk, protecting areas where there is imminent or projected loss of biodiversity.*"

Compensation Methods

To strive for equivalency between residual impacts and compensation, it is necessary to use the same assessment method for both impact and compensation sites. If a policy objective is functional replacement, then in-kind compensation or offsets should be preferred. Thus, if a development project causes unavoidable impacts to ephemeral wetlands, the compensation (whether it be restoration, enhancement, creation, or preservation) should involve ephemeral wetlands.

Compensation ratios or other credit assessment methods can affect the amount of compensation that must be provided. Such approaches "*...may be used to adjust for the relative quality of impact sites and [compensation] projects; to account for temporal losses; the distance between the [compensation] and the impact sites; and/or to factor in the uncertainty associated with a [compensation] site*" (Gardner et al. in press).

A compensation project can be delivered through a variety of mechanisms. The developer may undertake the compensation project itself or hire an environmental consultant to do so. In some jurisdictions (such as the USA), this is considered "permittee-responsible" compensation because the developer (permittee) remains legally responsible for the compensation project's outcomes.

Another compensation mechanism is wetland banking, which is also called wetland mitigation banking. As Ramsar Resolution XI.9 explains, it is:

> "*...a tool for providing wetland compensation to offset unavoidable impacts that remain after mitigation measures. It is most well developed in the USA, where it is viewed as an incentive-based approach to wetland protection. In its simplest form, a site owner generates compensation credits through the restoration, enhancement, creation, and/or preservation of wetlands. The amount of credits generated is based on the ecological improvements at the site. Credits are then sold to developers to offset adverse wetland impacts to the same type of habitat elsewhere*."

A significant feature of a wetland banking scheme is that the legal responsibility for the compensation shifts from the developer to the mitigation bank operator (Gardner 2011).

A third compensation mechanism, known as in-lieu fees or fee mitigation, also involves a shifting of legal responsibility. (The fees are paid in lieu of the developer providing compensation itself, hence the name.) In such a scenario, an in-lieu-fee administrator would collect and pool monies from developers to conduct compensation projects, generally after development impacts. The timing of the compensation is a significant factor distinguishing wetland banks and in-lieu fees. Wetland bank credits are released only after certain performance standards are met and the on-the-ground ecological replacements are usually provided closer in time to the impacts. With permittee-responsible compensation and in-lieu-fee arrangements, the lag time between impacts and offsets is typically greater (Gardner 2011).

All types of compensation raise concerns, such as those related to long-term stewardship of compensation sites. Although a number of countries have adopted "no net loss" policies that include compensation requirements, there have been a few studies assessing the effectiveness of these approaches (Ramsar 2012). Indeed, assessments in the USA have found that the goal of no net loss of wetlands is not being achieved, especially with respect to wetland functions, due in part to failure of permittee-responsible compensation projects. For example, the US National Research Council (2001) reported that "*...the goal of no net loss of wetlands is not being met for wetland functions by the compensation program, despite progress in the last 20 years*".

While a later US study reported a net gain in national wetland area from 1998 to 2004, it emphasized that this did not translate to a net gain in wetland function (Dahl 2006). Data did not exist to support such a conclusion. Moreover, Stedman and Dahl (2008) subsequently noted that during this timeframe certain regions (such as the southeastern US) and certain types of wetlands (such as forested wetlands) had lost area. These findings led the Ramsar Convention Conference of the Parties and others to reemphasize the importance of avoiding wetland impacts in the first instance (Ramsar 2012).

Note that compensation in this context does not refer to compensation or money to be paid to property owners (called in the US "just compensation" for example) when environmental laws or regulations are deemed to unduly infringe upon private property rights.

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Mitigation Banking for Wetlands

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Mark Everard

Contents

Abstract

The concept of mitigation banking relates to the "banking" of habitat preserved, enhanced, restored, or created, which may then be used to offset harm to other nearby habitats from development activities, particularly to a wetland, stream, or habitat conservation area. The intended outcome is to replace like-for-like the functions and values provided by wetland habitats that may be converted or otherwise adversely affected by proposed development activities. Under mitigation banking, "banked" wetland enhancement or creation schemes can be sold to development proponents as habitat compensation required by enforcement agencies. The USA has been a leading nation in implementing mitigation banking.

Keywords

Habitat banking · Mitigation · Mitigation credits · Development · Compensation · Clean water act · Interagency review team · Mitigation sequence

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Introduction

The concept of mitigation banking relates to the “banking” of habitat preserved, enhanced, restored, or created, which may then be used to offset harm to other nearby habitats from development activities, particularly to a wetland, stream, or habitat conservation area (US EPA n.d.-b). The intended outcome is to replace like-for-like the functions and values provided by wetland habitats that may be converted or otherwise adversely affected by proposed development activities. Under mitigation banking, “banked” wetland enhancement or creation schemes can be sold to development proponents as habitat compensation required by enforcement agencies. The USA has been a leading nation in implementing mitigation banking.

Mitigation Banking in the USA

In the United States, federal agencies as well as many state and local governments are obligated under section 404 of the Clean Water Act (US EPA n.d.-a) to require mitigation for the disturbance or destruction of wetland, stream, or endangered species habitat. Once approved by regulatory agencies, a mitigation bank may sell credits to developers whose projects will impact these various ecosystems.

Credits are units of exchange defined as the ecological value associated with converting to other economic uses a naturally occurring wetland or other specific habitat type. Mitigation credits to compensate for riparian impacts may be assigned in relation to the linear distance of a stream functioning at the highest possible capacity within the watershed of the bank.

What Is a Mitigation Bank?

In the USA, a mitigation bank is a wetland, stream, or other aquatic resource area that has been restored, established, enhanced, or (in certain circumstances) preserved for the purpose of providing compensation for unavoidable impacts to aquatic resources permitted under Section 404 of the Clean Water Act or a similar state or local wetland regulation (US EPA n.d.-a). A mitigation bank may be created when a government agency, corporation, nonprofit organization, or other entity undertakes these activities under a formal agreement with a regulatory agency. Mitigation banks have four distinct components:

- The bank site: the physical acreage restored, established, enhanced, or preserved
- The bank instrument: the formal agreement between the bank owners and regulators establishing liability, performance standards, management and monitoring requirements, and the terms of bank credit approval
- The Interagency Review Team (IRT): the interagency team that provides regulatory review, approval, and oversight of the bank

- The service area: the geographic area in which permitted impacts can be compensated for at a given bank

Details of how this process operates can be found in the US EPA (n.d.-c) *Compensatory mitigation resources* guidance, which specifies that "For unavoidable impacts, compensatory mitigation is required to replace the loss of wetland and aquatic resource functions in the watershed. Compensatory mitigation refers to the restoration, establishment, enhancement, or in certain circumstances preservation of wetlands, streams or other aquatic resources for the purpose of offsetting unavoidable adverse impacts." This is effected as the third phase of a "mitigation sequence."

The Mitigation Sequence

Compensatory mitigation is actually the third step in a sequence of actions that must be followed to offset impacts to aquatic resources. A 1990 Memorandum of Agreement (MOA) between the Environmental Protection Agency (EPA) and the Department of Army establishes a three-part process, known as the mitigation sequence. The purpose of this mitigation sequence is to help guide mitigation decisions and determine the type and level of mitigation required under Clean Water Act Section 404 regulations:

- Step 1. Avoid: Adverse impacts to aquatic resources are to be avoided and no discharge shall be permitted if there is a practicable alternative with less adverse impact.
- Step 2. Minimize: If impacts cannot be avoided, appropriate and practicable steps to minimize adverse impacts must be taken.
- Step 3. Compensate: Appropriate and practicable compensatory mitigation is required for unavoidable adverse impacts which remain. The amount and quality of compensatory mitigation may not substitute for avoiding and minimizing impacts.

The "avoid-mitigate-compensate" sequence is addressed in more detail elsewhere in the Wetland Book.

Compensatory Mitigation Measures

After addressing firstly the steps of avoiding or minimizing impacts, projects that proceed and will cause adverse impacts to wetlands, streams, and other aquatic resources typically require some type of compensatory mitigation. The Army Corps of Engineers (or approved state authority) is responsible for determining the appropriate form and amount of compensatory mitigation required. Methods of compensatory mitigation include restoration, establishment, enhancement, and preservation:

- Restoration comprises the reestablishment or rehabilitation of a wetland or other aquatic resource with the goal of returning natural or historic functions and characteristics to a former or degraded wetland. Restoration may result in a gain in wetland function or wetland acres, or both.
- Establishment (creation) entails development of a wetland or other aquatic resource where a wetland did not previously exist through manipulation of the physical, chemical, and/or biological characteristics of the site. Successful establishment results in a net gain in wetland acres and function.
- Enhancement covers activities conducted within existing wetlands that heighten, intensify, or improve one or more wetland functions. Enhancement is often undertaken for a specific purpose such as to improve water quality, flood water retention, or wildlife habitat. Enhancement results in a gain in wetland function, but does not result in a net gain in wetland acres.
- Preservation occurs when permanent protection of ecologically important wetlands or other aquatic resources is enacted through the implementation of appropriate legal and physical mechanisms (i.e., conservation easements, title transfers). Preservation may include protection of upland areas adjacent to wetlands as necessary to ensure protection or enhancement of the aquatic ecosystem. Preservation does not result in a net gain of wetland acres and may only be used in certain circumstances, including when the resources to be preserved contribute significantly to the ecological sustainability of the watershed.

Mechanisms for Compensatory Mitigation

Compensatory mitigation for unavoidable wetland impacts may be accomplished through three distinct mechanisms:

- Permittee-responsible mitigation, in which the permittee undertakes restoration, establishment, enhancement, or preservation of wetlands to compensate for wetland impacts resulting from a specific project. The permittee performs the mitigation after the permit is issued and is ultimately responsible for implementation and success of the mitigation. Permittee-responsible mitigation may occur at the site of the permitted impacts or at an off-site location within the same watershed.
- Mitigation Banking: A wetlands mitigation bank is a wetland area that has been restored, established, enhanced, or preserved, which is then set aside to compensate for future conversions of wetlands for development activities. Permittees, upon approval of regulatory agencies, can purchase credits from a mitigation bank to meet their requirements for compensatory mitigation. The value of these "credits" is determined by quantifying the wetland functions or acres restored or created. The bank sponsor is ultimately responsible for the success of the project.

Mitigation banking is performed "off-site," meaning it is at a location not on or immediately adjacent to the site of impacts, but within the same watershed. Federal regulations establish a flexible preference for using credits from a mitigation bank over the other compensation mechanisms.

- In-Lieu Fee Mitigation: Mitigation that occurs when a permittee provides funds to an in-lieu-fee sponsor (a public agency or nonprofit organization). Usually, the sponsor collects funds from multiple permittees in order to pool the financial resources necessary to build and maintain the mitigation site. The in-lieu fee sponsor is responsible for the success of the mitigation. Like banking, in-lieu fee mitigation is also "off-site" but, unlike mitigation banking, it typically occurs after the permitted impacts.

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Enforcement: Wetlands

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Meredith Weinberg

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Abstract

Enforcement of laws and regulations protecting wetlands is an important piece of the proverbial wetlands management puzzle. Enforcement usually works in concert with other actions like identification of wetlands, education on the importance of wetlands conservation, and restoration of wetlands that have been damaged or destroyed. Tools for wetlands enforcement vary greatly, and three examples of such tools are as follows: (1) criminal and civil enforcement of law protecting wetlands; (2) administrative actions ranging from notification to violators of wetlands protections to voluntary agreements requiring restoration and future compliance with wetlands protections; and (3) use of diplomatic relationships to enforce wetland protection measures.

Keywords

Protection · Management · Conservation · Restoration · Criminal enforcement · Civil enforcement · Diplomatic relationships

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Introduction

Enforcement of laws and regulations protecting wetlands is an important piece of the proverbial wetlands management puzzle. Enforcement usually works in concert with other actions like identification of wetlands, education on the importance of wetlands conservation, and restoration of wetlands that have been damaged or destroyed. Tools for wetlands enforcement vary greatly, and three examples of such tools are as follows: (1) criminal and civil enforcement of law protecting wetlands; (2) administrative actions ranging from notification to violators of wetlands protections to voluntary agreements requiring restoration and future compliance with wetlands protections; and (3) use of diplomatic relationships to enforce wetland protection measures.

Wetlands Enforcement Tools

Criminal and Civil Enforcement of Law Protecting Wetlands

An example of the use of criminal and civil actions to enforce laws protecting wetlands exists in the United States under the federal Clean Water Act. The law was passed in 1972 with the objective "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (33 U.S.C. § 1251(a)). Accordingly, the Clean Water Act (along with its regulations, which have the force of law) prohibits "discharging" without a permit any dredged or fill material into wetlands that come under the jurisdiction of the federal government (33 U.S.C. § 1344). Determining whether certain wetlands come within federal jurisdiction (and are thus covered by these prohibitions) can be controversial, based in part on a US Supreme Court decision in 2006. (*See Rapanos v. United States*, 547 U.S. 715 (2006)). In any event, the Clean Water Act provides tools for enforcing its prohibitions on alteration of federal wetlands.

Specifically, the Clean Water Act gives the US Environmental Protection Agency (US EPA) primary authority to enforce the Clean Water Act's prohibition on discharging into federal wetlands without a permit. EPA can enforce wetlands cases by bringing judicial actions in federal courts (through the US Department of Justice). In a civil judicial action, a federal court may order a violator to cease its discharge and to remove fill from a wetland, restore damaged wetlands, or conduct additional remedies and may also assess penalties of up to $51,570 per day, per wetlands violation (33 U.S.C. § 1319(b)). In a criminal judicial action, a federal court may order the same remedies but also may assess fines or sentence a violator to imprisonment if the violation of Clean Water Act wetlands protection was criminally negligent or done with knowing disregard of those protections (33 U.S.C. § 1319(c)).

Like the federal government, many states have laws protecting wetlands which they can enforce through civil and criminal actions. (See, for example, Florida Statute Annotated § 373.430(1)(b) (prohibiting discharge into state wetlands without a permit and providing authority for Florida Department of Environmental

Protection to bring civil or criminal suits against violators of statute)). Although neither US EPA nor the states have the resources to bring a criminal or civil action against every violator of Clean Water Act (or state law) wetland protection measures, both have achieved wetlands restoration in cases in which they have pursued civil or criminal actions. In one example from 2007, the US EPA brought a civil suit against two brothers who discharged into wetlands on their property in Kentucky without a permit. (*See United States v. Cundiff*, 480 F. Supp.2d 940 (W.D. Ky. 2007), *aff'd*, 555 F.3d 200 (11th Cir. 2009)). The court permanently enjoined the defendants from further discharges into the wetlands, assessed a $225,000 penalty, and ordered that the defendants conduct restoration of the wetlands they had damaged. In another example from 2005, the US EPA obtained a guilty verdict in the criminal prosecution of several wetlands violators who filled approximately 260 acres of wetlands and other federal waters without a permit. (*See United States v. Lucas*, Case No. 1:04-CR-00060 (S.D. Miss.), *aff'd*, 516 F.3d 316 (5th Cir. 2008)). The federal court sentenced the violators to prison terms, assessed fines, and ordered them to pay restitution of almost $1.4 million.

Wetlands Enforcement Through Administrative Action

Administrative action is more frequently used in the United States to enforce wetlands protections at both the federal as well as at the state and local level (See for example New York State DEE-6 and DEE-7: Freshwater and Tidal Wetlands Enforcement Policies (stating that "the primary goal of [New York state's wetlands] enforcement policy. . .is to enhance the preservation and protection of. . .wetlands by seeking restoration of wetland benefits and functions lost as the result of illegal activity," and describing the administrative actions that can be taken by the state Department of Environmental Conservation when a person violates state wetlands protections)). Examples of federal or state administrative action range from warning a person that he is violating wetlands laws, issuing a "notice of violation" with or without the threat of penalties, or promulgating administrative compliance orders that require restorative action and assess penalties (Environmental Law Institute 2008). In contrast to many civil or criminal judicial actions, administrative actions tend to have the advantage of quick resolution without the delay of litigation (See *Sackett v. Environmental Protection Agency*, 132 S. Ct. 1367, 1374 (2012), which holds that the US EPA's administrative compliance orders are immediately challengeable in federal court, but noting that administrative orders generally provide a way to notify persons of potential wetlands violations and quickly resolve those violations through voluntary compliance).

In one example of an administrative action used to enforce wetlands protections, in 2006, the US EPA and a wetlands violator voluntarily entered into an administrative compliance order on consent (See In the Matter of Robert E. Cross and Wolf Creek Assocs., Administrative Compliance Order on Consent, EPA Docket No. CWA-08-2007-0006). The order details a history of the violator's discharges into wetlands in Colorado, but the point of the order was "remedial, not punitive," and thus required restoration, enhancement, and preservation of wetlands going forward. The US EPA took the position that these activities were “achievable as a practicable matter through commonly used construction, digging, filling, revegetation, and best management practices.” By voluntarily entering the Administrative Compliance Order (ACO), the violator did not admit liability but agreed to abide by the terms of the order. In essence, by issuing the ACO instead of instituting a civil or criminal action against the violator in federal court, the US EPA was able to achieve wetlands restoration without further delay.

Wetlands Enforcement Through Diplomacy

International treaties are the best example of the use of diplomatic relationships for enforcing wetlands protections. A number of treaties seek to protect wetlands: the Convention on Wetlands of International Importance (“Ramsar Convention” adopted in 1971 and signed in Ramsar, Iran) is probably the best known. However, the Convention on Biological Diversity and the Convention on Conservation of Migratory Species of Wild Animals (both relating to global protection of species and their wetlands habitats) are also influential, as are a number of regional conventions protecting marine environments. Most of these treaties have only one avenue for enforcement: the expectation under international law that signatories will respect their treaty obligations and will suffer diplomatic and political pressure if they do not (Griffin 2012).

With 168 signatories to date, the Ramsar Convention’s mission is “. . .the conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world” (Ramsar Commision, undated). The Ramsar Convention does not include enforcement of wetlands protection as one of its purposes. Instead, it focuses on designating important wetlands on what is known as the “Ramsar List,” advising on best wetlands management practices, and brokering international cooperation concerning transboundary wetlands and the development projects that may affect such wetlands.

The Ramsar Convention contains no codified enforcement provisions, such as sanctions for destruction or delisting of wetlands. Indeed, the convention allows for delisting of protected wetlands inside a country’s borders where there is an undefined “urgent national interest” (see Convention on Wetlands of International Importance especially as Waterfowl Habitat, Ramsar (Iran), 2 February 1971, Article 2(5)). Ultimately, any enforcement of the convention’s principles relating to wetlands protection and preservation has to rely solely on political influence or diplomatic pressure from the international community, with the exception being possible referral to international courts with the acquiescence of two parties in a dispute (Kunich 2003, p. 68).

Future Challenges

The management of wetlands in the future likely will require a combination of enforcement of existing wetland protection measures in addition to careful thought both about what protections are needed as well as how such protections should be enforced. Close examination of the enforcement tools available today, including both successes and failures, can be a helpful way to start that conversation.

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Conservation Reserve Program (CRP): Example of Land Retirement

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Mark Everard

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Abstract

The US Conservation Reserve Program (CRP) is a nationwide land retirement program operating across the United States. Under the CRP, the US government offers incentives to landholders in areas of agricultural production, buffer areas, and wetlands across the United States, inducing them to enter into contracts to change land use on highly erodible and environmentally sensitive cropland and pasture. The CRP was instituted initially as a means to avert soil erosion in cropland, which was a significant problem in the USA in the 1970s when annual cropland soil erosion losses were estimated at 2–6.8 billion tons. Blowing soil also reduced visibility and air quality, most notably in Gaines County, Texas, as well as making land susceptible to wind erosion and reducing agricultural productivity. In a bid to resolve this problem, the US government formally established the CRP in 1985 following initial trials.

Keywords

Erosion · Ecosystem services · Habitat creation · Land retirement · Environmental Benefit Index · Reverse auction

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Introduction

The US Conservation Reserve Program (CRP) is a nationwide land retirement program operating across the United States. Under the CRP, the US government offers incentives to landholders in areas of agricultural production, buffer areas, and wetlands across the United States, inducing them to enter into contracts to change land use on highly erodible and environmentally sensitive cropland and pasture. The CRP was instituted initially as a means to avert soil erosion in cropland, which was a significant problem in the USA in the 1970s when annual cropland soil erosion losses were estimated at 2–6.8 billion tons. Blowing soil also reduced visibility and air quality, most notably in Gaines County, Texas, as well as making land susceptible to wind erosion and reducing agricultural productivity. In a bid to resolve this problem, the US government formally established the CRP in 1985 following initial trials.

The CRP has since evolved to address a "bundle" of connected ecosystem services including water management and water quality, biodiversity, and air quality protection. CRP payments support conversion of farmed land to grass, trees, wildlife cover, or other uses providing ecosystem services including surface water quality improvement, wildlife habitat creation, carbon storage, preservation of soil productivity, protection of groundwater quality, and reduction of offsite wind erosion damages.

How CRP works

The CRP (CRP website) operates as a land set-aside/retirement program under which the government, via the US Department of Agriculture (USDA), pays landowners incentives through 10- to 15-year contracts to change the use of specific lots of land for ecosystem service benefits (OECD 2010).

The CRP is now a major payment for ecosystem services (▶ Chap. 131, "Payments for Ecosystem Services") scheme, taking the form of a nationwide land retirement program that represents win-win outcomes for landowners and the American public. Under this program, the US government offers landholders incentives to change land use on highly erodible and environmentally sensitive cropland and pasture in order to secure a targeted range of desirable ecosystem services, effectively acting as a "broker" between landowners and the tax-paying public who will benefit from a range of ecosystem service enhancements.

The "sellers" in this PES market are land-owning farmers who, to be eligible for CRP enrolment, must hold cropland suitable for planting or else marginal pastureland suitable for use as a riparian buffer or similar water quality enhancement purpose. Furthermore, this potential "service provider" must have owned or operated the land for at least 12 months (with some allowance for extenuating circumstances). The "buyer" in this PES market is the US government, comprising four main agencies led by the United States Department of Agriculture (USDA), purchasing environmental services on behalf of the American tax-paying public who comprise the ultimate beneficiaries. The PES market and payment mechanism operates through the Commodity Credit Corporation (CCC), an intermediary corporation controlled by the USDA with whom landowners enter into contracts when enrolling in the program. The other federal players include the Farm Service Agency (FSA: the administrative body that runs the program for the USDA) and the Natural Resources Conservation Service (NRCS) which serves as the technical agency supporting CRP through to implementation on private lands.

General sign-ups to the CRP occur during specified enrolment periods, in which landowners compete nationally to enroll their land by submitting a bid for ecosystem services provided and the costs of management activities securing their provision. These competitive bids are then ranked according to an Environmental Benefits Index (EBI). The EBI quantifies overall benefits, integrating multiple services covering: wildlife habitat, water quality, on-farm production, long-term outcomes beyond the contract period, air quality, and scheme cost. The balance of benefits to costs forms the basis of prioritization, ranking all bids in comparison to each other in any one enrolment period. The FSA selects successful bidders from that ranking, and successful landowners are offered contracts lasting for 10–15 years, providing annual payments based on the agriculture rental value of the land.

This competitive bidding process, in which the landowner bids to maximize public benefit at the lowest cost, makes the CRP the largest and longest-running PES program utilizing inverse auctions in the world. This “reverse auction” process, in which farmers state the value they are prepared to accept for undertaking measures to safeguard or enhance ecosystem services, also minimizes transaction costs. It also focuses on measures and locations in which the greatest gains can be achieved, offering greater efficiency than simpler area-based payments. The repeated offerings mean that the scheme can be adapted to changing knowledge and needs, for example, enabling the targeting of additional ecosystem services each round or adaptively improving procedures to improve outcomes. Also, sellers have the option of whether or not to renew their contracts or to negotiate payments for different land parcels, providing them with flexibility. Contracts of 10–15 years also allow farmers to plan for the long term, assured of an income over the contracted period. Furthermore, target regions for CRP sign-up can be adjusted adaptively by Government to optimize delivery of desirable ecosystem services. The benefits produced by the CRP are genuinely additional, as estimates suggest that 51% of CRP land would be returned to crop production in the absence of CRP payments with a corresponding decline in expenditure on outdoor recreation of as much as $300 million annually in rural areas (Sullivan et al. 2004).

An additional continuous sign-up pathway allows landowners potentially contributing to an identified range of high-priority conservation practices to enroll at any time without competition. However, over 80% of the CRP land is registered using the competitive bidding process. Reflecting ongoing priorities such as food production and rural employment, CRP enrolment is capped at 25% of cropland within any one county and an overall limit of 36.4 million acres (14.7 million hectares) across the country at any one time.

Once contracts are arranged, land management practices are put in place by the landowner. CRP participants are provided with annual rental payments in return for establishing long-term, resource-conserving measures. Cost-share assistance may be provided to participants who establish approved cover on eligible cropland for up to 50% of participants' costs in establishing approved practices. Additional financial incentives of up to 20% of the annual payment for certain continuous sign-up practices may also be offered. If a landowner wishes to terminate a contract early, they are required to pay a penalty fee of 25% on rental payments paid, plus repayment, with interest, of all the funds already paid.

The farmer benefits directly, not merely through additional secure income sources, but also indirectly via reduced soil erosion and greater productivity due to improved air, water, and soil quality. Improved environmental conditions indirectly benefit people living in the local area and downstream in target catchments. The CRP has been found to achieve improvements in water quality and biodiversity, particularly for birds, through reduced pollution from agricultural runoff and increased/improved habitat. Since 1982, the FSA estimates that the CRP has reduced soil erosion by 454 million tons per year; restored 2 million acres (0.8 million ha) of wetlands; sequestered 48 million tons of carbon per year; established 3.2 million acres (1.3 million ha) of wildlife habitat; improved water quality with annual reductions in sediment (220 million tons = 200 million tons); substantially reduced nitrogen and phosphorus in run-off; increased wildlife populations (including, for example, increasing prairie pothole duck populations by 30%); and reduced flood damage.

The success of the CRP has served as a blueprint for similar projects in many other countries.

Future Challenges

Setting CRP rental rates is made more complex by fluctuating commodity prices, the commodity price spike of 2008 resulting in the value to landowners of production being significantly greater than returns from CRP rental rates. Furthermore, low rental rates may result in a correspondingly low recruitment rate of new CRP entrants as well as renewals of existing CRP contracts.

Although soil erosion is now a less significant problem in the USA than it was in the 1970s, the CRP contributing to a substantial reduction, many US rivers, streams, and wetlands remain critically depleted or polluted and biodiversity loss continues overall, despite some successes. This demands ongoing improvements to the CRP in addition to other wider measures to protect these critical natural resources. It is likely that increasingly complex requirements to address more ecosystem services may increase transaction and monitoring costs. Conversely, advances in modeling and data collection could improve the CRP's cost and environmental effectiveness.

Land retirement under the CRP can also contribute to fewer farming-related jobs, lower agricultural production, and inflated land rental rates. However, employment impacts may be reduced by diversification into opportunities provided by enhanced services, including hunting, fishing, and other recreational industries.

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Contribution of Wetlands to the Food-Water-Energy Nexus

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Mark Everard

Contents

Abstract

Since 2010, there has been increasing international interest and activity around "the nexus." The most common articulation of "the nexus" is recognition of the interlinked issues of water, food and energy (including the challenges of a changing climate). This is often considered in the context of emancipation of the "bottom billion" of the world's population from multiple, interlinked dimensions of poverty, as expressed in the Millennium Development Goals and their successor Sustainable Development Goals, but also in terms of the resilience of the developed world. The nexus approach recognizes the practical impossibility of effective management of each of these inherently interlinked elements in isolation, representing a practical step from a historically, narrow focus into a more systemic basis for management and decision-making.

Keywords

Energy · Water · Food · Climate change · Systems · Integration · Livelihoods

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Introduction

Since 2010, there has been increasing international interest and activity around "the nexus." The most common articulation of "the nexus" is recognition of the interlinked issues of water, food and energy (including the challenges of a changing climate). This is often considered in the context of emancipation of the "bottom billion" of the world's population from multiple, interlinked dimensions of poverty, as expressed in the Millennium Development Goals and their successor Sustainable Development Goals, but also in terms of the resilience of the developed world (Hoff 2011).

Systems and the Nexus

Wetlands are of substantial importance for the supply of a wide diversity of beneficial ecosystem services, of which the principal components of "the nexus" (water, food and energy) are a subset. By focusing more narrowly on this subset of interconnected services, representing immediate challenges facing both the developing world as well as the ongoing sustainability of the industrialised world, the concept of the nexus has gained considerable political as well as business traction at international, regional and national scales.

Politically, the nexus is seen as playing an important role in increasing resource efficiency, reducing trade-offs, building synergies and improving governance across sectors, emphasising the inevitable linkages between these different aspects (Allouche 2014). For example, unless there are significant changes to the ways that humanity produces and consumes, agricultural production will have to increase by about 70% by 2050, and about 50% more primary energy has to be made available by 2035; these increases would have far-reaching implications for water and land resources, with climate change acting as a further aggravating factor (Hoff 2011). These threats apply equally to rural people living most directly resource-dependent lifestyles as to the viability of developed world enterprises, built environments and the ongoing wellbeing of people living within them. The UN in particular is expressing interest in the nexus (for example UN Water 2014).

The nexus concept is also beginning to influence the thinking of some leading industrial sectors, which recognise that water, food and energy are pieces of the same puzzle and that it is therefore not practical to look at them in isolation (World Business Council for Sustainable Development 2009). Water shortages, for example, are recognised as posing both a direct resource threat as well as compromising supplies of food and farmed goods, the viability of water-intensive business activities such as paper pulping, irrigated farming and mining, and affecting power production.

Wetlands and the Nexus

Wetlands play multiple, interrelated roles across the nexus. These include significant roles in producing and regulating aspects of the nexus, as well as mitigation of impacts upon them.

Wetlands and Water

Wetlands play a key role in water capture, storage and purification, supporting a range of livelihoods from subsistence needs to industrial and power production. They also buffer extremes of flow brought about by catchment conversion for food production and other forms of land appropriation for forestry, urban, industrial or infrastructure purposes, as well as hydrological anomalies resulting from extremes of weather driven by climate change. Wetlands also process pollutants introduced into water systems by food production and other human activities.

Wetlands and Food

Wetlands are exploited directly for a range of food from fish to plant, wildfowl and other items. This includes both wild harvesting, for example capture fisheries or foraging, as well as modifications to increase the efficiency of food production, such as flood-retreating agriculture and finger ponds to trap fish. Wetlands are also frequently converted for more intensive food production, which is one of the principal drivers of the global loss and degradation of wetlands and the diversity of interconnected services they provide (including the narrower set addressed by the nexus). Wetlands also serve important roles in regenerating soils and cycling nutrients, supporting the productivity of connected terrestrial and open water habitats. There are clearly multiple conflicts between the conversion of wetlands for the narrow purpose of food production and the wider set of ecosystem services that they potentially provide.

Wetlands and Energy

Wetland products such as timber, peat and reeds may be used for both home- and industrial-scale energy production. At industrial scale, conversion of wetlands for palm oil production to supply biofuel is placing major pressures on tropical peatlands, which are being drained with the oxidation of peat then releasing substantially more sequestered carbon than is achieved in reductions in vehicle emissions. This illustrates the need for systemic thinking and policy-making, if one narrowly framed good intention (a mandatory biologically derived component in vehicle fuels to reduce net emissions of fossil fuel-related greenhouse gases) is not to result in wider unintended negative consequences for related greenhouse gas emissions as well as disruption of natural ecosystems and water cycles, food production and connected livelihoods.

Water flows are increasingly also being captured by dams and other forms of impoundment for multiple purposes including hydropower generation. Today, hydroelectric energy accounts for approximately 18% of the world's electricity supply (World Bank 2014). Brazil produces more than 90% of its electricity from hydropower projects. Whilst hydroelectric power is inherently renewable, plugging into natural fluxes of energy within the water cycle, it is not however automatically sustainable due to a wide diversity of connected environmental and social impacts. Compared to traditional fossil fuel-based generation, hydropower generation has a different profile of contribution to climate change, air pollution, acid rain and ozone depletion. However, the containment of natural flows also automatically constrains flows of the other important things that water carries – nutrients, sediment, organisms and many more attributes besides – that can have adverse implications for the production of food and energy, access to clean water resources and the ecosystems that provide them downstream in catchments and in the wider landscape. In addition, net claimed greenhouse gas savings from hydropower generation are substantially undermined by methane emissions from anaerobic habitats in deep reservoirs. Although formerly largely ignored, these methane emissions may be highly significant; as much as 104 +/- 7.2 Tg (million metric tonnes) of methane are produced each year from the world's large dams based on ICOLD (International Commission on Large Dams) inventories (Lima et al. 2007), contributing as much as 4% of global greenhouse gas emissions (International Rivers 2007). Although the debate surrounding this issue is vociferous, it highlights conflicts between the conversion of catchments and other wetland systems for the narrow purpose of energy generation and the wider set of ecosystem services that they provide.

Wetlands and Climate Change

As the above considerations demonstrate, the roles of different wetlands types in regulating but also mitigating climate change are as diverse as they are potentially substantial. Some forms of wetland, such as peatlands and wet forests, are amongst the most efficient habitats in terms of carbon sequestration, but equally will release substantial amounts of stored climate-active gases if degraded. They also offer additional beneficial services such as the buffering of spate flows resulting from climate change-induced factors.

Wetland systems can also be significant in affecting microclimate, as one facet of how their associated ecosystem services potentially contribute to multiple dimensions of the quality of urban environments.

Challenges

The nexus concept has been useful in introducing systemic thinking into areas of policy and business decision-making. Progressive thinking about these necessarily systemic connections is to be encouraged.

However, one weakness of debates about the food-energy-water nexus is that high-priority issues apparent at global and national scales may not reflect local concerns. As one example, water and energy security allied with boosting food security through irrigation are often central arguments behind the instigation of large dams and inter-basin water transfers, yet these heavy-engineering solutions can undermine the livelihoods and rights of often large numbers of local people (World Commission on Dams 2000). This is due to impacts expressed not merely at sites inundated on dam filling, but at far broader scales as a result of disruption of flows of water, solutes, sediment, nutrients, organisms, energy and habitat-forming processes as a result of the impoundment, as well as obstructions to both aquatic and terrestrial migratory species (Everard 2013). This dichotomy of perspectives at different scales can be compounded by monetisation of benefits, which may not take account of how local people connect with the services provided by wetland, water and other natural resources in their day-to-day lives. So the nexus approach is helpful, but is not a substitute for participatory approaches to the management of wetlands and other natural systems.

Nexus thinking is driving awareness of the interconnectedness of different facets of human activities, resource dependencies and vulnerabilities. The key challenge is to continue to accelerate the uptake of increasingly systemic thinking across all policy areas, business decision-making and levels of governance.

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Economic Incentives for the Nonregulatory Conservation and Management of Wetlands

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Bill Watts and Mark Everard

Contents

Abstract

The Ramsar Convention defines wetlands as: "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres" (article 1.1). Many of the ecosystem services provided by these diverse wetlands are public goods. This means that the goods are not saleable in normal markets constituting willing buyers and sellers. Furthermore, since beneficiaries cannot be excluded from deriving benefits from wetland ecosystem services, they have no incentive to pay. Equally, if the resource owner ("seller") cannot sell the service, they also lack an incentive to maintain a supply of publicly beneficial services. Thus, since many wetlands are private as distinct from public property, there is no compelling

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market incentive to prevent the owner draining or in other senses destructively employing the wetland to maximize private profit. There is an emerging political preference for 'payment for ecosystem services'-type approaches to ecosystem management as a means to encourage value realization from the ecosystem services provided by wetlands and other habitats, many of which have historically been omitted from both markets and wider societal decision-making.

Keywords
Payments for ecosystem services · Market failure · Ecosystem services · Purification processes · Systemic solutions · Optimization · Offsetting · Club goods

Introduction

The Ramsar Convention defines wetlands as: "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres" (article 1.1). Many of the ecosystem services provided by these diverse wetlands are public goods. This means that the goods are not saleable in normal markets constituting willing buyers and sellers. Furthermore, since beneficiaries cannot be excluded from deriving benefits from wetland ecosystem services, they have no incentive to pay. Equally, if the resource owner ("seller") cannot sell the service, they also lack an incentive to maintain a supply of publicly beneficial services. Thus, since many wetlands are private as distinct from public property, there is no compelling market incentive to prevent the owner draining or in other senses destructively employing the wetland to maximize private profit.

The Current Status of Market Failure

This externalization of many of the beneficial services that wetlands supply is a market-driven tragedy. This market failure explains why, historically, the world has lost much of its resource of wetlands and other productive habitats. This is augmented by perverse agricultural policies, often driven by a mixture of anachronistic priorities and capture by land-owning interests. For example, "food security" is commonly evoked as a rationale for management of wetlands for narrow self-interest, maximizing food production from ecosystems for which the diversity of other potential services are regarded as valueless because they can't be sold.

The long-term consequence of this is that many wetlands throughout Europe have been destroyed or severely degraded over the last 100 years through such mechanisms as drainage for agriculture and pollution (Jones and Hughes 1993). For coastal wetlands, this is exacerbated by the phenomenon of "coastal squeeze." This is widely witnessed throughout coastal Europe including the East Coast of the United Kingdom where the coastline sinks while sea levels rise, constraining the extent of intertidal wetland margins as they abut hard flood and sea defenses and infrastructure such as port facilities.

Some services are captured by regulatory mechanisms, for example, the EU Habitats Directive, the Wild Birds Directive, and aspects of the Water Framework and Marine Framework Directives. However, implicit in the protection of culturally favored services (particularly habitat for wildlife) is that the value of designated sites is infinite while that in a nondesignated site continues to be unvalued, or at best substantially undervalued. The challenge then is to develop economic incentives for the conservation and management of wetlands, without obstructive regulatory intervention.

The Scale of Current Economic Loss

An indication of at least some of the loss which society has allowed through the destruction of wetlands is provided by TEEB (The Economics of Ecosystems and Biodiversity), which gives minimum and maximum ecosystem values for a number of wetlands types, drawing upon value-transfer based estimates of a "basket" of the services that they provide (TEEB 2013) (TEEB calculations include an estimate of the potentially excludable category of provisioning services – those from which public benefit can be excluded – but for all wetland types this is a small part of the aggregate benefit).

TEEB calculations find a remarkable variation for all habitat types assessed, including: for coral reefs from $US14 to $US1.2 million per ha per year; for coastal systems from €248 to €80k; for mangroves and tidal marshes from $2k to $215; for inland wetlands, including peatlands, from $1k to $45k; and for rivers and lakes from $2k to $13k. (All TEEB figures are expressed in 2007 values.) These large variations in estimates are not surprising due to context dependency. As with housing, wetland location is a highly influential factor. Values deduced for inland and coastal wetlands depend on how many people live close by, the existence of substitute habitats (the more that there are in an area the less valued each is individually), and the income of people living close by. There is also "devil in the detail" with respect to a range of connected issues, such as the exchange rate used in calculations, Purchasing Power Parity (the strength of a particular currency for purchasing equivalent goods), and what is included and excluded from the estimates of service values. However, variance is buffered by the fact that, for most derived values, TEEB reports draw on a significant number of studies. For many wetland types, a median estimate might be a best default estimate of value taking account of a representative 'basket' of services, though care has to be taken in managing the transfer of values due to context dependency; one needs to consider all services in comparable terms.

Some valuation studies of the multiple ecosystem service benefits of coastal wetlands are highlighted in case studies elsewhere in this Wetlands Book, highlighting a number of UK coastal-managed realignment schemes. These include:

The Essex coast, addressed under the EU ComCoast project
Wareham, Dorset
Alkborough Flats, north Lincolnshire
The Steart peninsula, Somerset

Bringing Values into Markets and Decision-Making

A challenge remains to bring these diverse ecosystem service values into markets, or at least to ensure that they influence decision-making. This is important for a number of reasons, including:

- Ethically, exclusion of services from decision-making also excludes beneficiary groups leading to an uneven distribution of beneficiaries and victims of ecosystem management.
- Economically, while narrowly framed management solutions may yield benefits for target services, net impacts on the full spectrum of ecosystem services may have an overall negative benefit-to-cost outcome.
- Ecologically, key ecosystem processes may be lost resulting in ecosystem degradation, including resilience and their capacity to continue to provide important services into the future.

Mechanisms are therefore required progressively to assign values to wetland systems that are not currently regarded as profitable, but which are economic. We have seen early stages of this transition, for example, emerging recognition of the value of wetlands for the "natural" provision of flood risk benefits, as well as sequestering carbon as is beginning to be reflected in emerging carbon markets. Water service provider investment in catchment management to reduce contamination of raw water as a cheaper solution than cleaning up excessive pollution downstream is another example of an emerging market for the water purification services of catchment ecosystems, as exemplified by case studies on New York City's public water supply and the Upstream Thinking program (▶ Chap. 124, "Economics of Wetland Conservation Case Study: Catchment Management for Water Quality") and in the case for managed realignment for flood risk management (▶ Chap. 123, "Economics of Wetland Conservation Case Study: Learning from Managed Realignment").

The managed realignment case studies highlight the extent to which many services remain distant from markets and from decision-making forums. In the case of the Alkborough Flats managed realignment, a wetland system located close to both significant centers of population and historic sources of pollution, overlooked services such as pollution management may be extremely important though evading confident valuation due to a lack of quantification methods. The adjacent Humber estuary receives flows of water and associated pollutants from the heavily industrialized English Midlands, South Yorkshire, and Humberside areas. Nitrate, phosphorus, heavy metals, and micropollutants such as endocrine disruptors and persistent organic pollutants may have significant impacts on aquatic, terrestrial, and human systems including, for example, eutrophication and PBTC (Persistent, Bioaccumulative, Toxic, Carcinogenic as well as mutagenic and reprotoxic effects). Ongoing research led by the UK Water Industry Research group (http://www.ukwir.org.uk/site/web/content/home) demonstrates a high likelihood that many micropollutants will exceed the statutory limits set within the EU Water Framework

Directive (▶ Chap. 71, "European Union Water Framework Directive and Wetlands") throughout much of England and Wales, and it is logical to assume that these problems will also exist elsewhere in Europe and indeed much of the rest of the world with high urban densities. Wetlands may have an important role in addressing this emerging problem, as they can be highly effective in transforming, metabolizing, or sequestering nutrients, metals, and complex pollutants, in addition to providing efficient carbon sequestration processes. There is also potentially a strong economic case for investment in these wetland purification services due not merely to the regulatory status of many of these problematic substances but also the high expense (financially and in carbon and resource terms) and the unpredictability and need for continuous management of traditional end-of-pipe treatment technologies.

Scientific certainty about the efficacy of different wetland types to provide this service is an obstacle for wider uptake, as indeed is entrenched oversight of the unintended negative consequences for nonfocal ecosystem services in traditional, single-concern treatment technologies, but certainly warrants considerable further research. The same principle also applies to many of the services provided by wetlands which are currently not captured by markets or factored centrally into decision-making, including, for example, their roles in soil formation and erosion control, pest management, and landscape aesthetics.

Systemic Recognition of Wetland Ecosystem Services

Another of the current market failures, also reflected in failures of regulation and governance, is that ecosystem services (provided by all habitat types) tend to be regarded and managed in isolation. We have already explored how wetland systems have been degraded due to a narrow focus on food production, but the same principle also applies when they are regarded solely or largely on the basis of other "siloed" services such as their role purely as biodiversity resources, flood risk management assets, or where they are used solely to treat wastewater.

In fact, wetlands can, when managed optimally, provide multiple simultaneous benefits of substantial cumulative value. Everard and McInnes (2013) highlight the role of different wetland types, including, for example, integrated constructed wetlands, urban "green infrastructure," and washlands, as examples of "systemic solutions" (▶ Chap. 179, "Integrated Constructed Wetlands for Water Quality Improvement"). These "systemic solutions" are characterized by low resource inputs, working with ecosystem processes to deliver a portfolio of ecosystem services optimally (rather than maximizing one focal service but overlooking consequences for other services). Wetland management optimized to cover a range of services (amenity, flood risk regulation, habitat for nature, carbon sequestration, microclimate enhancement, green transport routes, noise, and visual buffering, etc.) can be of substantial cumulative benefit, also requiring little management compared to traditional engineered solutions and so representing greater net public benefit.

Of course, legacy regulations, institutional structure, and ring-fenced budgets can frustrate the quest for multibenefit outcomes, focused as they are on narrowly framed outcomes. Nevertheless, real economic gains for society can be achieved when ecosystem services are managed simultaneously, possibly by a consortium of institutions at the system level. This approach may also lead to innovative solutions to intractable, apparently unrelated social problems such as tackling obesity (by providing safe and attractive places near people for exercise) and addressing urban "heat islands," air quality concerns, and local flooding.

Market-Based Instruments

Market-based instruments can act to align stakeholder interests with conservation of wetlands and their multiple services. Use of market-based instruments has attracted a great deal of interest both in the United Kingdom and elsewhere. For example, the UK government explicitly recognizes the potential of market-based instruments to "mainstream" the value of nature into all spheres of societal decision-making (HM Government 2011a).

An early example of the use of market-based instruments was enacted in the United States wherein a developer may apply for a Section 404 permit (to discharge dredged or fill material into the navigable waters at specified disposal sites) issued under the Clean Water Act 1972, which introduces a process of offsetting wetland destructive development through the creation of mitigating essentially restored wetland elsewhere. Similar types of offset processes characterized by the provision of compensation for some form of loss have also been developed in Australia, Costa Rica, and Brazil and are actively being considered elsewhere including exploratory "biodiversity offsetting" schemes in the UK. These offsetting schemes provide payment for the continued maintenance, and potentially the provision of services, from a habitat which remains intact.

"Payments for ecosystem services" (PES) are another form of strongly emerging market-based instrument. PES is founded on the establishment of voluntary markets for ecosystem services, production of which depends upon habitat management that is additional to statutory obligations. The OECD (2010) estimated that there were 300+ PES schemes established globally and that number has escalated substantially since that time. The English government Department for Environment, Food and Rural Affairs (Defra) has an active environmental economics program including promotion of PES, including recent publication of a Payments for Ecosystem Services Best Practice Guide (Smith et al. 2013).

The Defra PES Best Practice Guide notes that:

> "The term PES is used to describe a wide variety of schemes in which the beneficiaries, or users, of ecosystem services provide payment to the stewards, or providers, of ecosystem services. In practice, PES is often used in reference to schemes which involve a continuing series of payments to land or other natural resource managers in return for a guaranteed flow of ecosystem services, or at least management actions likely to enhance their provision..."
>
> "...The basic idea behind PES is that those who provide ecosystem services – like any service – should be compensated for doing so. PES therefore provides a mechanism for

pricing historically undervalued ecosystem services such as water quality regulation and the provision of habitat for wildlife and, in doing so, brings these into the wider economy. The novelty of PES arises from its focus on the 'beneficiary-pays Principle', as opposed to the 'polluter-pays Principle.'"

The scientific knowledge necessary to introduce PES markets from wetland is substantially in place for issues such as carbon sequestration, nutrient cycling, pollution remediation, fisheries, recreation, and floodwater attenuation. Water service provider investment in catchment management to reduce contamination of raw water as a cheaper solution than cleaning up excessive pollution downstream referred to previously (see the case studies on New York City's water supply and the Upstream Thinking program ▶ Chap. 124, "Economics of Wetland Conservation Case Study: Catchment Management for Water Quality") are founded on PES principles, each constituting substantial exemplar programs based on "hard" bottom-line outcomes.

Notwithstanding intense interest in policy, academic, and business circles, Salzman and Ruhl (2002) caution that PES may not always represent an apparent win-win outcome for conservation and development, with significant challenges including ensuring that benefits are genuinely additional and also adequately addressing issues of moral hazard, social justice, and market size and market penetration. PES is no panacea, but it does constitute an additional form of policy instrument. Neither does the emergence of PES as a viable policy instrument abrogate responsibility for regulation. PES design and auditing is critical to avoid unintended negative outcomes for nontarget services and to avoid fraud; a legal process is required to describe the obligations and institutional mechanisms which permit and document transfer of obligations to third parties. Nevertheless, strong case studies around the world (see the water service provider catchment management case studies ▶ Chap. 124, "Economics of Wetland Conservation Case Study: Catchment Management for Water Quality") and progressively evolving "best practice" (such as Smith et al. 2013) are revealing significant benefits that may arise from well-targeted and designed PES markets.

Growing Interest in Internalization of the Value of Wetland Ecosystem Services

There is a global growth in interest in the internalization of the value of ecosystem services into the mainstream of societal decision-making, including those provided by wetlands. For example, the Aichi Targets, adopted by the Convention on Biological Diversity in 2010, contain the target "...by 2020, ecosystems that provide essential services, including services related to water, and [which] contribute to health, livelihoods and well-being, are restored and safeguarded..." (CBD 2013). The EU has adopted the target of halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020 and restoring them insofar as is feasible. Thus, in 2011, the European Commission adopted a new biodiversity strategy with a commitment to "...promote the development and use of innovative financing mechanisms, including market-based instruments" and states that "Payments for Ecosystem Services schemes should reward public and private goods from agricultural, forest and marine ecosystems" (EU 2011). In England, the Natural Environment White Paper, *The Natural Choice* (HM Government 2011a), emphasizes "...real opportunities for land managers to gain by protecting nature's services, and trading nature's benefits with businesses, civil society and the wider public sector." Further reference to the role of economic instruments to secure water services is contained in the UK Government Water White Paper, *Water for Life* (HM Government 2011b).

This stimulus is also evident among the UK's devolved administrations. A land use strategy for Scotland, *Getting the Best from our Land* (Scottish Government 2011), set out a series of principles for sustainable land use including explicit encouragement of opportunities for land use to deliver multiple benefits. The Welsh Government Green Paper, *Sustaining a Living Wales* (Welsh Government 2012), has a central tenet of moving to an Ecosystem Approach in environmental regulation and management, also highlighting "...genuine opportunities for the market to 'pay for ecosystem health and services'" and cites the example of corporate sponsors investing in woodland creation as a recognized method of sequestering carbon emissions.

Observations in the section above addressing the *Systemic recognition of wetland ecosystem services* are equally valid for PES schemes. Unintended conflicts with nontarget services may occur if there is too narrow a focus on one or a few favored new marketable services and their privileged beneficiaries. Addressing the multiplicity of ecosystem services provided simultaneously by wetlands and other ecosystems, the Defra PES Best Practice Guide (Smith et al. 2013) deals not only with the need to explore the impacts of proposed management on a systemic basis, but also explores opportunities for combining ecosystem services in markets through "bundling" (a single buyer or consortium of buyers pays for a package of ecosystem services), "layering" (multiple buyers pay separately for different ecosystem services), or "piggybacking" (not all ecosystem services are sold but there is a buyer for a single service, or possibly several services, with other services accruing to users free of charge).

Conclusions

Wetlands produce many ecosystem services, a substantial proportion of which are public or "club" goods. These nonmarketed services tend not to be remunerated and thus privately "profitable" when the habitat is owned as private property. Ecosystem services such as flood regulation, carbon sequestration, pollution management, fish nursery functioning, and landscape benefits may tend to be suppressed by private owners in favor of socially and environmentally suboptimal marketed goods and services.

There is a strong political preference for PES-type approaches to ecosystem management as a means to encourage value realization from the ecosystem services provided by wetlands and other habitats, many of which have historically been omitted from both markets and wider societal decision-making. However, an ecosystem service market regime should provide accountability and transparency, be subject to monitoring, be fungible both in the sense of service definition and substitution, and also allow for public participation.

There may be examples where it may be cheaper to acknowledge that, for critical wetland habitats such as intertidal habitats which provide coastal protection among a wide range of other publicly beneficial services, public ownership and collective management may be favorable to private ownership permitting the optimization of multiple, publicly beneficial services rather than maximization of a few privately beneficial goods.

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Economics of Wetland Conservation Case Study: Learning from Managed Realignment

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Abstract

The term "coastal squeeze" describes the loss of intertidal wetland habitat, a combined result of a sinking coastline and rising sea levels constraining the extent of intertidal wetland margins as they abut hard flood and sea defenses and infrastructure such as port facilities. This is driving innovation in the management of coastal flooding through such techniques as managed realignment, where land formerly "reclaimed" for agriculture and other uses is allowed to revert to intertidal wetland habitat. As these habitats reform, a range of beneficially and formerly undervalued services such as flood risk regulation, habitat for wildlife including fishery recruitment, nutrient cycling, and characteristic landscapes are

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restored. This chapter draws lessons about value creation by the recreation of these various ecosystem services drawing upon studies of four managed realignment schemes.

Keywords
Blackwater · Wareham · Managed realignment · Flood defense · Coastal defense · Cost benefit analysis · Stakeholders · Ecosystem services · Fisheries · No net loss · Coastal squeeze · Missing markets

Introduction

The term "coastal squeeze" describes the loss of intertidal wetland habitat, a combined result of a sinking coastline and rising sea levels constraining the extent of intertidal wetland margins as they abut hard flood and sea defenses and infrastructure such as port facilities. This is driving innovation in the management of coastal flooding through such techniques as managed realignment, where land formerly "reclaimed" for agriculture and other uses is allowed to revert to intertidal wetland habitat. As these habitats reform, a range of beneficially and formerly undervalued services such as flood risk regulation, habitat for wildlife including fishery recruitment, nutrient cycling, and characteristic landscapes are restored. This chapter draws lessons about value creation by the recreation of these various ecosystem services drawing upon studies of four managed realignment schemes. Dixon et al. (2007) provides a summary of early experiences with managed realign-
38 ment and the steps needed to optimize habitat creation.

Case Study 1: The ComCoast Project

The EU-funded ComCoast project (ComCoast 2007) investigated the multifunctional ecosystem services provided by intertidal habitats in Northern Europe, exploring more defensible and sustainable approaches to coastal flood defenses including "soft" land management approaches rather than hard engineered barriers. The research analyzed cobenefits for the whole coastal community, in the construction of environmentally and economically sound Flood Risk and Coastal Defence solutions covering a range of ecosystem services, such as fish recruitment, carbon sequestration, water quality, and cultural services.

ComCoast was funded under the EU North Sea Interreg IIIb Programme with key partners in the Netherlands, England and Wales, Germany, Belgium, and Denmark. It included three UK-based PhD studentships examining: (1) fish utilization of managed realignment sites, (2) nutrient capture and carbon sequestration on managed realignment sites, and (3) the economic case for a more integrated approach to estuary management, drawing on the results of the other two science-orientated PhD studies. These PhD studies were based in a number managed realignment sites and mature saltmarshes in the Blackwater and Roach & Crouch estuaries in Essex, in the East of England.

The fisheries research strand of ComCoast (Fonseca 2009) highlighted the links between changes to physical habitat and the resulting societal benefits arising from increased fish stocks relevant to both commercial and recreational fishing. Early work summarized by Colclough et al. (2005) had demonstrated a close association of the early life stages of important commercial species such as sea bass (*Dicentrarchus labrax*) with the existence of high marsh, confirming the provisional findings of French authors (such as Laffaille et al. 2001). These observations are important given the requirements of the EU Water Framework and the Marine Strategy Framework Directives. Fonseca et al. (2011) describe highly dynamic and pronounced seasonal use of the habitats under study, also providing detailed evidence of the early feeding of the bass fry in newly created habitats. An attempt was made during the study period to quantify fish utilization.

The fish survey data fed into an economic analysis, though the weight of this evidence was restricted by the size and amount of time available for fish sampling. This may also help explain the relatively small component the conservative fishery quantification represented in the overall valuation (Luisetti 2009). Clearly, multiyear quantitative data sets would have added more analytical power. Furthermore, commercial (and not recreational) value data were only available for sea bass. Other marine fish species with commercial and/or recreational significance were also observed exploiting experimental saltmarshes, but no valuation data were available for these. (This issue of partial valuation of fish recruitment is covered in detail by Vieira da Silva 2012.) More recent studies of juvenile fish on some of the same Essex sites have also demonstrated high site fidelity over the first summer for young of year sea bass and other species (Green et al. 2012).

Conclusions drawn in the economic study of fish recruitment services were thus low, omitting most species and the higher-value recreational services provided by the affected realignment. If additional benefits arising from fish recruitment on intertidal wetlands can be substantiated scientifically, beyond observation (for example by S. Colclough & W.Watts, pers.obs.), there could well be a case for public funding for the construction or rehabilitation of intertidal habitats through funding mechanisms such as the EU Common Fisheries Policy. Quantification of cobenefits such as coastal flood regulation, amenity, and landscape aesthetics may open up other potential cofunding streams and may justify realignment, even where the potential fishery service is not sufficient, in itself, to make the case.

The nutrient and carbon cycling research strand of ComCoast found that carbon (C) and nitrogen (N) contents in natural intertidal sediments were higher than in managed realignment sediments (Adams 2008). However, mature managed realignment sites possessed C and N burial rates at least as great as natural marshes and, if increased sedimentation in these predominantly low-lying intertidal areas is accounted for, mature managed realignment sites far outstrip natural marsh C burial rates.

Both natural and managed realignment intertidal wetlands were also found to be small sources of the greenhouse gases methane and nitrous oxide (each with a stronger climate-forcing effect than carbon dioxide), offsetting to a limited extent the net climate regulation benefits of carbon sequestration. The observed current carbon sequestration rate at the Blackwater estuary managed realignment sites was ~690 tonnes (metric) of CO_2eq year^{-1} (carbon dioxide equivalents per year).

The economics research within ComCoast highlighted some of values of ecosystem services provided by saltmarshes. It deduced a "willingness to pay" among members of the public polled under the ComCoast program for the creation of new saltmarsh in the Blackwater estuary, accounting for the existence of other saltmarshes in the area and the potential substitution of these alternative saltmarshes by the public in the delivery of wetland services, in their valuation of the worth of a new realignment. Saltmarsh was considered important for amenity and recreation as well as for biodiversity both in terms of access to the site (use value) and improved environmental quality (nonuse value). While there was general public support for saltmarsh recreation, people felt uncomfortable having them too close to their houses. This may be linked to nuisance associated with visitors, though other studies suggest that having water closer to their homes is perceived as a greater risk even though, hydrologically, the risk of flooding in actual fact may be decreased by these now closer but larger floodwater storage areas.

Luisetti (2009) synthesized this "willingness to pay" data with physical science-based fisheries and biochemistry data, producing an overall assessment of the value of various realignment options within the Blackwater estuary. This subsequent study, based on the Composite Environmental Benefit variable which includes both use and nonuse value, found that irrespective of the economic discount rate applied in cost-benefit analysis, a substantial scheme of realignment within the Blackwater was justified.

Potential air quality and water pollution control benefits may also arise from habitat recreated under managed realignment explored in the ComCoast study. However, these purification benefits were not reflected in the benefits elicited, due to scientific uncertainty about the magnitude of these effects. The absence of such data is of course symptomatic of the often low public awareness of far-field environmental effects, even though there may well be substantial potential health benefits to be gained from realignment upwind of urban areas.

The ComCoast project extended knowledge about many of the ecosystem services associated with intertidal habitat recreation through managed realignment, highlighting the need to include all ecosystem services into optimal decision-making about sustainable coastal defense strategies.

Case Study 2: Wareham

The first instance where the ecosystem valuation approach was applied to Flood Risk Management within the UK was at Wareham in 2006 (summarized in Defra 2007). Wareham is located close to two adjacent estuaries of the rivers Frome and Piddle where they enter the western end of Poole Harbour, Dorset, an area of international importance for nature conservation. These estuaries were lined by 50-year-old flood banks, providing some protection against flooding for 42 properties and 400 ha of poor quality grazing marsh.

As the flood defenses were nearing end-of-life, various options were considered for their future management. These included "No Intervention" and "Do Minimum," which would have put people and property at risk as well as adversely affecting existing nature conservation interests, navigation, and rights of way. These issues could have exposed the regulatory authority (the Environment Agency) to litigation. A "Hold the Line" scenario considering rebuilding defenses to maintain the status quo was also found to be economically unsustainable, the expenditure not meeting the Cost Benefit Analysis-based funding criteria of Defra (the relevant UK government department). This left various managed realignment options.

Economic valuation of outcomes for ecosystem services supported the managed realignment approach, setting back smaller-scale flood defenses and allowing the recreation of intertidal habitat. This was supported by local stakeholder dialogue using a simple semiquantitative scoring system to assess the likely magnitude of changes, from significantly positive to significantly negative or unknown, for a range of ecosystem services as a result of these different options. Although the Wareham assessment was equivocal in terms of absolute economic benefits, it indicated a broader balance of costs and benefits associated with a particular managed realignment option and highlighted where further information or research was necessary.

The Wareham study showed that constructive engagement with a wide range of stakeholders is essential to build understanding and support for the results of the process. This was helped considerably by expressing benefits in readily understood ecosystem services terms (amenity areas, protection of farmed land, etc.) evaluated via a simple semiquantitative ecosystem valuation process.

Case Study 3: Alkborough Flats

Everard (2009) led an ex-post ecosystem service analysis of the Alkborough Flats managed realignment on the Humber Estuary. The realignment site includes around 440 hectares of farmed land which was to be periodically inundated on high tides through an engineered breach in the former flood bank. The breach is tightly controlled to protect the adjacent navigation channel that forms an important trading route up the rivers Ouse and Trent. This realignment project was driven by EU Habitats Directive requirements to ensure "no net loss" of habitat, caused both by sea level rise-induced "coastal squeeze" and development elsewhere in the estuary. It is also a flood defense scheme, providing flood storage and thus allowed a delay in the construction of new flood defense structures upstream, yielding a significant benefit through delayed construction costs.

The need to protect the navigation route and create substitute habitat led to suboptimal outcomes from a narrowly flood defense perspective, though the traditional property-saving benefits of the scheme totaled £12.3m over the likely life of the project, which was marginally greater than the aggregate of site acquisition and development costs of £10.2m. Evidently potential flood risk and habitat-related benefits alone would not normally have affected the management of the site by a private owner.

However, the wider lessons from the ecosystem services valuation study reveal a more than doubling of benefit cumulatively from changes in various other ecosystem services, with an approximate aggregate benefit of £23m. This figure excludes some potentially significant values that eluded monetization, including air quality regulation and fish recruitment. These omissions highlight the "missing markets" phenomenon and absent scientific data.

The maximization of marketed outputs prior to realignment to achieve wider public benefits (principally flood risk management and habitat for wildlife) suggests that "provisioning service" values should have fallen after realignment. In fact, Everard (2009) found that the fall in provisioning services related to declining arable production was approximately balanced by gains from rare breed grazing (including sales of meat, wool, breeding stock, and other products). Therefore, it is at least sometimes possible to achieve "win-win" outcomes across ecosystem services when innovative ideas and practices are deployed. This runs contrary to many established assumptions about "win-lose" trade-offs being inevitable when realignment takes place.

Case Study 4: Steart Peninsula

Similar conclusions about positive-sum gains from realignment were drawn in a study of ecosystem service outcomes likely to result from managed realignment of the Steart Peninsular in Somerset, UK (Vieira da Silva 2012; Vieira da Silva et al. 2014). This study demonstrated that "*A conservative, yet considerable, net annual benefit range of £491k to £913k has been deduced*" from combined changes in services.

In agreement with the findings of other ecosystem services case studies, Vieira da Silva (*ibid.*) also notes that many "*. . .research and knowledge gaps exist with respect to several ecosystem services, particularly the supporting services for which market values are clearly elusive. This is of concern as it affects our current ability to quantify and/or value them, and hence to include these important aspects of ecosystem integrity, functioning and resilience into decision-making. . . ...These gaps therefore demonstrate a clear need for further research, both theoretical and through long-term monitoring of these schemes once established.*" Several areas of necessary further research were identified, including the contribution of intertidal habitats to fish recruitment, the net contribution of recruitment to local and national fish stocks, values associated with potential alternative farming systems (shellfish, salicornia, etc.), microclimate effects of coastal wetlands, nutrient burial studies, and the effect of intertidal habitats on micropollutants.

Conclusions

Studies at these four managed realignment sites underline that many of the problems we have with assigning values to wetland systems are not economic, but are a function of absent underlying science as well as societal recognition of these values. Nevertheless, the absence of markets for these often unrecognized services means they are not valued and often, to the extent they are thought about at all, deemed valueless. These four case studies illustrate that these values can be significant and, with the correct policy and/or market instruments, can be exploited for net societal gain.

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Economics of Wetland Conservation Case Study: Catchment Management for Water Quality

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Contents

Abstract

Traditional approaches to the abstraction and treatment of water for public supply depend on a range of engineered approaches including dams, interbasin water transfers, and groundwater abstraction, generally with a high reliance on energy- and chemical-intensive treatment processes post abstraction. Substantial energy inputs, associated climate-active gas emissions, chemical inputs and linked supply chain concerns, waste generation including disruptive vehicle movements, and maintenance activities mean that the sustainability of this approach is questionable. These concerns are allied to the risks associated with declining water availability resulting from less predictable weather patterns. Overcoming these problems stemming from a largely reactive and piecemeal approach to water management, a more systematic and ecosystem-based approach is gaining momentum. This takes the form of a catchment-based approach, emphasizing the benefits of controlling contamination of water at source rather than committing greater retrospective investment in more intensive cleanup of water

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abstracted lower in catchments. This transition is driven largely by water quality considerations, though water quantity concerns are increasingly also influencing planning.

Keywords
Payments for ecosystem services · Trusted broker · Upstream thinking · Raw water · Farming · Westcountry rivers trust · Win-win · New York city · Water supply · Watershed protection

Introduction

Traditional approaches to the abstraction and treatment of water for public supply depend on a range of engineered approaches including dams, interbasin water transfers, and groundwater abstraction, generally with a high reliance on energy- and chemical-intensive treatment processes post abstraction. Substantial energy inputs, associated climate-active gas emissions, chemical inputs and linked supply chain concerns, waste generation including disruptive vehicle movements, and maintenance activities mean that the sustainability of this approach is questionable. These concerns are allied to the risks associated with declining water availability resulting from less predictable weather patterns.

These problems stemming from a largely reactive and piecemeal approach to water management have led to calls for a more systematic and ecosystem-based approach. This has been translated into a catchment-based approach, emphasizing the virtues of controlling contamination of water at source rather than greater retrospective investment in more intensive cleanup of water abstracted lower in catchments. This transition is driven largely by water quality considerations, though water quantity concerns are increasingly also influencing planning. Although leadership has often been shown by sectors such as NGOs and the water industry, the catchment-based approach is increasingly reflected in government strategies (including, for example, the UK's emerging catchment-based approach: Defra 2013).

Payments for ecosystem services (PES) approaches are emerging rapidly across the world, with markets for water services a common type of novel market. This chapter addresses two case studies – the "Upstream Thinking" program in the south west of England and the New York City water supply – to illustrate how the economics of wetland functioning at landscape scale are being brought into the mainstream of water management activities.

Case Study 1: Upstream Thinking, South West England

Water resource challenges in the south west of England include a shortage of groundwater due to widespread impermeable geology, allied with a substantial rise in population during the summer vacation season when water resources are at their most stressed. This leads to a heavy dependence on abstraction of surface water for public supply, both directly from reservoirs as well as from rivers augmented by releases of stored water. Both sources are vulnerable to contamination from farming activities in this predominantly rural English region.

The "Upstream Thinking" program (http://www.upstreamthinking.org/) is founded on understanding and protecting the functioning of catchment ecosystems as the basis for production of clean water and a range of co-benefits. These have intimate interactions with the practices and economics of working farms which comprise the overwhelming bulk of the catchment area. Upstream Thinking was developed as an innovative, partnership-based program agreed with government under the 2010–2015 UK water industry investment period. It operates by directing a proportion of water service revenues from customer bills as incentives for farm improvements that go beyond statutory requirements, or in other words a classic PES scheme.

The key players in the Upstream Thinking PES market are South West Water (SWW: www.southwestwater.co.uk), the regional water utility, and the Westcountry Rivers Trust (WRT: www.wrt.org.uk). WRT was set up as an environmental charity in 1995 to protect and improve the fresh waters in England's west country. The Trust has, since its inception, developed strong relationships with farm businesses, helping them save money on chemical, water, waste, and habitat management in ways simultaneously advantageous to river health. SWW has long recognized the benefits of working with WRT as a cost-effective means to protect water resources through reducing contamination from farmed land. Under the Upstream Thinking program, WRT serves a key intermediary role between the water company and the numerous farm businesses dispersed throughout key catchments, acting as a "trusted broker" simultaneously addressing practical farming priorities and the vitality of the wider catchment ecosystem. The WRT works with farm businesses to target investment and advice to where it can most efficiently contribute to improvements in catchment (water) quality and quantity. This reduces both the costs and the carbon footprint of water supply. The WRT approach includes the subsidization of a wide range of measures that achieve simultaneous farming and river ecosystem benefits, many of them protecting wetland habitat and associated beneficial functions. Measures include fencing stock away from river banks and wetlands, separating "clean" roof water from "dirty" farmyard water, relocating gates, armoring or bridging stream crossings to avoid silt ingress, and better storage facilities and advice on more economical, targeted application of fertilizers and pesticides.

In calculations audited by Ofwat (the economic regulator of the water and sewerage sectors in England and Wales), Upstream Thinking is expected to deliver a substantial 65:1 benefit-to-cost ratio based on projected improvements to raw water quality and consequent savings on treatment costs alone. This benefits the water industry, including its bill-paying customers. However, multiple additional co-benefits arise through the focus on improved catchment functioning and health. Protection of biodiversity is prominent among these co-benefits, including rare species and salmonid fish stocks which are themselves a valuable resource for angling, as well as enhancement of ecotourism, landscape aesthetics, and the security of the rural economy.

This is "win-win" result, increasing net farming revenue while also saving money for the water utility with savings passed on to customers, achieved in substantial part through targeted protection and restoration of catchment wetland functioning.

Case Study 2: New York City Public Water Supply

A prominent and substantially cost-efficient American example of a PES-based approach to catchment management for water resource protection is seen in the New York City public water supply (see review by Everard 2013). The Department of Environmental Protection of the City of New York delivers over 1.2 billion US gallons (4.5 billion liters) of water daily to 9 million people.

While early residents drew water from local sources, urban growth necessitated accessing remote sources. In 1905, New York City identified the Catskills Mountains as a prime water source, constructing various reservoirs and dams throughout the Catskills through to 1928, also turning in 1927 to additional sources in Delaware County. The Catskills–Delaware system, known as the Cat/Del system, was implemented in stages between 1937 and 1964, today providing New York City with the largest unfiltered surface water supply in the world. However, by the 1980s, industrial-scale agriculture and forestry, tourism, and residential and industrial development posed increased risks to raw water quality, while increasingly stringent public health standards were coming into force. The vast cost of installing expensive filtration plant – $US 4–6 billion (£2.1–3.2 billion at the then-current exchange rates) in capital costs plus annual running costs of more than $US 200 million (£160 million) at 1990 prices – drove city planners to rethink their water supply strategy. Cost-benefit analysis suggested that a comprehensive "watershed protection program" would cost substantially less than filtration.

Putting this into operation would be a tough challenge. Top-down, punitive regulations had generally ended in failure elsewhere. Consequently, city planners opted to explore an urban-rural watershed protection partnership. This offered benefits both to city residents and rural communities in the Catskills and Delaware catchments through improved management of the water-yielding landscape. A process of dialogue, mutual understanding, and consensus-building between farmers and city representatives was consequently instigated. By the end of 1991, an urban-rural watershed protection partnership had been developed including targeted, economically efficient integrated agricultural pollution controls at individual farm business scale backed up by grants, inclusion of forestry interests, land acquisition programs, and ecologically based land management.

By January 1997, the constituent parties formalized a comprehensive Memorandum of Agreement, to which the city committed funds of approximately $US350 million (£190 million) in addition to the costs of various other initiatives in the watershed. The total cost of the watershed protection program was approximately $US1.3 billion (£700 million), a fraction of the capital and operating costs, let alone the wider environmental impacts, of traditional filtration solutions. The partnership approach, linking rural and urban stakeholders into a mutually beneficial arrangement based on the ecosystem services provided by the land, was the key to maintaining the city's pristine water quality as well as the viability of farming for the foreseeable future.

Conclusions

The Upstream Thinking and New York City case studies highlight how investment in protection of important wetland habitats and wider landscape-scale wetland functions can have significant benefits for the economics of public water supply, quantifiable in "hard" financial terms and forming the basis for PES markets. These lessons are augmented by those from the Working for Wetlands (WfW) program in South Africa, where benefits for water availability are linked with biodiversity and also social outcomes that include training and employment for disadvantaged people. In all cases, a wide range of co-benefits arise from better-functioning catchment ecosystems, augmenting both the business and ethical cases for investment in the restoration of catchment wetland functions.

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Economics of Wetland Conservation Case Study: "Systemic Solutions" for Integrated Water Management

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Mark Everard and Robert J. McInnes

Contents

Abstract

Unintended consequences arising from narrow consideration of outputs from water and environmental management technologies are of increasing concern, including, for example, climate-active gases and solid waste generated by increasing intensity of traditional wastewater treatment. Formerly overlooked environmental and financial costs, as well as social concerns along supply chains and local disruption due to vehicle movements, are also giving cause for concern. Broader consideration of inputs and outputs highlights the need for low-input solutions that optimize outcomes across multiple ecosystem services. These "systemic solutions" are defined as "*. . .low-input technologies using natural processes to optimise benefits across the spectrum of ecosystem services and their beneficiaries*."

Keywords

Ecosystem services · constructed wetlands · integrated constructed wetlands · water quality · biodiversity · landscape fit · life cycle costs · wastewater treatment

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Introduction

Unintended consequences arising from narrow consideration of outputs from water and environmental management technologies are of increasing concern, including, for example, climate-active gases and solid waste generated by increasing intensity of traditional wastewater treatment. Formerly overlooked environmental and financial costs, as well as social concerns along supply chains and local disruption due to vehicle movements, are also giving cause for concern. Broader consideration of inputs and outputs highlights the need for low-input solutions that optimize outcomes across multiple ecosystem services. Everard and McInnes (2013) explore opportunities for "systemic solutions" are defined as "*. . .low-input technologies using natural processes to optimise benefits across the spectrum of ecosystem services and their beneficiaries*."

Examples of 'Systemic Solutions'

Everard and McInnes (2013) explore progress and further opportunities to build the value of wetland ecosystem functions into water and environmental management systems by examining inputs and outputs associated with a range of ostensibly ecosystem-based techniques:

- Constructed wetlands applied as tertiary "polishing systems"
- A range of urban ecosystem-based technologies (including, for example, SuDS, green infrastructure, water-sensitive urban design, and multifunctional urban river restoration) as reviewed by Everard and Moggridge (2012)
- Washlands
- Integrated constructed wetlands (ICWs)

This analysis revealed a spectrum of "footprints," from those that differ little from conventional electromechanical engineering techniques through to techniques addressing many of the attributes of a "systemic solution." The "worst case" of constructed wetlands applied as tertiary treatment solutions, and the "best case" of integrated constructed wetlands (ICWs) are considered in the following section. This comparison illustrates the multiple benefits achievable through harnessing wetland ecosystem services as "systemic solutions," including increased societal value achieved through optimizing outcomes across a range of ecosystem services while reducing capital and maintenance inputs and associated costs.

Constructed Wetlands Applied as Tertiary Treatment Solutions

Constructed wetland systems have been deployed in wastewater treatment processes since the 1950s, initially as tertiary "polishing" systems attached to conventional engineered wastewater treatment systems (Nuttall et al. 1997; Vymazal 2011).

Tertiary treatment wetland systems make positive contributions to the narrowly framed problems they are designed to manage. However, examination by Everard and McInnes (2013) found that these tertiary treatment constructed wetlands, as they are commonly implemented today, have high inputs and limited outputs.

Barriers to achieving wider benefits and reducing inputs are as much in the planning mindset as in the potential of the techniques themselves. Frequently, these systems are implemented purely as a "bolt-on" to traditional intensive treatment techniques, with "reeds in a box" solutions (plant-filled engineering systems) lacking hydrological connection with the surrounding landscape and providing minimal additional ecological benefits (Worrall et al. 1995). This is a consequence of narrow consideration of potential outcomes, generally focused purely on wastewater treatment, rather than recognizing the wide range of potential beneficial ecosystem services that wetlands provide. Hence, the many potential co-benefits that wetland systems can provide, such as carbon sequestration, landscape aesthetics, amenity and support for biodiversity, remain largely unrecognized (Chan et al. 2006; McInnes 2011).

Inputs to constructed wetlands designed narrowly as wastewater treatment systems include infrastructure development, pumping energy, earthworks, lining media, replacement of macrophytes and planting media, with associated supply chain issues, waste disposal, and transport. This differs little, if at all, from inputs to other alternative tertiary treatment options. Outputs are equally generally narrowly conceived, relating largely to reducing selected chemical parameters such as biological oxygen demand (BOD), ammonia, and phosphorus or other pollutants.

As one example, a rectilinear, concrete-walled wetland covering 2.1 ha was constructed at London Heathrow Airport for a cost of £1.2 million (2000 prices) primarily to treat glycol-contaminated runoff (Revitt et al. 2001). This draws into focus the potential disparity between the input costs and the output benefits. There may, in this case, be scope for better uses of this investment and physical footprint to achieve a greater range of ecosystem service outcomes rather than just solving one or a few focal issues. Alternative wetland design could, for example, also provide carbon sequestration, visual and noise screening, and potential settlement of fine airborne particulates by tall vegetation, as well as wider benefits including those considered when addressing "green infrastructure" later in this paper, cumulatively achieving greater societal value.

Appropriately designed constructed wetlands could provide multiple benefits for sustainable water management (Greenway 2005). However few, if any, other ecosystem service outcomes are currently routinely considered in design or operation relative to other tertiary treatment options, beyond potential efficiency gains for energy inputs and associated carbon emissions through the use of gravity-fed systems. The biological component – reeds and associated microorganisms – is considered within a narrow engineering paradigm that overlooks their wider potential benefits. Therefore, traditional approaches to constructed wetlands are characterized by limited outputs and high inputs to maximize generally narrow design objectives.

Vymazal (2010) summarized the range of investment costs associated with various constructed treatment wetland systems designed specifically to provide wastewater treatment. Total capital investment costs were observed to vary between US\$29 per m^2 in India to US\$335 per m^2 in Belgium. Some forms of constructed wetlands, such as free water surface flow systems which usually lack the requirement for the importation of a planting medium, represented a moderate cost saving, but subsurface flow systems carried a similar capital cost to conventional water treatment technologies. However, for gravity-fed systems, which obviate the need for pumping and hence associated costs, operational costs may be a factor of two to ten times lower than for conventional electromechanical technologies (Vymazal and Kröpfelová 2008; Vymazal 2010).

Notwithstanding there being almost parity between the capital costs associated with treatment wetlands and those required for conventional wastewater treatment technologies, opportunities to optimize the benefits provided are still the exception rather than the rule. This is despite the fact that the potential for treatment wetlands to deliver multiple benefits has been recognized for almost two decades (Knight 1997).

Integrated Constructed Wetlands (ICWs)

Conversely, integrated constructed wetland (ICW) systems are designed explicitly as low-input systems optimizing ecosystem service outcomes. Everard and McInnes (2013) found that ICWs represent a high level of benefits, optimized across ecosystem service categories rather than simply maximizing one or a few focal services while overlooking potential implications for others, while minimizing inputs through the use of natural processes.

ICWs explicitly integrate three objectives: water quantity and water quality management, including flood risk management; improving site aesthetic values through appropriate "landscape-fit"; and the improvement of biodiversity. The explicit integration of these objectives seeks to achieve positive synergies that might not otherwise be delivered through more traditional land management strategies (Harrington and McInnes 2009). Thus, ICWs are inherently designed to optimize ecosystem service outcomes while reducing inputs, including both capital and maintenance activities (Everard et al. 2012). They thereby represent a practical application of the Ecosystem Approach (Harrington et al. 2011). The ICW approach has now been developed beyond Ireland and is being applied elsewhere in the world (for example, Boets et al. 2011).

"Landscape fit" includes congruence both with the natural landscape (i.e., valleys in which water would naturally flow and wetlands form) and the socioeconomic landscape (addressing needs such as intercepting runoff from farm yards, urban sewage systems, field, or forests). Thus, "landscape fit" ensures that ICWs replicate natural landscape hydrological, biogeochemical, and other processes, therefore significantly reducing dependence on anthropogenic inputs.

A comparative assessment of costs associated with an ICW constructed in 2006/2007 in County Monaghan, Ireland, to treat combined sewage from the village of Glaslough in North Monaghan was undertaken by Doody et al. (2009). Original estimates of the design capacity to treat the village of Glaslough using a traditional wastewater treatment plant equated to an estimated capital cost, including a research element to inform future designs, of €1,530,000 (2008 prices) for a plant with a 650 population equivalent (pe) capacity. Doody et al. (2009) did not assess operational expenditure associated with traditional wastewater techniques, but these are necessarily significant due to dependence on continuing inputs including energy, chemicals, waste disposal, vehicle movements, and other management operations, such that likely combined life cycle costs over a 20-year period are conservatively in the order of tens of millions of Euros. By contrast, the total capital cost of designing and constructing the Glaslough ICW (including land lease and monitoring equipment) was €770,000 including value added tax (2008 prices), with final system design capacity almost three times (1750 pe) that of the costed traditional plant. Simple comparison of capital costs with a traditional treatment facility indicates that the Glaslough ICW provides approximately three times the pe capacity at half the price. Although not assessed in detail, Doody et al. (2009) estimate that the operational and maintenance costs of the ICW system, depending as they do on less material inputs, were less than 5% of those associated with a traditional treatment plant (Doody et al. 2009). Thus, ICW life cycle costs may be one or two orders of magnitude lower compared to traditional engineered technologies.

Optimization of the ecosystem service outputs of ICWs are part of their inherent design. Each ICW is matched to local needs, but all contribute to wastewater treatment and hydrological buffering, nutrient cycling and biodiversity support, landscape aesthetics including retaining water cycling within landscapes, amenity, carbon sequestration, and a range of other services (Harrington and McInnes 2009). Some ostensibly "treatment system" ICWs on farms also provide such apparently counterintuitive benefits as provision of fisheries, support for wildlife, and improved landscape aesthetics (Everard et al. 2012) including revenue-earning uses such as recreational angling and horse riding (Harrington, pers comm.).

Thus, comparison of substantially reduced life cycle inputs with a greater array of outcomes distributed across a range of societal beneficiaries creates a strongly positive benefit-to-cost ratio compared to traditional, engineered wastewater management solutions.

Conclusions

The way that the multifunctional potential of wetland ecosystem services are considered in planning, design, and operation of water and environmental management solutions is crucial to the realization of multiple beneficial outcomes and minimization of inputs.

All techniques reviewed by Everard and McInnes (2013) were found to present opportunities for further optimization of outputs and hence for greater cumulative public value. However, ICWs are solutions designed for the express purpose of optimizing outcomes, and consequently cumulative public value, across ecosystem services rather than merely maximizing one service at net potential cost to others.

Further progress towards "systemic solutions" – "*. . .low-input technologies using natural processes to optimise benefits across the spectrum of ecosystem services and their beneficiaries*" – can contribute to sustainable development both by averting unintended negative impacts as well as optimizing benefits to all ecosystem service beneficiaries, increasing net economic value. However, this does require overcoming hurdles imposed by legacy legislation addressing issues in a fragmented way, associated "ring-fenced" budgets, and established management assumptions.

However, Everard (2011) and Everard and McInnes (2013) highlight how flexible implementation of legacy regulations, recognizing their primary purpose rather than slavish adherence to detailed subclauses, may achieve greater overall public benefit through optimization of outcomes across ecosystem services. Systemic solutions are no panacea if applied merely as "downstream" fixes, but are part of, and a means to accelerate, broader culture change towards more sustainable practice. This includes incorporating the benefits of natural wetland and other ecosystem processes more integrally into societal activities and economic systems.

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Ecosystem Credit and Payment Stacking: Overview

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Royal C. Gardner

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Abstract

Environmental credit markets have been established as a mechanism to offset wetland impacts in some jurisdictions with a well-developed regulatory program. In the USA, for example, wetland mitigation banking contemplates that a private entity may engage in a wetland restoration project, thereby producing wetland credits; these credits can then be sold to a developer, which will use them to satisfy its legal obligations to provide offsets. Similar environmental markets or offset regimes have been implemented or are in development with respect to endangered species habitat, water quality, and carbon sequestration. Accordingly, a single restoration project may have the potential to produce multiple types of environmental credits. These credits, arising from a spatially overlapping area, are often referred to as *stacked credits*. Although a properly designed credit stacking regime could induce greater investment in conservation actions, significant ecological concerns remain.

Keywords

Additionality · Mitigation Bank · Offsets · Stacked credits

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Introduction

Environmental credit markets have been established as a mechanism to offset wetland impacts in some jurisdictions with a well-developed regulatory program. In the USA, for example, wetland mitigation banking contemplates that a private entity may engage in a wetland restoration project, thereby producing wetland credits; these credits can then be sold to a developer, which will use them to satisfy its legal obligations to provide offsets (Gardner 2011). Similar environmental markets or offset regimes have been implemented or are in development with respect to endangered species habitat, water quality, and carbon sequestration (Fox et al. 2011). Accordingly, a single restoration project may have the potential to produce multiple types of environmental credits. These credits, arising from a spatially overlapping area, are often referred to as *stacked credits* (Cooley and Olander 2012) (Fig. 1).

Credit Stacking Scenarios

There is a multitude of possible credit stacking scenarios. Several are illustrated below (Fig. 2).

An important policy question is whether these stacked credits may be "unbundled" and sold in different markets to offset impacts from multiple activities (Gillenwater 2012; Gardner and Fox 2013). A properly designed credit stacking regime could induce greater investment and participation in conservation actions. However, ecological concerns are significant.

Robertson et al. (2014) note the accounting challenges associated with interrelated and integrated ecosystem functions, cautioning that credit stacking could result in environmental losses. Other concerns include the need for: clear rules regarding

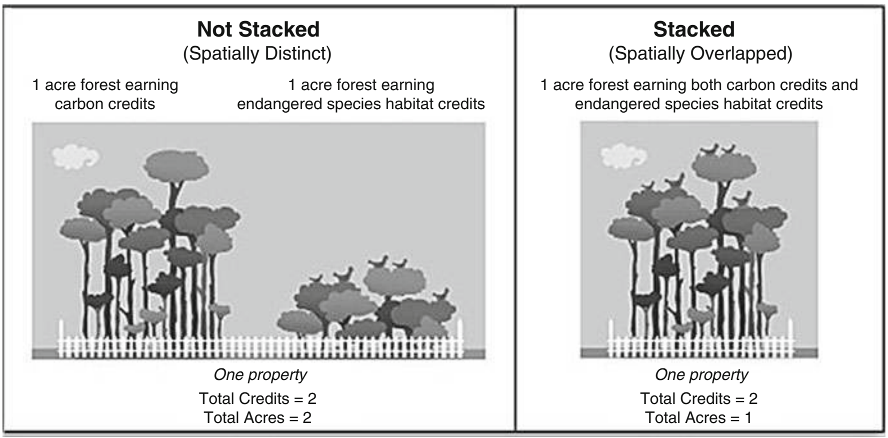

Fig. 1 Contracting spatially distinct and stacked credits

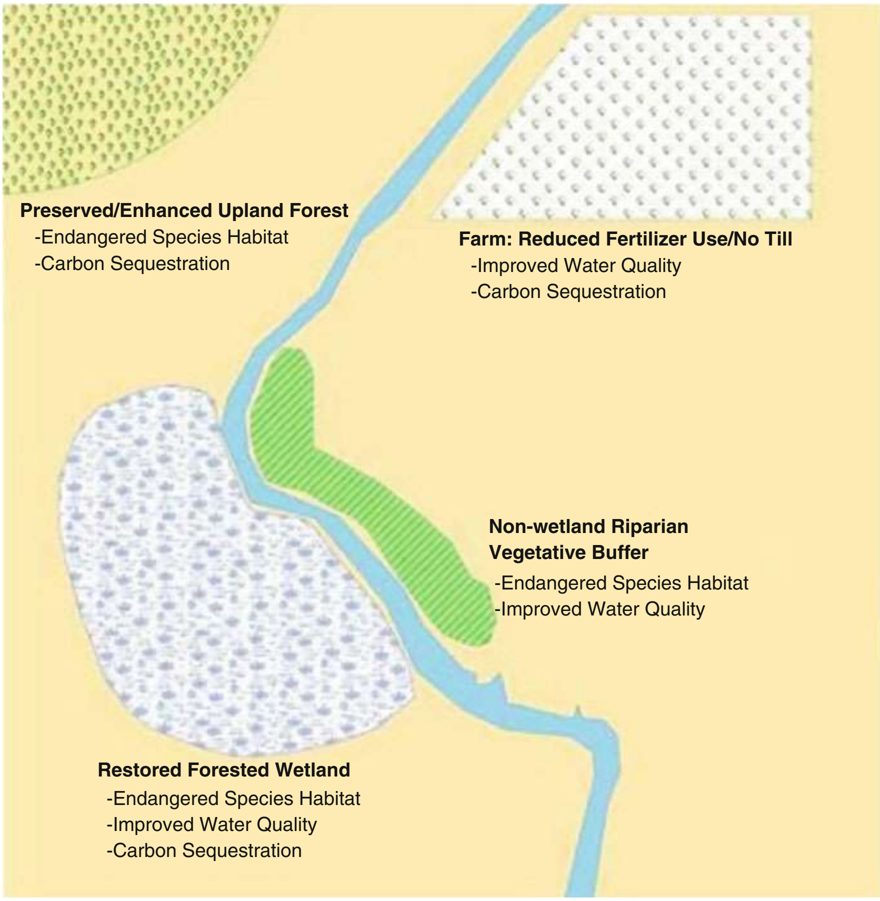

Fig. 2 Possible stacking scenarios

additionality; coordination among various agencies; regulatory capacity to verify ecological improvements, monitor the site, and take enforcement actions if necessary; and transparency (Gardner and Fox 2013). While such concerns are common in any single environmental credit market, the environmental risks are compounded when a single conservation project is being relied upon to offset multiple impacts.

Payment stacking is a related concept, which raises different concerns about undue subsidies. Payment stacking occurs when an environmental project receives "...two or more payments for the same conservation action, at least one of which was government funded" (Gardner and Fox 2013). As discussed in Gardner and Fox (2013), in the USA:

> "...a farmer might be paid by the USDA [US Department of Agriculture] through the Wetlands Reserve Program or the Conservation Reserve Program to take certain conservation actions to improve wildlife habitat. If these actions generated environmental credits that

> the farmer then sells, the farmer would be engaged in payment stacking, having received payment from both the USDA and a credit purchaser. While the farmer would benefit from the multiple revenue streams, taxpayers and environmental advocates may object since public dollars are generally intended to advance conservation, rather than offset private - sector impacts."

Currently, although USDA rules appear to permit a farmer to sell government-subsidized environmental credits, wetland regulatory agencies prohibit the use of such credits to offset wetland impacts.

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127 Corporate Wetlands Restoration Partnership: Banrock Station

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Abstract

Banrock Station is a major global wine brand that is also one of the fastest growing across major wine drinking nations. In 1994, Banrock Station purchased a 1,750 ha (4,200 acre) property 200 km north-east of Adelaide, Australia, to establish a vineyard supplying premium grapes. The property comprises river frontage, mallee woodland, wetlands, and previously cereal cropped areas. A significant part of the success of Banrock Station is built upon its close relationship with the environment, including wetland sponsorships around the world. The idea of wetland sponsorship arose from observation that the Banrock property had been intensively farmed for approximately a century with a range of negative impacts upon the fragile environment. Conservation group Wetland Care Australia had carried out some wetland restoration work to restore the Banrock Lagoon prior to the purchase by Banrock Station, this conservation work serving as a catalyst for environmental involvement by Banrock Station sponsored on the basis of donation of part of the proceeds from sale of wine. Banrock Station Wines has become an exemplar of a private sector company taking a wide range of measures to benefit wetland conservation and wise use – locally, nationally, and abroad – for a linked set of self-beneficial and publicly beneficial outcomes.

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Keywords
Banrock station · Wine corporate · Wetland care Australia · WWF-Australia · Biological control · Wetland centre

Introduction

Banrock Station is a major global wine brand that is also one of the fastest growing across major wine drinking nations. The brand started marketing in 1995 following the purchase in 1994 of a 1,750 ha (4,200 acre) property at the junction of Banrock Creek and the Murray-Darling River, near Kingston-on-Murray, about 200 km north-east of Adelaide, Australia. The purpose of this purchase was to establish a vineyard supplying premium grapes. The property consists of 12.5 km of river frontage, 600 ha of mallee woodland, 900 ha of wetlands, and 300 ha of previously cereal cropped areas for the development of premium grape varieties.

At least part of the success of Banrock Station is built upon its close relationship with the environment, including wetland sponsorships around the world. The idea of wetland sponsorship initially arose from observation that the Banrock property had been intensively farmed for approximately a century, the fragile environment suffering a range of negative impacts from prolonged farming and grazing. Prior to the purchase by Banrock Station, the conservation group Wetland Care Australia had carried out some wetland restoration work to restore the Banrock Lagoon. This early conservation work was the catalyst for environmental involvement by Banrock Station, launching the concept of establishing wetland projects outside of the property in conjunction with Landcare Australia. Environmental schemes around Australia are sponsored on the basis of donation of part of the proceeds from sale of wine.

Following this initial success, both environmentally and in terms of brand identity, the concept of supporting environmental and particularly wetland schemes using part of profits from wine sales was launched internationally. Sponsorship has benefitted more than 130 projects in 13 countries (WWF-Australia 2014) via the Banrock Station Environmental Trust. A major part of the success story relates to Banrock Station's close relationship between good earth, fine wine, associated sponsorship of environmental projects around the world, and an ethos of environmental wine marketing which is a key element in the positioning of the now world famous Banrock Station brand.

Recognition and Benefits

Banrock Station recognizes that healthy wetlands have increased the value of the property and created opportunities to generate income, also providing a natural mechanism for biological control for insects harmful to vineyards as well as being aesthetically pleasing. Banrock Station is also a demonstration of how Australian agriculture can foster a sustainable future by emphasizing a high level of simultaneous environmental, social, and economic responsibility.

The wetlands save water, with further water saving achieved through rainwater capture. In 2007, with the support from the South Australia State Government, Banrock Station was able to relocate irrigation pumps from the main lagoon to the river. This created conditions enabling Banrock Station to recreate a complete dry phase in late summer and autumn, replicating a more natural hydrological regime consisting of winter/spring floods and summer/autumn drying of short 6–8 month duration. Restoration of this unique hydrological regime has enabled the wetland system to return to healthy functioning condition, augmented by the removal of introduced European carp, and improving soil health. These actions have encouraged the return of native fish and bird species along with native flora. Reintroducing the natural dry phase also saves massive water loss through evaporation.

Banrock Station has established a Wine and Wetland Centre on site, creating opportunities for people to enjoy an outstanding wine, food, and nature experience (Banrock Station undated). The Wine and Wetland Centre overlooks the vineyards and wetlands of Banrock Station, also showcasing produce available in the Riverland including the wines. The Centre also contains four self-guided walking trails allowing people to experience the restored wetlands, with "story centers", information huts and bird-viewing hides.

To mark International World Turtle Day, Banrock Station announced a new partnership with the conservation organization WWF-Australia to help preserve Australia's iconic and World Heritage-listed site, Great Barrier Reef and its wildlife (Accolade Wines 2014). This takes the form of support, via the Banrock Station Environmental Trust, of approximately $750,000 to the *Rivers to Reef to Turtles* research initiative (WWF-Australia 2014), a four-year program seeking to identify and measure key pollutants in rivers, the Great Barrier Reef and in green turtles themselves.

Banrock Station has received a wide range of awards spanning tourism, building development and the Prime Minister's inaugural Australian Business Award for Environmental Leadership in 2000. In 2002, Banrock Station was awarded the Ramsar Wetland Conservation Award and Evian Special Prize (Ramsar 2002).

Ramsar Site

The Banrock Station wetland complex was designated as a Ramsar site in October 2002. It represents a floodplain wetland complex typical of the lower River Murray floodplain comprising areas of freshwater and secondary salinized floodplain with discrete wetland basins and channels, the largest wetland basin known as Banrock Lagoon. The wetland complex supports a high representative diversity of ecological communities and species, including over 120 species of plants and 138 species of birds of which 14 species are water bird and wetland-dependent species listed as threatened. There are over 85 species of woodland birds, 8 of which are considered rare in South Australia, as well as 7 native mammals, 14 species of reptiles, and

8 amphibians. Recreation and tourism are also key features of the site, primarily offering interpretive nature walks along designated trails. Justification of the Ramsar listing criteria includes:

- Criterion 2: The Banrock Station Wetland Complex supports two nationally listed species, the Vulnerable Regent Parrot and the Vulnerable Southern Bell Frog.
- Criterion 3: This site supports the range of biological diversity (including a large number of habitat types) found in the region within a relatively small area.
- Criterion 4: The Banrock Station Wetland Complex also provides nonbreeding habitat for 10 migratory water birds listed under international migratory bird treaties between Australia, China, Korea, and Japan (Australian Government 2002).

Banrock Station Wines have thereby become an exemplar of a private sector company taking a wide range of measures to benefit wetland conservation and wise use – locally, nationally, and abroad – for a linked set of self-beneficial and publicly beneficial outcomes.

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Financial Incentives for Wetland Protection and Restoration

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Nadia B. Ahmad

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Abstract

Wetland protection policies across the world lack specific, comprehensive national wetland laws. Relying on laws intended for other purposes, federal statutes regulating or protecting wetlands have evolved over the years. This chapter explores financial incentives for wetland protection and restoration.

Keywords

Ramsar convention · Economics · Regulation · Wetland mitigation

Introduction

Wetland protection policies across the world lack specific, comprehensive national wetland laws. Instead federal statutes regulating or otherwise protecting wetlands have evolved over the years, using laws originally intended for other purposes (Mitsch and Gosselink 1993). Generally, jurisdiction for wetland protection is spread over several agencies, and federal wetland protection is not as effective or cohesive as it could be.

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Market-oriented environmental regulation can encourage innovation for new management techniques in wetland protection and restoration (Bosselman 2009). Among the issues to consider are the size of the geographic area to be addressed, the identification of the biogeochemical processes, the translation of the policy objectives, and the choice of management technologies to achieve these objectives (Bosselman 2009). The most serious issue is whether newly formed markets can acquire the same reputation for integrity as well-established markets (Bosselman 2009).

Financial Incentives

Wetlands provide protection from flood, improve water quality through filtering, stabilize shorelines, recharge groundwater quality through filtering, and enhance biodiversity (Harness 1991; Tiner 1984). As particularly efficient converters of solar energy, biomass in a wetland environment serves as food for a wide variety of terrestrial and avian species, which makes a **wetland** the ideal environment for resident birds, providing both a year-round habitat and a critical breeding ground (McHugh 1966).

Wetlands, in their natural state, contribute a variety of environmental and socioeconomic values to society. For example, **wetlands** help maintain water quality, control erosion, discharge and recharge groundwater, and provide opportunities for the harvest of indigenous products including timber, fish, shellfish, peat, cranberries, and wild rice (Tiner 1984). **Wetlands** also provide valuable recreational opportunities, such as bird watching, canoeing, hunting, and fishing (Tiner 1984).

On account of the important functions of wetlands, state policymakers are no longer questioning whether wetlands should be protection, but instead are looking to how to protect freshwater wetlands (Harness 1991). New **wetland** protection programs are successful when they are based on comprehensive long-range planning. Another consideration is the establishment of incentives for private protection in the form of tax **incentives**, subsidies, or inducements to leave **wetlands** intact (Harness 1991).

The economic benefits of **wetland** protection manifest to society as a whole, while the costs of conservation fall on the property owner in the form of lost investment opportunities, reinforcing negative attitudes toward **wetland** conservation (Babcock 1991). The public benefits of leaving the resource in its natural state are not as easily quantified as are lost investment opportunities (Babcock 1991).

Ramsar Convention

The Convention on Wetlands of International Importance Especially as Waterfowl Habitat (the Ramsar Convention) defines wetlands as “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt including areas of marine water, the depth of which at low tide does not exceed 6 meters” (Mitsch and Gosselink 1993).

The Ramsar Convention's Small Grants Program offers financial incentives for wetland restoration to developing nations and those in economic transition. Several of the projects have focused on **restoration** activities, notably in Armenia (Lake Sevan), Ghana (mangroves and coastal **wetlands** in the Lower Volta Delta), Moldova (**wetlands** downstream of the Dniester River), and the Slovak Republic (**wetlands** adjacent to the Morava River) (Gardner 2003).

Other Funding Sources

A government may establish a trust fund that has multiple funding sources (Gardner 2003). The **Wetlands** Conservation Project, administered by the North American **Wetlands** Conservation Council, relies on appropriations, interest earned on trust fund monies, and fines and penalties collected for violations of the Migratory Bird Treaty Act (16 U.S.C. § 4406 2000). The National Coastal **Wetlands** Conservation Grant Program is funded by taxes on fishing equipment and motorboat and small engine fuels (16 U.S.C. § 3954 2000). A government may also use its bonding authority to raise money for **restoration** projects, as New York State did under its environmental bond act (Odell 2001). New York's Department of Environmental Conservation raised more than $2.5 million by selling items such as prints, posters, and stamps (Odell 2001).

Nongovernmental organizations in the United States and Canada frequently contribute funds to programs that encourage private landowners to **restore wetlands** (Gardner 2003). Under the North American **Wetlands** Conservation Act, the US federal government may provide no more than half of a project's costs (16 U.S.C. § 4407 2000). Provincial, state, and local governments, NGOs, and landowners must provide matching funds (16 U.S.C. § 4407 2000).

The private sector may also provide funding for **restoration** projects. In the United States, the Corporate **Wetlands Restoration** Partnership is a mechanism by which corporate contributions are matched 4:1 by federal and state agencies (Corporate **Wetlands Restoration** Partnership). In return for the donations, corporations are recognized as "corporate sponsors" of the **restoration** projects.

Individual landowners play a pivotal role in biodiversity preservation, open-space conservation, and wetlands management (Stern 2006). Three-quarters of all threatened or endangered species depend on private land for habitat, food, or breeding grounds (Defenders of Wildlife 1996). The majority of wetlands, which filter impurities and provide other ecosystem services, are located on private rather than publicly owned land (Morrisette 2001).

Wetland Mitigation

Wetland mitigation comes in two forms, on-site or off-site, and it is managed in three different ways: developer banks, public banks, and private banks (Whitsitt 1997). On-site mitigation forces the developer to (a) hire another who is in the business of environmental **restoration**, (b) monitor the growth and stability of the on-site **wetland** creation project, and (c) pay for added costs of the permit requirement (Sokolove and Thompson 1994).

Mitigation protects **wetlands** in three ways (Bolger 2000). First, it provides an **incentive** for landowners with **wetlands** on their property to maintain the **wetlands** (Dunec 1998). It **financially** compensates **wetland** property owners who want to develop their land, but cannot obtain a permit (Dunec 1998). **Wetland** property owners can preserve, enhance, or **restore** their **wetlands** and then sell tax credits to developers in other areas who need to purchase tax credits as part of their permit requirement (Dunec 1998). Second, it prevents takings litigation by developers (Whitsitt 1997). Mitigation allows the government to control the use of the property without voiding it of all its purpose and function, thereby infusing the **wetlands** with economic value (Whitsitt 1997). Third, mitigation banking **restores** high-grade **wetlands** that have been polluted or degraded, preserves healthy and functional **wetlands** in existence, and even enhances **wetland** areas to promote their expansion and growth (Sapp 1995).

Mitigation banking attempts to remedy problems associated with traditional, permittee-provided mitigation (Gardner 1996). The federal agencies define a mitigation bank as "**restoration**, creation, enhancement and, in exceptional circumstances, preservation of **wetlands** and/or other aquatic mitigation in advance of authorized impacts to similar resources" (Federal Guidance 1995). The key concept is the timing of the mitigation action; by definition, the **restoration** (or creation and enhancement) activities should occur before development impacts (Gardner 1996). Since the mitigation is provided in advance of impacts, there is less uncertainty about the success of the mitigation (Federal Guidance 1995). Mitigation banks are typically located on larger parcels, and "[i]t may be more advantageous for maintaining the integrity of the aquatic ecosystem to consolidate compensatory mitigation into a single large parcel" (Federal Guidance 1995). Another reason that mitigation banks may be more successful than traditional, permittee-provided mitigation is that banks "can bring together **financial** resources, planning and scientific expertise not practicable to many project-specific compensatory mitigation proposals" (Federal Guidance 1995). Furthermore, a consolidated mitigation site, such as in the case of mitigation banks, "increases the efficiency of limited agency resources in the review and compliance monitoring of mitigation projects, and thus improves the reliability of efforts to **restore**, create or enhance **wetlands** for mitigation purposes" (Federal Guidance 1995).

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Granting Exclusive Use of Wetland Area 129

Misty A. Sims

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Abstract

Views of wetlands are wastelands and sources of disease that may only be redeemed via reclamation is an anachronistic viewpoint. Today, the multiple ecological functions wetlands play are becoming increasingly widely recognized, according greater priority to their protection. This chapter addresses some of legal instruments granting exclusive use of wetland areas in the USA. Given the fragile nature of wetlands coupled with the social costs of wetland destruction, the US Congress established a permitting program in the 1977 Amendments to the 1972 Clean Water Act (CWA), specifically to preserve wetlands from unintentional or intentional destruction. In addition, "adjacent" wetlands, which may have never been inundated by the waters of the actual wetlands, are subject to federal regulation. "General permits" are issued by the US Army Corps of Engineers for activities likely to have a minimal adverse impact on wetlands, and "individual permits" are mandatory for activities likely to have significant impacts on

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wetlands such as dredge and fill activities. Unfortunately, isolated wetlands with no "significant nexus" to any navigable water body (30–60% of US wetlands) are not subject to this revised federal law, limiting their protection.

Keywords
Disease · Ecological functions · Protection · USA · Clean Water Act · CWA · Adjacent wetlands · General permits · Army Corps of Engineers · Individual permits · Dredge and fill · Isolated wetlands

Gaining Access to Wetlands

Often thought of as dismal, dank, and murky mosquito-infested pools of water, wetlands were either avoided or destroyed in the name of real estate development. However, asserting wetlands are wastelands of disease that may only be redeemed via reclamation is a viewpoint of the past. Now the important ecological functions wetlands play are becoming increasingly widely recognized and, therefore, protecting such areas is receiving greater priority.

Licensing Wetland Use in the USA

This chapter addresses some of legal instruments granting exclusive use of wetland areas in the USA. The US definition of wetlands (see 33.C.F.R. §328.3 (b)) describes them as "...areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions." Wetlands may purify air and water, serve as habitats to marine species and birds, form a natural break for a structure, or absorb rainwater to prevent a flood. Given the fragile nature of wetlands coupled with the social costs of wetland destruction, the US Congress established a permitting program in the 1977 Amendments to the 1972 Clean Water Act (CWA), 33 U.S.C.A. §1251.

Although the general purpose of the CWA is to reduce national water pollution problems, 33 U.S.C.A. §1344 specifically preserves wetlands from unintentional or intentional destruction. Congress' authority to regulate water pollution is derived from the Commerce Clause of the US Constitution in which all navigable waters may be subject to federal legislation. A wetland is deemed navigable given the CWA defines "navigable waters" broadly, and the Environmental Protection Agency (EPA) and Army Corps of Engineers defined "waters of the United States" to include wetlands. 33 C.F.R. §328.3(a)(7). In addition, "adjacent" wetlands, which may have never been inundated by the waters of the actual wetlands, are subject to federal regulation under the §404 permitting process.

Who May Grant the Power to Use a Wetland?

The CWA authorizes the Army Corps of Engineers to issue permits for applicants to discharge dredged or fill materials into navigable waters at particular locations. The Corps can issue the permit only if it finds that the dredge and fill activity will not significantly pollute the nation's waters and no practicable alternative exists that would be less degrading to the aquatic environment.

How May One Possess Exclusive Use of a Wetland?

Section 404 provides two types of permits. "General permits" are issued by the Corps for activities, which are likely to have a minimal adverse impact on wetlands. General permits apply on a national, regional, or state-wide basis for particular categories. In contrast, "individual permits" are mandatory for activities, which are likely to have significant impacts on wetlands, such as dredge and fill activities. Individual permits may only be awarded after public notice and comment. In determining whether a permit may be granted for dredge and fill activities, a balancing test is invoked by the Corps. The test requires the Corps to analyze the probable impact of the proposed activity and its intended use on the public interest. Under 33 C.F.R. §320.4, the benefits which reasonably may be expected from the activity are balanced against the foreseeable detriments. However, even if the Corps determines granting a permit is in the public interest, the EPA may still veto the decision if less detrimental alternatives to the dredge and fill activity or the permitted activity would have an unreasonable effect upon recreation areas, water supplies, fish, or wildlife.

The States' Role

US states also play a critical role in the federal wetlands permitting program as provided in §401(a) of the CWA, 33 U.S.C.A. §1341(a). In addition to providing states with veto power over §404 permits, §401(d) provides states the ability to impose "any. . .limitations, and monitoring requirements necessary to assure. . .compliance with any applicable effluent limitations or other limitations. . .of this title. . ." Such limitations become "a condition on any Federal license or permit." Attempts to minimize the states' authority under §401 have been defeated.

Prospective Challenge

Isolated wetlands with no "significant nexus" to any navigable water body are not subject to federal law §404 dredge and fill permitting under the US Supreme Court's holding in *Solid Waste Agency of Northern Cook County v. United States Army*

Corps of Engineers, et al. (2001). Unfortunately, such interpretation equates to 30–60% of the US wetlands, which are therefore unprotected by the CWA. Despite the federal government being precluded from regulating isolated wetlands, state governments have independent regulatory authority to limit dredge and fill activity within such areas.

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In-Lieu Fees in Wetlands

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Mark Everard

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Abstract

In the USA, the concept of in-lieu fees (ILFs) relates to payments for compensatory mitigation for loss, damage or disturbance of a wetland, stream or endangered species habitat through development permitted under Section 404 of the Clean Water Act or the Endangered Species Act. Through the ILF process, a permittee provides funds to an in-lieu-fee sponsor (a public agency or nonprofit organization), who pools funds from multiple permittees to provide financial resources for building and maintaining mitigation site, or sites. The in-lieu fee sponsor is responsible for the success of the mitigation.

Keywords

Compensation · Mitigation banking · Permittee-responsible mitigation · Tradable · Mitigation sequence · Avoid-mitigate-compensate · Clean Water Act

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Introduction

In the USA, the concept of in-lieu fees (ILFs) relates to payments for compensatory mitigation for loss, damage or disturbance of a wetland, stream or endangered species habitat through development permitted under Section 404 of the Clean Water Act or the Endangered Species Act.

How In-Lieu Fees Work

Through the in-lieu fee (ILF) process, a permittee provides funds to an in-lieu-fee sponsor (a public agency or nonprofit organization) (US EPA n.d.). Usually, the sponsor collects funds from multiple permittees in order to pool the financial resources necessary to build and maintain a mitigation site, or sites. The in-lieu fee sponsor is responsible for the success of the mitigation.

Like mitigation banking, in-lieu fee mitigation relates to "off-site" mitigation measures, with the intended outcome of replacing like-for-like the functions and values provided by wetland habitats that may be converted or otherwise adversely affected by proposed development activities. However, unlike mitigation banking, ILFs may be collected to perform various environmental enhancement activities throughout a watershed rather than at one particular site. Furthermore, compensatory sites are not always completely constructed prior to the environmental impacts taking place. Once enough money is received by the ILF program, the ILF program implements the project in that watershed (FHWA n.d.).

The Role of In-Lieu Fees

In-lieu fees occur within a statutory framework, stemming from permitted development under the US Clean Water Act (1972) Section 404. In-lieu fees are one of a range of three distinct mechanisms through which compensatory mitigation for unavoidable wetland impacts may be accomplished. The remaining two mechanisms are:

- Permittee-responsible mitigation, wherein the permittee undertakes restoration, establishment, enhancement, or preservation of wetlands undertaken to compensate for wetland impacts resulting from a specific project.
- Mitigation Banking, in which a wetland area that has been restored, established, enhanced, or preserved is registered as potential tradable compensation for future conversions of wetlands for development activities.

As for mitigation banking and permitee-responsible mitigation, in-lieu fees are effected as the third step of a "mitigation sequence" established to help guide mitigation decisions and determine the type and level of mitigation required under

Clean Water Act Section 404 regulations. The three steps of the mitigation sequence are:

- Step 1. Avoid: Adverse impacts to aquatic resources are to be avoided and no discharge shall be permitted if there is a practicable alternative with less adverse impact.
- Step 2. Minimize: If impacts cannot be avoided, appropriate and practicable steps to minimize adverse impacts must be taken.
- Step 3. Compensate: Appropriate and practicable compensatory mitigation is required for unavoidable adverse impacts which remain. The amount and quality of compensatory mitigation may not substitute for avoiding and minimizing impacts.

The "Avoid-mitigate-compensate" sequence is addressed separately in the Wetland Book.

Roles and Responsibilities

Federal regulations require that ILF programs be administered by government entities or nonprofit organizations. Development proponents of the ILF program pay a predetermined amount per mitigation credit to an ILF sponsor, who performs the actual mitigation. After fees are paid into the ILF programme, development proponents are relieved of any further mitigation responsibility with all risks transferred to the ILF sponsors. The ILFs sponsor then has to ensure mitigation credits will be available when needed. This lowers project costs as well as risks, at least for the development proponent.

In the USA, State Departments of Transportation are amongst the biggest users of the ILF process. Across the USA, some states have never used mitigation banks or ILF programs, while others use banks or ILF programs for the majority of their mitigation needs (FHWA n.d.). However, both ILFs and mitigation banking are proven methods for shortening project delivery time by removal of hurdles to project delivery, such as developing a mitigation plan or constructing the mitigation site.

Regulatory agencies also favour mitigation banks and ILF programs as they can often provide mechanisms to protect and restore larger blocks of habitat that provide greater ecosystem benefits than small, project-by-project mitigation.

Unsurprisingly, in-lieu fees are not universally accepted. In particular, impact fees are regarded simply as a means to collect revenue in many communities, potentially constraining and hurting the local economy as they may serve as a de facto tax which can result in slowing or terminating development in one area and instead diverting investment into other areas that do not levy impact fees (Nelson and Moody 2003). Another argument is that in-lieu fees increase the price of housing, especially new construction, where developers who pay the fees pass the cost of the fees onto the future property owners. Another concern is the negative effect that in-lieu fees may have on a local economy by curtailing job growth and reducing the amount of employment opportunities in an area.

Future Challenges

The principal challenge entailed in ILFs, as indeed other forms of compensation for wetland loss or damage, is demonstration and enforcement of equivalence. Wetlands are diverse in character, location, functions, and the range of ecosystem service benefits that they provide. Transparently auditing what is lost and what is gained is a major challenge, and particularly in an operational context in which money – in-lieu fees – is the medium by which agreements are cemented. As demonstrated “No net loss US example 2”, current mitigation wetlands generally lack the range of compensatory characteristics and services that would be required for full equivalence, perpetuating the ongoing decline of natural capital and its capacity to sustain future human wellbeing.

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Payments for Ecosystem Services

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Mark Everard

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Abstract

The basic principle of "payments for ecosystem services" (PES) is the creation of markets linking the "suppliers" of ecosystem services with their "users" (beneficiaries of ecosystem services). Some services that are currently traded, such as food, fiber, and some water-based products, have established markets, albeit that market value is generally assigned only to the target service with implications for other services largely overlooked in the production process. However, most services are entirely omitted from markets (for example, most aspects of natural flood and pest regulation, pollination, soil formation processes, etc.), though there are examples of nascent markets for some services recognized as important (such as carbon markets, biodiversity offsets, etc.) PES seeks to develop markets for services that may formerly have been omitted from economic systems.

Keywords

Externalities · Markets · Buyers · Sellers

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Introduction

The importance of linking economic systems with ecosystems began to be grasped towards the middle of the twentieth century, driven by growing awareness of largely unforeseen and certainly unintended costs manifesting as environmental "shocks" (pollution and drying out of rivers, salinization of soils, acidification of forests, depletion of fisheries, wetland loss, etc.) "Externalization" – where the action of one actor generates an unintended gain or loss to another without any payment or compensation – of the value of nature and natural processes is one of the major factors leading to their progressive degradation under economic pressures (Millennium Ecosystem Assessment 2005a), particularly through unsympathetic agricultural overexploitation of wetland resources (Millennium Ecosystem Assessment 2005b).

The emerging framework of ecosystem services has been helpful in making explicit the often formerly unappreciated but nevertheless direct links between ecosystem functioning and different dimensions of human wellbeing. Because wellbeing end-points are the basis of economics, these services are also inherently amenable to economic valuation. The ecosystem services framework also exposes the importance of bringing the multiple values of nature into economics and decision-making.

Payment for Ecosystem Services

The basic principle of "payments for ecosystem services" (PES) is the creation of markets linking the "suppliers" of ecosystem services with their "users" (beneficiaries of ecosystem services). Some services that are currently traded, such as food, fiber, and some water-based products, have established markets, albeit that market value is generally assigned only to the target service with implications for other services largely overlooked in the production process. However, most services are entirely omitted from markets (for example, most aspects of natural flood and pest regulation, pollination, soil formation processes, etc.), though there are examples of nascent markets for some services recognized as important (such as carbon markets, biodiversity offsets, etc.)

PES can create incentives for the positive management of wetlands and other ecosystems by recognizing the economic value of services provided by these habitats for people who may be either local or remote to the site under management. Examples from different spatial scales include payments for carbon sequestration in wetlands and forest (global scale); wetland and catchment management for optimal floodwater retention, erosion control, and water quality protection (catchment scale); or controlled access to wetlands for amenity, aesthetic, or educational purposes (local scale).

The term "PES" is often used loosely as an umbrella term for a variety of schemes in which the beneficiaries, or users, of ecosystem services provide payment to the stewards, or providers, of ecosystem services (Daily and Ellison 2002; Jenkins et al. 2004). However, there are some more precise definitions. Wunder (2005) defined PES as a form of market for ecosystem services in which "…a voluntary, conditional agreement between at least one "seller" and one "buyer" over a well defined environmental service – or a land use presumed to produce that service". Other important concepts associated with PES schemes include "additionality" (the PES results in benefits that would not otherwise occur and are additional to statutory obligations) and "leakage" (ensuring that protective measures in one subsidized area do not simply result in damage being transferred elsewhere) (OECD 2010).

Payments may sometimes be made directly by beneficiaries to providers, as is the case with those paying for recreational access to an owned landscape (Defra 2010). However, other potential market configurations include: (1) one-to-one as for a dam owner and the owner of its large upland catchment area; (2) one-to-many, as in the case of an intensive water-user paying (directly or indirectly) for sensitive land use by multiple farmers; (3) many-to-one, as in the case of many customers paying via charges for a water service company to subsidize sensitive, nonpolluting land use on land owned by a large owner; or (4) many-to-many as in the case of agrienvironment subsidies. Often, governments or other bodies (such as water service companies) may pay providers on behalf of multiple beneficiaries. Distinctions between schemes that fulfil all criteria defining a PES and those that are described as "PES-like" are not necessarily helpful; the essence of the PES approach is the creation of a market between buyers and sellers, often through the agency of an intermediary (such as a nongovernmental organization) and supported by a range of "knowledge-providers" (academics, economists, legal advisors, etc.).

There is a rapid proliferation of PES schemes around the world as PES is perceived as one of the most effective means to protect important functional habitat by applying the "beneficiary pays" principle to linkages between the benefits enjoyed by defined stakeholders and management of the ecosystems that produce beneficial services (Parker and Cranford 2010; TEEB 2010). Practical examples of PES can be found in all continents (▶ Chap. 132, "Payments for Ecosystem Services: Examples from Around the World") addressing water services (water supply, erosion control, fisheries and nature conservation protection, etc.), carbon sequestration (the ecosystem service of climate regulation), and protection of biodiversity and heritage landscapes. The OECD (2010) estimates that some 300 PES or PES-like schemes were operating around the world in 2010.

PES is then a still-emerging market-based approach reflecting the value of ecosystem services and bringing it into mainstream economic and decision-making frameworks.

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Payments for Ecosystem Services: Examples from Around the World

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Mark Everard

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Abstract

Payments for ecosystem services (PES) (Payments for Ecosystem Services: Definition) – the creation of markets for ecosystem services between beneficiaries and "suppliers" influencing their production – is becoming increasingly widespread around the world. Many PES schemes address aspects of the water cycle and water-mediated ecosystem services, and so have direct relevance to wetlands and wetland processes within landscapes. Global examples of PES schemes also address a diversity of other services ranging, for example, from biodiversity, fishery, and water resource protection to carbon sequestration, amenity and tourism access, landscape aesthetics, and some elements of spiritual values. This chapter provides examples of water-related PES schemes drawn from Asia, Europe, the Americas, Australasia and Africa.

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Keywords
Water resources · Biodiversity · Cultural values · Markets · Ecosystem services · Subsidies · Policy · Carbon sequestration

Introduction

Payments for ecosystem services (PES) (▶ Chap. 131, "Payments for Ecosystem Services") – the creation of markets for ecosystem services between beneficiaries and "suppliers" influencing their production – is becoming increasingly widespread around the world. Many PES schemes address aspects of the water cycle and water-mediated ecosystem services, and so have direct relevance to wetlands and wetland processes within landscapes. Global examples of PES schemes also address a diversity of other services ranging, for example, from biodiversity, fishery, and water resource protection to carbon sequestration, amenity and tourism access, landscape aesthetics, and some elements of spiritual values.

In 2010, the OECD estimated that there were already some 300 PES or "PES-like" schemes in operation around the world, and this figure is rapidly growing as PES is increasingly appreciated as a means to bring the values provided by ecosystems into decision-making.

Water-Related PES Examples from Around the World

The following subsections address selected water-related examples of operational PES schemes from Asia, Europe, the Americas, Australasia, and Africa. Though far from comprehensive, they illustrate something of the diversity and global pervasion of water-related PES schemes.

Example of Water-Related PES in Asia

China's Sloping Land Conversion Policy (SLCP) was initiated following devastating floods in the Yangtze River in 1998, signaling a significant change in the country's approach to ecosystem management. The Yangtze floods, partly attributed to deforestation in the watershed, killed 4,150 people, displaced more than 18 million people, and caused economic losses of 255 billion Yuan (about US$38 billion) (Embassy of the People's Republic of China in the United States of America (2010). Of the 34 million ha of farmland in the Yangtze and Yellow River basins, 4.25 million ha is on slopes greater than 25° where, with adequate forest cover, erosion can be reduced by 80% (Xu et al. 2006). The SLCP, also known as *Grain for Green* and set to run until 2018, aims to reduce soil erosion and flood risk and help alleviate poverty. It is the largest land retirement program in the developing world (Bennett and Xu 2005). It promotes the return of farmland on slopes over 25° to forest or grassland by providing compensation to farmers who plant trees and grass. Farmers receive annual compensation for loss of agricultural production, to increase access to health and education, and for seedlings or saplings to be planted on these slopes (Weyerhaeuser et al. 2005). The SLCP is being implemented in more than 2,000 counties across 25 provinces involving tens of millions of rural households (Bennett and Xu 2005) and, by the end of 2006, contributed to conversion of 9 million ha of cropland to forest (TEEB 2009).

Concerns raised about the SLCP include low flexibility to address local circumstances, overemphasis on tree planting, insufficient consideration of the ecological and economic functions of grasslands in semiarid areas (Forest and Grassland Taskforce of China 2003), a lack of strong sanctions for noncompliance in part resulting from conflicts between the program's dual goals of environmental amelioration and poverty reduction (Bennett and Xu 2005), and short subsidy periods which raise concerns about the permanence of ecosystem services particularly compared to time required to establish sustainable orchards or plantations of trees with medicinal value (Bennett and Xu 2005). However, SLCP is extensive and contributes to abatement of catastrophic losses of soil and peaky flows from steeply sloping catchment landscapes.

Example of Water-Related PES in Europe

In France, the Vittel PES scheme protects the valuable water source from which bottled water of the Vittel brand is obtained (Perrot-Maître 2006). Water marketed under the Vittel label is drawn from the "Grande Source" ("Great Spring") located in the town of Vittel in north-eastern France. The water has been ascribed a range of beneficial properties since Gallo-Roman times and has a value of many $US millions per annum. Contaminated water is not only disastrous to corporate reputation but also, under French law, "natural mineral water" must be bottled at source from a well-protected, specific underground supply of stable quality and with no further treatment other than elimination of traces of potentially problematic metals.

In the early 1980s, the then family owners of the Vittel brand recognized that intensification of agriculture in the Vittel catchment posed a risk to nitrate and pesticide levels in the Grande Source, as traditional hay-based cattle ranching was being replaced with intensive maize-based systems. The owners assessed five options to ensure water quality over the next 50 years (Déprés et al. 2005): (1) "do nothing" which would result in the business closing down, (2) relocate to a new catchment which would lose the valuable "Vittel" brand, (3) purchase all the land in the spring catchment which was infeasible and contrary to French law, (4) force farmers to change practices for which there was no legal requirement, or (5) provide incentives for farmers to change their practices voluntarily. The fifth option, the only viable one, led the family to instigate a negotiation process in 1988, leading to a ten-year process to transform conflict between farming and bottling interests into a successful partnership.

During this time, the Vittel brand and source was taken over by the multinational Nestlé Waters company. The dialogue generating progressively greater mutual understanding between farmers and the spring owners entailed sound scientific analysis of risks, careful targeting, trust building, understanding the diversity of farmers and their needs, the formation of a trusted broker organization, and the development of appropriate economic incentives over long timescales (18- or 30-year contracts). By 2004, landowners covering 92% of the subbasin area had reverted to extensive farming practices, securing the quality and value of the spring.

Example of Water-Related PES in the Americas

The Department of Environmental Protection of the City of New York delivers over 1.2 billion US gallons (4.5 billion liters) of water daily to 9 million people. The complex water supply infrastructure had grown piecemeal, initially from wells and smaller water bodies but, as the city grew, necessitated development of a network of aqueducts and reservoirs to bring in water from more distant sources of high natural quality. The Catskills Mountains to the north became a prime source through to 1928, with the water capture area expanding to sources in Delaware County from 1927. The Catskills–Delaware system was implemented in stages between 1937 and 1964. Today, New York City has the largest unfiltered surface water supply in the world (New York City Department for Environmental Protection undated).

However, by the 1980s, industrial-scale agriculture was replacing traditional methods, and residential and industrial development added to the threats to water resources running from the Catskills–Delaware region. At the same time, increasingly stringent public health standards were being instituted. Faced with substantial potential costs of installing filtration plant – between $US 4 and $US 6 billion capital costs in 1990 plus annual running costs of over $US 200 million – city planners began to think in broader terms about protecting the resource. Cost-benefit analysis suggested that a comprehensive "watershed protection program" would cost substantially less than filtration.

Top-down enforcement had generally ended in failure elsewhere, so the city instead sought a mutually beneficial urban-rural watershed protection partnership. Through a process of dialogue and consensus building, farmers were educated about the environmental and economic risks associated with some farming methods while, in turn, the farmers educated the city about the pressures they faced (Blaine et al. 2006). By 1991, the city and the farmers began implementing a new urban–rural watershed protection partnership based on Whole Farm Plans supported by advice from a range of statutory bodies and payments for targeted pollution control investments. The scheme has since broadened to address forestry interests, along with some targeted land acquisition.

The total cost of the watershed protection program was a fraction of the capital and operating costs, let alone the chemical inputs and climate change from traditional "hard engineering" solutions, and will maintain the city's water quality for the foreseeable future. As of 2011, 95% of farms had joined the scheme reducing agricultural pollution by 75% and stabilizing the economics of farming in the Catskills. Broadening the concept of farming from production of certain commodities (“provisioning services”) alone into a model that recognizes the value of water production, biodiversity, landscape, and cultural and other types of ecosystem services proved effective, but required social and economic innovation to make it happen.

Example of Water-Related PES in Australasia

Indigenous Maori landowners are expressing interest in markets for ecosystem services and nature conservation payments to help maintain their livelihood and culture in North Island, New Zealand (Funk 2006). A Maori conservation reserve program called Nga Whenua Rahui already provides a mechanism enabling landowners to allow land to remain in, or revert to, native bush. Development of wider markets for ecosystem services covering biodiversity protection, watershed restoration, and carbon sequestration may be essential to secure resources to support and offset the impacts of New Zealand’s rapidly urbanizing economy.

Maori land is subject to a complex system of communal ownership. However, environmental initiatives to prevent erosion, preserve water quality, restore forests, and protect biodiversity resonate intuitively with traditional Maori resource management thinking. Indeed, *kaitiakitanga*, the Maori ethic of stewardship, demands a balanced approach to safeguarding the legacy of previous generations, the needs of current generations, and opportunities for future generations: a fair articulation of the contemporary concept of “sustainability.” Ngati Porou Whanui Forests Limited (NPWFL) has been established as a tribal cooperative bringing together Maori landowners and Maori agencies to benefit from market opportunities for ecosystem services, including both government incentives for the management of erodible land and foreign investment, for example, to fund carbon sequestration services. These new markets for ecosystem services may potentially become a significant element of economic growth among Maori people.

Example of Water-Related PES in Africa

South Africa’s innovative water laws have enabled the development of advanced approaches to PES. The *Maloti Drakensberg Transfrontier Project* (Maloti Drakensberg Transfrontier Project 2007) explored hydrological and economic linkages between uplands that “produce” water and the consumption of water lower down in selected river catchments, progressing this into the design of market mechanisms for payment from consumers for the protection, restoration, and management of upper catchment areas critical for dependable run-off of clean water. Markets are founded on significant economic benefits to communities downstream in catchments, particularly heavy water users such as forestry, intensive agriculture, mining, and industries such as paper mills, for which the marginal value of enhanced or protected flows becomes particularly significant in dry periods.

The Maloti-Drakensberg market model is finding favor with the South African government as a replicable market mechanism, for example, as a remedy to current unsustainable land-use practices on extensively farmed land in the Baviaanskloof and Tsitsikamma watersheds (Mander et al. 2010). These poor practices are problematic for the sustainable supply of important ecosystem services including good veld condition, carbon sequestration, and overall water security for downstream water users including farmers and the water-stressed urban areas of Port Elizabeth, Jeffreys Bay, and Cape St Francis. PES offers a win-win opportunity by realigning the incentives available to farmers to deliver better for the needs of wider society.

Future Challenges

There are many more examples of PES around the world. The challenges are many including, for example, ensuring that benefits are genuinely additional to regulatory requirements and that there is some permanence to changed resource stewardship. Also that the distribution of benefits and costs is equitable and that market transaction costs are kept low. There remains always the potential for strong vested interests to capture these processes, so strong governance is required.

A persistent criticism is that valuation of ecosystem services puts a "price on nature" that may be traded off against economic growth; this needs to be countered with awareness that ecosystem resources underpin the economy and its continued growth, while also ensuring that some newly traded services are not maximized at net cost to a linked set of both traded and non-traded services.

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Cost-Sharing and Direct Payments for Wetland Protection

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Anastasia Telesetsky

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Abstract

Given the significance of wetlands for habitat, carbon sequestration, water quality, and groundwater recharge, government agencies frequently sponsor programs that permit either cost-sharing or direct payments for wetland protection as well as wetland restoration. Many cost-sharing or direct payment programs for wetland protection are designed to either restore degraded wetlands or to pay for nondevelopment of a critical wetland area. The terms of these programs will differ depending on the government entity that coordinates the program. In some regions, such as the Philippines and Thailand, intergovernmental organizations or national governments may make payments to wetland communities to participate in projects such as mangrove reforestation as part of a larger project to enhance coastal human security. These programs have secondary benefits of protecting a wide range of additional wetland functions. For those parts of the world where

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there are inadequate nonmonetary incentives to protect wetland systems, designing programs that provide protection through "payments for ecosystem services" may prove effective in ensuring long-term protection for wetlands. By introducing a monetary incentive for long-term wetland protection of "natural capital," communities may be willing to forego short-term benefits associated with immediate agricultural or commercial development. In order for these programs to be successful over the long term, there may need to be legal interventions beyond simply introducing a cost-sharing or direct payment program. For communities that continue to have insecure land tenure, people may be unwilling to forego short-term benefits of using land in exchange for long-term payments for restoration projects.

Keywords
Carbon sequestration · Water quality · Cost-sharing · Protection · Restoration · Payments for ecosystem services · Natural capital · Incentives · Tenure

Introduction

Given the significance of wetlands for habitat, carbon sequestration, water quality, and groundwater recharge, government agencies frequently sponsor programs that permit either cost sharing or direct payments for wetland protection as well as wetland restoration.

Key Principles

Many cost sharing or direct payment programs for wetland protection are designed to either restore degraded wetlands or to pay for nondevelopment of a critical wetland area. The terms of these programs will differ depending on the government entity that coordinates the program. In some regions, such as the Philippines and Thailand, intergovernmental organizations or national governments may make payments to wetland communities to participate in projects such as mangrove reforestation as part of a larger project to enhance coastal human security. These programs have secondary benefits of protecting a wide range of additional wetland functions.

For those parts of the world where there are inadequate nonmonetary incentives to protect wetland systems, designing programs that provide protection through "payments for ecosystem services" may prove effective in ensuring long-term protection for wetlands. By introducing a monetary incentive for long-term wetland protection of "natural capital," communities may be willing to forego short-term benefits associated with immediate agricultural or commercial development. In order for these programs to be successful over the long term, there may need to be legal interventions beyond simply introducing a cost-sharing or direct payment program.

For communities that continue to have insecure land tenure, people may be unwilling to forego short-term benefits of using land in exchange for long-term payments for restoration projects.

Examples of Cost-Sharing and Direct Payment Programs

Wetland Reserve Program

In the United States, the voluntary Wetland Reserve Program managed by the United States Department of Agriculture (USDA) as part of the Farm Bill provided for a cost-sharing arrangement for individual landowners including corporations and Native American tribes who committed to removing cropland from production in order to restore degraded wetlands that are suitable for wildlife habitat. The statutory reserve program permitted eligible landowners to opt for either an easement or a more permanent agreement.

Individual landowners including corporations could agree to have the government place either a permanent easement or a 30-year easement on their land in exchange for payments. In the case of a permanent easement, the government through the USDA provided an easement payment once the easement has been properly filed. The easement was specific to the protection of the wetland values and the owner still maintained (1) control of access, (2) title and right to convey title, (3) quiet enjoyment, (4) undeveloped recreational uses, (5) subsurface resources, and (6) water rights. Additionally, the government provided for a cost-sharing agreement between the owner and the Natural Resource and Conservation Service for restoration and maintenance of the wetland resource. For a 30-year permanent easement, the government provided an easement payment of 75% of the value of a permanent easement once the easement has been properly filed. The owner of the 30-year easement was also eligible for reimbursal of up to 75% of their eligible restoration costs. Where owners were unwilling to provide an easement over their land, the owner could still be eligible to enter into a restoration cost-sharing agreement as long as the owner continued to maintain the wetlands for at least 10–15 years. Native American Tribes were also permitted to negotiate agreements.

Since its initiation, the program has been relatively successful in terms of recruiting 2.3 million acres of volunteer easements and protecting important migratory bird corridors such as the Rainwater Basin in Nebraska. Yet, this has fallen far short of the potential number of wetland acres that could be held in conservation reserves. From 2010 to 2013, the statute authorized a cap of 32 million acres at any one time in conservation reserves. Even though there is capacity for additional acreage, the Wetland Reserve Program, as of September 2013, was no longer enrolling members until either the current legislation is extended or a new Farm Bill is negotiated. While either of these may happen in due course, the temporary lapse of the program reflects one weakness of cost-sharing programs that depend on continued long-term government funding for their success.

Additional US federal programs that provide for cost-sharing for wetland protection include the US Fish and Wildlife's Partners for Fish and Wildlife scheme, providing financial support for wetland restoration activities and the US Wildlife Habitat Incentive Program providing 5–10 year contracts on developing wetland habitat particularly for threatened and endangered species. A number of individual states in the United States also have landowner incentive programs, for example Delaware which provides a rental payment for loss of agricultural income of farmed wetlands plus a flat rate of $2,220/acre for a shallow water wetland greater than three-fourths of an acre or $1,665/acre for a smaller wetland.

South Africa's Working for Wetlands (Program adopted in response to South African National Water Act (Act No.36 of 1998))

Wetlands play a significant role in provisioning services for individual communities within South Africa, in addition to a diversity of regulatory services of importance to the local environment and livelihoods. The loss of these wetlands has implications for poverty and development. In response to the degradation of national wetlands, the Government of South Africa created a pro-poor development program focused on restoring damaged wetlands by providing direct payments to landowners and community members to restore wetlands. The program is coordinated by the South African Department of Environmental Affairs, Department of Water Affairs, and the Department of Agriculture, Forestry and Fisheries and is housed within the South African National Biodiversity Institute.

There are two forms of payments for ecosystem services associated with the program. For landowners, the government supports financially the rehabilitation of degraded wetlands based on a contractual agreement signed with the landowners to allow contractual work on their land, to avoid activities that would jeopardize the success of the rehabilitation effort, and to ensure that other wetlands on their properties are not degraded.

The more innovative component of the program is providing direct employment programs to community members who may be long-term unemployed and may not have land tenure. For these individuals, direct payments from the government in the form of salaries are provided for work done to restore and rehabilitate wetlands as part of the Expanded Public Works Program. They also receive on-the-job training which may increase their future employability.

Over the course of 12 years, restoration work under the program has improved the condition of 70,000 ha of wetland area through the provision of 2.2 million person days of work and 168,400 days of training in both vocational and life skills. This model of providing direct payment to employ temporary workers for wetland restoration holds great promise for communities where there may be existing obstacles for achieving land tenure yet degraded ecosystems need attention. While it may not be tenable to continue these employment programs in the long term, it does provide a stopgap measure to halt further degradation by initiating much needed restoration efforts that also address a lack of employment opportunities and employment skills. Ultimately, land tenure concerns will need to be addressed to ensure that there is no reversion back to a pattern of degradation.

Future Challenges

The greatest challenge with any cost-sharing program or direct payment program is setting the appropriate payment levels that will incentivize participation in the program while also ensuring the maximum number of participants in the program. One of the primary challenges with "payment for ecosystem" programs has been ensuring adequate side payments that are both fair and feasible over the long-term. Maintaining funding is a chronic challenge across both the global North and South. Without strong political support, programs such as the United States Wetland Reserve Program can no longer continue to increase protected acreage. This is problematic since restoration and conservation efforts by private owners are vital for supporting government efforts to enhance ecological connectivity for wetlands. Subnational groups such as states like California and Delaware who are independently pursuing wetland restoration efforts may alleviate some of this potential funding shortages by the federal government.

While one should be optimistic about the support that the voluntary cost-sharing programs have been receiving, there remains inherent future challenge for programs built on the models of the US Wetland Reserve Program or the Working for Wetlands program. Both of these programs have the potential to create social expectations that ecosystem health will always depend on the availability of market incentives. When there are no government-funded market incentives available to do restoration work, it remains to be seen what level of restoration and protection efforts individual parties will undertake independent of any government reimbursal.

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Ontario Wetland Habitat Fund

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Mark Everard

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Abstract
Wetlands are recognized as one of the Canadian province of Ontario's most diverse and productive habitats, upon which a wide diversity of wildlife and human benefits depend. The Wetland Habitat Fund was created to support and encourage land owners who conserve and enhance wetland habitat for wildlife. The Fund provides private land owners with financial assistance for projects that improve the ecological integrity of wetland habitats.

Keywords
Ontario · Canada · Wildlife Habitat Canada · Hunters · Hunting · Permit · Habitat conservation · Duck stamp

Introduction

Wetlands are recognized as one of the Canadian province of Ontario's most diverse and productive habitats, upon which a wide diversity of wildlife and human benefits depend. The Wetland Habitat Fund was created to support and encourage land

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owners who conserve and enhance wetland habitat for wildlife. The Fund provides private land owners with financial assistance for projects that improve the ecological integrity of wetland habitats.

The Wetland Habitat Fund

Wildlife Habitat Canada (http://whc.org/) is a national, not-for-profit organization based in Ottawa. Wildlife Habitat Canada generates funds – the Wetland Habitat Fund – for conservation projects through the Wildlife Habitat Conservation Stamp Program. Hunters of migratory waterfowl are required to purchase these conservation stamps when obtaining a Migratory Game Bird Hunting Permit (Eastern Ontario AgriNews 2000).

The Wetland Habitat Fund then represents a financial resource for which wildlife habitat enhancement proponents can bid. Fund administrators favor project submissions with conservation plans that:

- Contribute to the improvement or restoration of local wildlife habitat
- Address local wetland wildlife habitat issues
- Target specific wetland habitat issues but reflect broader landscape ecology
- Lead to benefits that can be enjoyed either directly or indirectly by the general public
- Encourage partnerships and foster cooperation among landowners, interest groups, and conservation agencies
- Provide evidence of permission from the landowners for any work proposed on private lands

Habitat projects that conform to the criteria may be eligible for funds to a maximum of 50% of the project cost or $5000, whichever is less. Projects of an exceptional nature may be funded with different ceilings.

Preference is given to projects that use practical, cost-effective techniques to address habitat issues identified in a wetland conservation plan. Examples of techniques endorsed include:

- Establishing, protecting, or enhancing vegetative buffers around wetlands
- Building small water retention (control) structures
- Planting vegetation for wildlife
- Planting dense nesting cover plots
- Adding in wetland nesting sites
- Improving grassed waterways to control soil erosion into wetlands
- Restricting livestock access and alternative watering systems

Applicants complete a wetland conservation plan and an application form for funding, submitting this funding proposal to Wildlife Habitat Canada.

The basic principle operates rather like a tax on beneficiaries (hunters) of wetland services (migratory game bird hunting) that is hypothecated back into wetland protection or enhancement. As such, it has some of the attributes of a ► Chap. 131, "Payments for Ecosystem Services" (PES) scheme, in which multiple "piggy-backed" additional services (flood control, water quality protection, fish recruitment, etc.) are delivered on the back of the paid "anchor service" (habitat for ducks and other wildlife).

About Wildlife Habitat Canada

Wildlife Habitat Canada (n.d. -a) works to conserve, restore, and enhance wildlife habitat in Canada. It achieves this by funding habitat conservation projects, promoting conservation action, and fostering coordination among conservation groups.

One of the principal mechanisms by which Wildlife Habitat Canada raises funds is through the sale of Canadian Wildlife Habitat Conservation Stamps, upon which is printed (each year since 1985) a painting by a Canadian wildlife artist. The vast majority of stamps are purchased by hunters to validate their Migratory Game Bird Hunting Permit. The majority of proceeds from these sales fund Wildlife Habitat Canada's grant program. Stamps and also prints of the selected paintings are also purchased by individuals and collectors to raise funds for wildlife habitat conservation.

In addition to the revenue from the Habitat Conservation Stamps and prints of the artwork, Wildlife Habitat Canada raises additional funds from a variety of governmental and industry partners to undertake specific activities and projects.

The contribution from the Canadian hunting community is particularly significant in terms of funds available for wildlife habitat conservation (Wildlife Habitat Canada n.d. -b). In addition to purchasing Wildlife Habitat Conservation Stamps, the wider hunting community raises additional funds through conservation fundraising dinners and activities as well as volunteering on committees and projects benefiting wildlife.

Wildlife Habitat Canada also builds relationships with nonhunters, who support and contribute to many habitat conservation projects and help to bring the community of interests in wildlife habitat conservation together. By creating a funding base – so-called duck stamp dollars – Wildlife Habitat Canada aims to overcome the common complaint that finances thwart practical, beneficial habitat conservation projects (Bailey 2010).

Challenges Remaining

The Wetland Habitat Fund is essentially voluntary. Embedding hypothecation of revenues from wetland uses into the mainstream of economic thinking remains the overriding priority.

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Property Rights

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Mark Everard and Norman A. Dupont

Contents

Abstract

The term legal "property" encompasses ". . .anything that is owned by a person or entity". Property is divided into the two types: "real property" comprising interests in land, real estate, growing plants or the improvements on it; and "personal property" comprising everything else such as "intellectual property". There are other subsidiary definitions of property, including "common property" (ownership by more than one person) and "public property" (ownership by a governmental body such as the federal, state, county, or city governments or their agencies). However, the principal distinction drawn in this article is between property that is owned by private persons or corporate entities, and that which is a "common" to humanity.

Keywords

Legislation · Policy · Subsidy · Societal levers · Property · Rights · Obligations · Public benefits · Private benefits

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Introduction

The term legal "property" encompasses "*. . .anything that is owned by a person or entity,*" divided into the two types of "real property" (interests in land, real estate, growing plants or the improvements on it) and "personal property" (a term that covers everything else such as "intellectual property") (Hill and Hill 2005). There are other subsidiary definitions of property, including "common property" (ownership by more than one person) and "public property" (ownership by a governmental body such as the federal, state, county, or city governments or their agencies).

The principal distinction drawn in this article is between property that is owned by private persons or corporate entities, and that which is a "common" to humanity the water cycle itself is not owned, but is a natural "common property" essential to life on this planet and to the wellbeing of all of humanity. The water cycle connects, at the same time deeply affecting and deeply influenced by, water, land, and air media as a contiguous whole. Although aspects of the oceans and air are claimed as property, the bulk of these media are fluid, transboundary, massive, and effectively ownerless, though both are significantly affected by their uses by humanity including as a common into which wastes are emitted. However, land and freshwater bodies are appropriated as property widely across the world, be that by states, individuals and communities, or by corporations.

In the USA, private property can clearly include "wetlands" within the physically owned real property. That is, one can own a home (or prospective home) that includes, as part of the "parcel," a wetland. But, the traditional rights of a property owner (exclusive possession and use) may be limited when it comes to use of the wetlands area of the property. The conflict over the extent of the right of a private property owner to "use" his property by filling in part of a wetland area was the subject of a US Supreme Court case, *Rapanos v. U.S., 547 U.S. 715*(2006) ("*Rapanos*").

Rapanos and the Debate Over Property Rights as to Wetlands

The *Rapanos* case (which included a companion consolidated case involving the Cababell property owners) provides a textbook illustration of the current legal battle between "property rights" and use of wetlands. The Rapanos family and their affiliated businesses owned property in the Midland, Michigan area. Although being advised that the wetland areas at three of their sites were deemed to be within the scope of the federal Clean Water Act (CWA) because they ultimately drained into various creeks and rivers that constituted "waters of the United States" regulated under the CWA, the Rapanos proceeded to deposit fill material into three wetland areas without a permit. The federal government sued the Rapanos, claiming that they had violated the CWA, which requires a permit (issued by the Army Corps of Engineers) before filling in a wetland.

The Carabells sought a permit to deposit fill material in a wetland located about one mile from a major lake, Lake St. Clair. They wanted to construct condominiums on part of their land and proposed to do so by filling in 12 acres of existing wetlands.

The wetland area had a drainage ditch on one side that largely (although not completely) inhibited the flow of water from the wetland into a ditch that ultimately connected to Lake St. Clair (via another ditch or drain flowing into a creek). The Army Corps found that the wetlands in question were subject to federal control under the CWA. The Carabells sued, alleging that their wetlands were not regulated within the jurisdictional scope of the CWA, but rather were isolated and did not require a permit.

Two separate sets of lower courts affirmed rulings that both sets of wetlands were regulated under the CWA. The US Supreme Court granted petitions for review and consolidated both cases for argument and decision in 2005.

In 2006, the Supreme Court issued a divided and divisive ruling. A four-member plurality of the Court held that the wetlands owned by the Rapanos and Carabell interests were not within the scope of regulation under the CWA because they were not "waters of the United States," a term left undefined in the statute. The plurality expressly rejected the argument that narrowing the definition of "waters of the United States" will hamper federal efforts to preserve the Nation's wetlands, essentially concluding that the wisdom of such efforts "is beyond us" (547 US at 745-46).

Justice Kennedy concurred in the judgment of the four-member plurality, but wrote a separate concurring opinion that took a much different view of the CWA. Justice Kennedy argued that the state's ability to regulate privately owned wetlands turned on whether there was a "significant nexus" between the wetlands and more traditional "navigable waters." Justice Kennedy rebuked the plurality opinion as "unduly dismissive" of the important interests serve by protection of wetlands. He cited several amicus briefs and other documents for the proposition that: ". . .wetlands play a critical role in controlling and filtering runoff" (547 US at 777-78). For Justice Kennedy, "wetlands" could be regulated (even if they were privately owned) if ". . .the wetlands, either alone or in combination with similarly situated lands in the region, significantly alter the chemical, physical, and biological integrity of other covered waters more readily understood as 'navigable'" (547 US at 780).

Many commentators have noted that the Supreme Court's decision in *Rapanos* was open to multiple interpretations and left many property owners with no clear map of when their wetland areas were subject to federal regulation.

In an effort to clarify when a property owner might reasonably anticipate that his or her wetlands property might be subject to federal regulation, the US EPA and Army Corps of Engineers issued a new draft regulation intended to clarify the scope of what constitutes "waters of the United States." The draft regulation, together with a lengthy preamble discussing it, consumes 88 pages of fine print in the *Federal Register* (http://www2.epa.gov/sites/production/files/2014-04/documents/fr-2014-07142.pdf). The new draft rule is based upon EPA's and the Army Corps' conclusion that they should adopt Justice Kennedy's "significant nexus" standard to determine whether the wetlands are within the scope of federal regulation. The draft rule was published in April 2014, and a comment period on the proposed rule extends to 14th November 2014. More than 5,000 comments have been received at the time of writing (October 2014), and it is likely that a number of additional comments will be submitted prior the deadline.

We examined the USA as one major country that has encountered much litigation and regulatory efforts to address which types of wetlands are of sufficient national concern that they can be regulated by federal agencies. The answer is: "Maybe." If the wetlands at a particular point can be shown to have a "significant nexus" with other waterways that are clearly navigable, the federal government can regulate them under the terms of the CWA. If, however, no such "significant nexus" can be demonstrated, the private property owner has full control over the wetlands and can use them (or fill them in) as he or she deems appropriate.

As noted below, however, other nations have significantly different approaches to wetlands and limitations upon ownership rights.

Rights and Responsibilities

Wetlands are multibenefit systems, providing a range of ecosystem services of benefit to humanity. Consequently, use and management of wetlands for private uses – such as wetland conversion for agriculture, drainage for development, or alternatively institution of sustainable fishery practices – can have profound influence on the balance of benefits and risks flowing to connected communities. Various countries have recognized the interconnection of wetlands systems with preservation of habitats necessary for endangered species. For example, the EU Council Directive 92/43/EEC of 21stMay 1992 (the EU Habitats Directive) recognizes the need to conserve various sites, including wetlands, as necessary to ensure biodiversity in the EU community.

More recently, lawsuits in the USA have been initiated over the destruction of wetlands, arguing that such destruction leads to increased possibility of uncontrolled flooding (Rich 2014).

Notwithstanding the multiple benefits provided by wetlands, and increasing recognition of the important role they play in species preservation, biodiversity, flood control, and other uses, the appropriation of property rights continues to impact wetlands and their contribution to a range of beneficiaries. Property rights take many forms. For example, many tribal and community ownership models effectively treat landscapes and their wetlands as commons, traditional practices evolved often over longer time scales erecting a range of formal and informal governance arrangements, protocols, taboos, and sanctions that constitute effective and sustainable stewardship (Ostrom 2000). However, other private ownership models have effectively conferred largely untrammelled rights on owners, manifesting for example in the widespread loss or degradation of wetlands for agriculture or urban development regardless of the implications of lost services for societal wellbeing and future risk (Millennium Ecosystem Assessment 2005). As an extreme example of problematic, locally interest-led exploitation of water, wetlands, and broader landscapes, regardless of ramifications for the users of other water-mediated benefits, the "riparian principle" enshrined within much South African legislation of the apartheid era included, for example, the Irrigation Act of 1912 which states that "He can do whatsoever he pleases with it and neither the owners of lower-lying land nor even the public can claim to be entitled to make any use at all of that water." Control of land, but above all of water and the services it provides across broader landscapes, is the medium for either enforcement of asymmetrical power relationships across society or a powerful means for their dissolution.

However, at least in much of the developed world, private property rights have been in transition over the past century, with awareness of the wider ramifications for society progressively manifesting as a set of responsibilities, constraints, and inducements influencing resource use. This transition in the UK and much of the developed world takes as its starting point the beginning of the twentieth century when, as the common saying put it, "An Englishman's home is his castle" reflecting that property rights implied largely unconstrained rights to use that land as the (generally male) owner desired. Yet, by the close of the twentieth century, the freedom of action of landowners had been substantially constrained by a corpus of environmental, industrial, planning, and other legislation, a growing body of common case law relating to the impacts of resource use on the rights of other people, incentives to manage the land in certain culturally-preferred ways, taxes to dissuade undesirable activities, novel markets for sustainably sourced goods as well as biofuels and feedstock crop production partly displacing dependence on fossil resources, catchment management strategies favouring water-sensitive land uses, measures to secure public access, and a range of other changes (Everard and Appleby 2009; Everard 2011). Progress over the century may have occurred beyond the span of an average human life, and so have been less obvious to those living through it. However, the telescope of history reveals a broad and profound revolution which, in historical terms, has been very rapid indeed. This transformational journey is defined by recognition that land and other environmental assets are valuable not merely as private property but because they produce a diversity of publicly beneficial ecosystem services, regardless of their ownership status. Beneficial services such as open spaces and fresh air, buffering of storm energy and floodwater, cleansing of air and protection of water resources, soil, and biodiversity have become progressively recognized and then subsequently valued and institutionalized, be that in statute or common law, shifting social norms, market differentiation and reform, or any of a range of other "societal levers" (Everard et al. 2014).

This example of progressive awareness, not merely of the potential for exploitation for private benefit of wetlands and other land and water resources but of the implications of their stewardship for a wider set of ecosystem services, some of them manifesting remotely in time, space, and affected communities (including future generations), highlights sequential progress towards what we now frame as an Ecosystem Approach. Consequently, the constantly evolving toolkit of "societal levers" addressed above serves to influence – through a set of obligations, liabilities, incentives, and protocols – private wetland use and management decisions, informed incrementally by evolving awareness of their implications and implicit responsibilities.

The inherent tension between private profit and other motives and the wider societal implications of the use and management of wetlands and other land and water resources constitutes a major factor shaping the mosaic nature of landscapes, where mixed ownership occurs throughout much of world. The mosaic of wetlands and wider landscape features resulting from the resolution of private choices with wider formal and informal policy environments affecting features of the natural environment do not automatically result in the production of a spectrum of ecosystem service outcomes underpinning all of the diverse needs of society, including future wellbeing and security. However, they do at least assure a degree of heterogeneity and resilience. The same cannot automatically be claimed for landscapes subject to more authoritarian and centralized control and uniform management, such as in soviet-era Russia.

Challenges

There remains a pressing need to achieve greater coherence between the choices made by private wetland and other land and water resource owners, with higher-level aspirations and commitments towards achieving sustainability and implementation in full of the ecosystem approach. Moreover, long-held legal theories of private property rights continue to trump some state and national efforts to preserve and protect wetlands.

At present, the diverse toolkit of "societal levers" lacks this coherence and is in need of systemic review. This is not to enforce standard practices but to achieve a better optimization of socially beneficial outcomes across landscapes rather than merely the localized maximization of private profit or other owner-driven, potentially narrowly framed outcomes, with all of their associated externalities (Everard et al. 2014).

The values of wetlands need to be more strongly promoted and represented across the societal policy toolkit, more effectively to influence the choices of private owners. For wetlands and other resources in public ownership, this transition to greater coherence between policy-setting and land use decision-making scales is, at least in theory, more straightforward, though in practice it may be complicated by tenancy rights. Nonetheless, achievement of the wise use of wetlands depends on more concerted progress addressing coherence amongst the broader range of regulatory, incentive, and other tools influencing the decisions of private and public owners and managers of wetlands.

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Safe Harbor Agreements

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Marcela Bonells

Contents

Abstract

This article provides an overview of Safe Harbor Agreements (SHAs) as voluntary mechanisms for the conservation of wetlands in private lands, along with their advantages and disadvantages, and examples of similar (albeit non-identical) agreements around the world. SHAs are voluntary agreements between private landowners and US conservation government agencies in which landowners commit to undertake measures to restore, maintain, preserve, or enhance habitat, including wetlands, for species protected (or listed) under the federal Endangered Species Act (ESA). In exchange, they are provided regulatory relief from land use restrictions. While SHAs take place in the context of a statutory

The views expressed in this article are those of the author and do not represent the views of the Ramsar Secretariat.

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framework, they are non-traditional tools for the conservation of protected species, in that the incentive they provide is regulatory relief.

Keywords
Safe Harbor Agreements · Wetlands conservation tools · Endangered Species Act · Voluntary conservation assurances · Stewardship agreements · U.S. Fish and Wildlife Service · Biodiversity conservation tools · Habitat conservation · Regulatory relief

Introduction

Safe Harbor Agreements (SHAs) are voluntary agreements between private landowners and US conservation government agencies, whereby landowners consent to undertake measures to restore, maintain, preserve, or enhance habitat, including wetlands, for species protected (or listed) under the federal Endangered Species Act (ESA) (National Wildlife Federation 1997; Environmental Defense 1999). In exchange, landowners are relieved from land use restrictions and additional management measures, which could otherwise result if listed species were attracted to the improved habitat, or if population numbers increase for a species as a result of landowners' actions (USFWS 2011). At the conclusion of the agreement, landowners can return the property to the conditions that existed prior to entering into the agreement (baseline conditions) (idem).While SHAs take place in the context of a statutory framework, they are nontraditional tools for the conservation of protected species, in that the incentive they provide is regulatory relief (Defenders of Wildlife 2006; Gardner 2003).

Even though they are primarily used for species conservation, SHAs can be, and have been, used for the conservation of wetlands as habitat for protected species (Gardner 2003). Given that most habitats for listed species under ESA are found in private property, SHAs are a valuable tool to incentivize private conservation of habitat for listed species, including wetlands (USFWS 2011). While the concept of SHA, as discussed in this entry, is a United States tool, there are similar (albeit not identical) tools around the world that can be used for similar wetland conservation purposes.

Part I of this entry provides a brief overview of the regulatory framework for SHAs, its scope, application, and operation. Part II outlines the benefits of SHAs for listed species' habitat, including wetlands, and their application as wetland conservation tools. Part III provides some examples of agreements or tools around the world. Part IV discusses future challenges for SHAs as tools for wetland conservation.

Regulatory Background

SHA agreements have their legal bases under section 10 (a) (1) (A) of the ESA, which is the primary federal statute for the recovery of endangered and threatened species, as well as their habitat, in the United States (Bean 2010). Section 9 of the

ESA prohibits the taking of species listed under the Act (Gardner 2003). The term "take" has been broadly defined to include significant habitat degradation or modification, resulting in actual harm or killing of a protected species (Quarles and Lundquist 2010; Gardner 2003). This can include, for instance, the degradation or destruction of a wetland upon which a species depends for foraging (Gardner 2003; Bonells 2012). This prohibition imposes significant land use restrictions on landowners and discourages voluntary habitat conservation measures for fear that such measures could result in land use restrictions if species were attracted to the enhanced habitat (ED 1999).

As a response, the US Fish and Wildlife Service (USFWS), the National Oceanic and Atmospheric Administration (NOAA), and the National Marine Fisheries Service adopted the Safe Harbor Policy to encourage activities beneficial to a protected species and its habitat, for example, the restoration, creation, or maintenance of a wetland (Bean 2010). Because SHAs do not affect existing restrictions in the property, resulting from endangered species already present in the property, or "baseline conditions," the government issues an enhancement of survival permit to the landowner allowing the future take of a species, as long as baseline conditions are not violated (Sullins 2001; Bean 2010; ED 1999).

There are two types of SHAs: individual and umbrella SHA. Individual SHAs are between individual landowners and the government (USFWS 2005a). Umbrella or programmatic SHAs involve a region or state and are administered by a government or nongovernment entity(idem). This entry focuses on individual SHAs.

How Do Safe Harbor Agreements Work?

Any nonfederal landowner, the FWS, or state agency can initiate a SHA. SHA may involve a single landowner (individual SHAs) or multiple landowners (umbrella SHAs) (USFWS 2011). Landowners, with the assistance of the government, will establish a SHA and apply for an enhancement survival permit, after having gathered all relevant information regarding the property, establishing baseline conditions, and proposing management actions (USFWS 2011). The government will describe baseline conditions, which can encompass population distribution and estimates, or habitat characteristics sustaining use by the species (idem). While the benefit to a species may vary depending on the nature, duration, and extent of activities undertaken, the SHA must provide a "net conservation benefit" to the species included in the agreement (idem).

Developing a SHA can take between 6 to 9 months or longer depending on the circumstances (USFWS 2011). Importantly, SHAs do not have to provide permanent conservation to a species, and their duration will vary (idem). SHAs may last as long as needed to accomplish the intended benefits (Kishida 2001). For example, they can last one season to restore a particular type of wetland or several years for the prescribed burning of a habitat (ED 1999). Additionally, SHAs can be renewed upon expiration for as long as the landowner and the government agree to do so. However, if the agreement is not renewed, the relief from land use restrictions or assurances included in the agreement will no longer apply, and the land owner will not be exempt from the "take" prohibitions allowed under the permit (idem).

SHAs can run with the land, which means that they can be transferred if the property is sold or given away, as long as the new land owner agrees to be bound by the SHA (USFWS 2011). Similarly, the rights and obligations under a SHA can be transferred from generation to generation if a land owner dies (ED 1999). SHAs can also be modified to account for situations beyond a landowner's control, such as a prolonged drought eliminating a population of rare butterflies inhabiting a meadow (ED 1999). In such cases, the baseline conditions can be adjusted (idem). Baseline conditions can also be adjusted to include species not previously covered in the agreement, but that may have arrived in the property as a result of the improved habitat (Kishida 2001). Land owners may also voluntarily increase their baseline, which can serve to mitigate activities conducted by that landowner or by others (idem). SHAs will not apply to species not covered by the agreement or if excluded at the request of the landowner (idem). In such cases, a separate agreement will be needed (idem).

SHAs will terminate upon the expiration date stipulated in the agreement (ED 1999). However, landowners may be able to terminate them early due to unforeseen circumstances, such as disease or economic hardship (USFWS 1999). Permits implementing SHAs may be revoked, as a last resort, when activities would jeopardize a species covered by the agreement (USFWS 1999).

Benefits and Drawbacks of Safe Harbor Agreements

There are many ecological benefits associated with SHAs, including, for example, maintenance, enhancement, creation, or restoration of a wetland upon which a protected species depends. Other ecological benefits include increased habitat connectivity and reduced habitat fragmentation; creation of buffers around protected areas; and capacity to test novel management techniques and to address an increase in or stabilization of populations of protected species (USFWS 2011). SHAs can also be helpful mechanisms to protect wetland ecosystems that may be threatened by development or may be underrepresented in a country's system of publicly owned lands (Bonells 2012). Benefits to landowners include the ability to carry out activities for the benefit of protected species and their habitat without incurring additional land use restrictions or giving up typical land uses (USFWS 2011). Additionally, SHAs encourage voluntary stewardship actions that would not otherwise occur under traditional regulatory command and control tools.

While SHAs are primarily used for the recovery of protected species, they have been used to promote wetland restoration. For example, in 2001 the US government and a private landowner entered into a SHA for the restoration of wetland (palustrine emergent marshes) and upland habitats for two listed species of waterfowl, the koloa or Hawaiian duck and the nene or Hawaiian goose (USFWS 2005b; Gardner 2003). Another example includes a SHA to relocate Oregon chub to a landowners' perennial, spring-fed pond. As part of the SHA, the landowner created a buffer surrounding the pond and refrained from conducting activities adjacent to the site that would be harmful to the chub's habitat, such as pesticide use (USFWS 2005b).

Despite the multiple benefits that SHAs can offer, there are drawbacks. Criticisms associated with SHAs include their lack of permanence, since landowners are not obliged to undertake perpetual or permanent conservation measures (Bean 2002). Another drawback of SHAs is that they do not promote proactive conservation of valuable species or habitats, since it applies only to species listed under the ESA (Bean 2002; Bonells 2012). Finally, SHAs do not offer financial incentives. Thus, the relief from landuse restrictions within a SHA may not be strong enough to persuade landowners not to put their land to more profitable uses.

Similar Agreements Around the World

Although this entry discusses SHAs as tools to promote the recovery of protected species and habitat under US law, there are similar agreements in other countries such as Australia, Austria, Canada, and Colombia, among others. It should be noted that this entry does not intend to provide an exhaustive list of such agreements or programs, but merely to provide examples.

One such example is in Australia, where conservation agreements have been put in place between the national government and individuals for biodiversity conservation activities inland or sea (Australian Government n.d.). These agreements are legally binding on all parties (idem). Like SHAs, conservation agreements may require management activities, impose land use restrictions, and/or provide technical incentives (idem). Interestingly, such agreements may also provide for the conservation of the ecological character of Ramsar Sites (idem). However, unlike SHAs, they may provide financial incentives.

Similarly, in Canada, there are voluntary stewardship agreements at the provincial level for the conservation of species at risk, under the federal habitat stewardship program for species at risk. For example, in Saskatchewan, landowners with qualified projects will receive funding and enter into a verbal agreement to "maintain and protect" "native prairie or/and riparian areas" (South Saskatchewan River Watershed Stewards Inc. n.d.). These agreements are transferable, and landowners efforts are recognized with a sign and certificate of appreciation signed by the Minister of Environment. However, unlike SHAs, eligible landowners can receive financial assistance.

In Austria, landowners can enter into voluntary agreements with the government for biodiversity conservation under the Forest Reserves Programme. To qualify, landowners must set aside a specific natural forest reserve within their property. These agreements are governed by private law for a period of 20 years (Frank and Muller 2003). Landowners can opt out of the agreement under specific circumstances, and they will receive financial compensation (idem).

In Colombia, landowners can voluntarily designate their land as a private nature reserve or nature reserve of civil society for biodiversity conservation under the National Protected Areas System (Parques Naturales Nacionales de Colombia n.d.).

Landowners are required to create conservation areas in their property, but are able to conduct sustainable, productive activities in their property (idem). They are eligible for technical and financial assistance and are able to benefit from the goods and services of the ecosystems they protect (idem). These reserves can be used to protect rapidly disappearing ecosystems, including wetlands (Bonells 2012).

Future Challenges

SHAs can therefore comprise useful mechanisms for wetland conservation. However, they lack permanence, and their lack of financial incentives may not persuade landowners to forgo more profitable uses of their land in order to undertake habitat conservation measures.

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Section X

Management of Provisioning Services

Mark Everard

Management of Provisioning Services: Overview

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Mark Everard

Contents

Abstract

Provisioning services comprise extractable material and energy provided by ecosystems. Many of the services that are most directly exploited by people are provisioning services, including for example food, fiber, water, and medicinal products. Dependable flows of food, water, and other key provisioning services are an essential basis for human wellbeing.

However, overexploitation of many provisioning services such as intensive food or timber production can overlook the importance of other services and the integrity of productive ecosystems themselves. Agriculture in particular is a major contributor to the degradation of wetland quantity, quality, and diversity globally, as well as at national scale.

The challenge of management of wetlands for provisioning services is two-fold. Firstly, sustainable management is required to ensure reliable flows of provisioning services. Secondly, there is a need for systemic management of ecosystems such that provisioning services are not extracted at rates and by means that erode the supply of other ecosystem services and the integrity of productive habitats.

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Keywords
Natural capital · Resources · Systemic management · Food · Timber · Water resources

Introduction

Provisioning services (▶ Chap. 138, “Provisioning Services: The Basics”) comprise extractable material and energy provided by ecosystems. Many of the services that are most directly exploited by people are provisioning services, including for example food, fiber, water, and medicinal products.

Dependable flows of food, water, and other key provisioning services are an essential basis for human wellbeing. However, as observed by the Millennium Ecosystem Assessment (2005a) general synthesis study as well as the report directly addressing Wetlands and Water (Millennium Ecosystem Assessment 2005b), the overexploitation of many provisioning services, overlooking the importance of other services and the integrity of productive ecosystems themselves, is a major contributor to the degradation of wetland quantity, quality, and diversity globally. This trend is mirrored nationally in the UK, as exposed by the UK National Ecosystem Assessment (2011). This appears also to be the case in other countries, contributing to a significant overall erosion of the value that ecosystems provide to society (TEEB 2009).

The challenge of management of wetlands for provisioning services is two-fold. Firstly, sustainable management is required to ensure reliable flows of provisioning services. Secondly, there is a need for systemic management of ecosystems such that provisioning services are not extracted at rates and by means that erode the supply of other ecosystem services and the integrity of productive habitats.

Management for Provisioning Services

Provisioning services underpin many dimensions of human wellbeing, including such fundamental resources as fresh water and food. Wetlands also provide a range of other provisioning services essential for a good quality of life, including building products (thatch, aggregates, timber) as well as ornamental, medicinal, biochemical, genetic, and other materials.

The UN Sustainable Development Goals (SDGs) (United Nations, 2015) to which signatory nations committed in September 2015 comprise 17 goals to improve the world. SDG6 explicitly addressed clean water and sanitation, SDG14 is concerned with life below water and SDG15 relates to life on land. However, most SDGs in one way or another are supported by the ecosystem services provided by wetlands including for example SDG2 concerned with zero hunger (wetland food production is significant globally), SDG3 addressing goods health and well-being and SDG13 concerned with climate action (wetlands are major carbon sinks). Programmes to secure fresh water and a linked set of wetland ecosystem services, including governance to ensure gender equality (SDG5) and reduced inequalities (SDG10), are fundamental to all dimensions of human wellbeing across the world.

Wetlands clearly play a fundamental role in securing water resources and maintaining services essential to human progress, including for example their role in sanitation and assimilation of other wastes, and hence are a priority for sustainable management. This is one of the drivers prompting the Ramsar Convention on Wetlands to issue guidance on the inclusion of wetlands into integrated water resource management (Ramsar Convention Secretariat 2010). However, provisioning services are just one of the suite of services provided by wetlands and by wetland processes operating across landscapes, and it is therefore essential that all of these services and their productive ecosystems are managed on a coherent and integrated basis.

Sustainable Harvesting of Provisioning Services

Water and food sufficiency are central to the development agenda, both in the developing world as well as globally in the face of spiralling population growth with its escalating demands for food, water, and other key resources. Sustainable management of land and water resources to safeguard their potential to provide these services is then essential if hardships and conflict are to be avoided. Water has often been described as likely to be the basis of future conflicts (The Economist 2011; UNESCO undated), though in reality it has been since at least the 1968 Six Day War in the Middle East and competition for water resources has been a driver of conflicts throughout human history. So water security, and the effective comanagement of aquatic ecosystems, is a vital element of transnational, regional, and global security including raising people out of poverty. Management of wetlands for the reliable supply and equitable sharing of provisioning services is then a priority globally, including explicitly within the MDGs. Many of the essential provisioning services provided by wetlands have been the subject of attempts at sustainable management.

A variety of tools have been developed to encourage the sustainable management of fresh water, including for example Integrated Water Resource Management (IWRM), which integrates a range of sciences including the economic, social, ecological, and hydrological. The purpose of IWRM is to treat catchments, perceived as logical management units, in an integrated way to seek consensus on balancing economic and subsistence activities with the “carrying capacity” of water fluxes throughout water catchments.

Fish harvested from wetlands as diverse as lakes, rivers, and coastal seas comprise a sizeable proportion of the protein consumed by people worldwide. Some of these fisheries also support trades as diverse as the ornamental fish market and fish-derived products such as is in glass and caviar. Sustainable harvesting of fisheries is therefore a priority if ecosystems are not to become degraded along with their potential to support people into the future. Many nations have imposed fishery limits related in some way to equitable sharing of the maximum sustainable yield of these fisheries, including for example the EU-wide Common Fisheries Policy (CFP) and other international agreements such as under the United Nations Convention on the Law of the Sea (UNCLOS: www.un.org/Depts/los/). However, crude quantitative limits are insufficient to safeguard the integrity and functioning of ecosystems, with destructive methods such as scallop dredging causing substantial damage to reefs, other sea bed systems, and a range of wider ecosystem services (Everard and Appleby 2008). Further progress towards sustainable fisheries is taking place in supply chains including, for example, the Marine Stewardship Council (MSC) certification scheme established in partnership between major food manufacturers (particularly Unilever) with an interest in long-term resource security, as well as NGOs and local producers. MSC acts as an independently certified "chain of custody" from fishery through manufacturer to distributor, retailer, and onwards to plate. Since some 50% of global food fish is now produced from aquaculture, an industry with some significant environmental issues including adverse implications for wetlands, a nascent parallel initiative, the Aquaculture Stewardship Council (ASC), is also being developed.

Security of other locally harvested food sources as diverse as lotus roots, water chestnuts, and wildfowl requires stewardship of resources to ensure that productive wetland systems do not become degraded by over-exploitation.

Systemic Management of Wetlands

A wider problem arises from the widespread global conversion of wetlands for agricultural production. The evolution of current economic forces mean that most incentives relate to the production of a range of provisioning services, particularly food and fiber, production of which therefore tends to be maximized at the expense of other services and the integrity of ecosystems. Warnings of the consequences of this overconsumption of the natural world go back many years to times of substantially lower global population and demands, including for examples Aldo Leopold's (1949) warnings of a world irrevocably diminished by human appetite.

Maintenance of the productive capacity of wetlands is important for a wide variety of subsistence and trading purposes. However, important though they may be, wetlands are essential not merely for their provisioning services but for a range of regulatory, cultural, and supporting services. Systemic management of wetlands protective of this full suite of services but also the integrity and resilience of the wetland resource itself is essential to secure ongoing human wellbeing.

Systemic management is implicitly recognized by the Ramsar Convention's "wise use" concept (▶ Chap. 55, "Wise Use Concept of the Ramsar Convention"), describing the needs to balance the provisioning benefits provided by wetland systems with the production of all other services cumulatively contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation. Attempts to operationlize this broader conception of the benefits and vulnerabilities of wetlands include, for example, SWAMP (Sustainable Wetland Areas Management Programme) developing in the early

1990s to support the aspirations of the Ugandan Wetland Programme to achieve the "wise use" of its valuable wetland resource (Everard et al. 1995).

However, outside of a very few integrated wetland programmes worldwide, wetlands and other habitat types are in general threatened by overexploitation, or conversion for production, of a narrow subset of provisioning services. Owing to their general omission from markets, management of wetlands and other habitats has tended to overlook the value of most ecosystem services notwithstanding their substantial value to society. The Millennium Ecosystem Assessment Wetlands and Water synthesis (2005b) summarized many of the ways in which different types of wetlands produce a different balance of valuable services, emphasizing the importance both of the diversity and location of wetland types across landscapes. Nevertheless, there has been a substantial erosion of the wetland resource in many parts of both the developed and developing world.

It is essential that wetland integrity and the security of the full suite of ecosystem services that they produce are adequately reflected in valuation and decision-making. Growing examples of catchment management schemes incentivizing farmers to safeguard water supplies, such as the New York City water supply and the UK's Upstream Thinking programs demonstrate the public value of appropriately rewarding the farming of land simultaneously for food, water, and a range of other benefits. There are many other initiatives around the world acknowledging the benefits of sustainable management of wetlands for water resource purposes, including for example under the 2008 FAO report *Scoping agriculture–wetland interactions*, which recognizes that, "*Agriculture-wetland interactions (AWIs) are becoming more important as rising demand for food production exacerbates pressures on wetlands*" (Wood and van Halsema 2008). The FAO report was the first output of GAWI, the Ramsar Convention's *Guidelines on Agriculture, Wetlands and Water Resource Interactions Project* undertaking a range of case studies seeking to understand and devise more sustainable strategies for productive land uses that stay within the "*. . .ecological and resilience boundary*" of the wetland systems in which they operate.

Much remains to be done to change the current paradigm of farming, which tends to liquidate most ecosystem services to maximize the yield of just a few such as production of food and fibre with some diversification into chemical feedstock and biofuel. Redressing established practices and assumptions is essential if we wish to protect wetland processes and thereby to restore the climate, air, and water regulating services as well as cultural and supporting services necessary to secure our collective future, as well as to retain the viability of both wetlands and farming.

Integrated Planning for Sustaining Provisioning Services

Inclusion of wetlands and their ecosystem services into policy and practice necessarily involves recognition that a secure supply of provisioning services is essential to support human wellbeing, as indeed are sustained flows of regulatory, cultural, and supporting services. This requires concerted management that recognizes the importance of these services across government policy areas, and that includes valuation of all services to ensure that short-term exploitation does not override long-term sustainability and security. Concerns about resource economics and equity need to be integrated centrally into land use planning, requiring a "sea change" in policy direction, economic valuation, and public perception of the long-term value of wetlands.

Future Challenges

Numerous challenges are entailed in achieving sustainable management of wetland provisioning services. These include long-term stewardship of the ecosystem resources providing these services, including valuation of all provisioning services beyond those with immediate market returns. Provisioning services have also to be managed in balance with the breadth of regulatory, cultural, and supporting ecosystem services for which wetlands are also important.

Beyond incentives – regulatory, market, supply chain expectation, and other – there is also a need to embed the long-term value of wetland provisioning services across all policy areas. Wetland conservation can no longer safely be assumed to be a matter of traditional nature conservation activities, but must be addressed as a pressing issue of resource security supporting trade, industry, and export markets, as central to food security and the international development agenda, relevant to water resource and flood management, and impinging centrally on all policy areas.

The challenge of the sustainable management of wetland provisioning services is then in reality one of the systemic management of all fields of human interest respecting wetlands and other habitats as the core natural capital supporting long-term wellbeing.

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Provisioning Services: The Basics

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Mark Everard

Contents

Abstract

Ecosystem services describe the diverse benefits that the natural world provides to people. The UN's Millennium Ecosystem Assessment program harmonization a range of pre-existing ecosystem service classification schemes into a consistent system suitable for comparison of major habitat types on a global basis. The primary division within the MA classification scheme was the grouping of ecosystem services into four major categories: provisioning services, regulatory services, cultural services, and supporting services.

Provisioning services comprise extractible products from ecosystems such as food, water, timber, and fiber. The whole of the human economy is driven by the goods and services provided by ecosystems and natural resources including minerals, fossil fuels derived from ancient vegetative matter, water, timber and food, as well as medicinal and biochemical products produced. These are all provisioning services, and wetland systems may be significant producers of them. Intensive extraction and production of these provisioning services has a profound impact on the environment, particularly as driven by ever more efficient technologies focused narrowly on the maximization of single services to the exclusion of most others. Management of ecosystems for provisioning services production need to be cognizant of wider implicaitons for toher services.

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Keywords
Food · Timber · Aggregates · Fossil fuel · Extractible materials · Goods

Definition

The concept of ecosystem services, describing the diverse benefits that the natural world provides to people, has been emerging as a pedagogic and management tool since the late 1980s. Since that time, disparate classification schemes have been developed often addressing specific habitat types (wetlands, coral reefs, rangelands, croplands, forests, etc.) and/or bioregions.

One of the many contributions of the UN's Millennium Ecosystem Assessment program (Millennium Ecosystem Assessment 2005a) was the harmonization of these prior schemes into a consistent classification system suitable for comparison of major habitat types on a global basis. The primary division within the MA classification scheme was the grouping of ecosystem services into four major categories: provisioning services, regulatory services, cultural services, and supporting services.

Provisioning services are defined by the Millennium Ecosystem Assessment as extractible products from ecosystems including, for example, "*. . .food, water, timber, and fiber*" (Millennium Ecosystem Assessment 2005a). The whole of the human economy is driven by the goods and services provided by ecosystems and natural resources including, for example, minerals derived from geological deposits; gas, coal, and oil from ancient deposits of vegetative matter; water for human consumption, industry, and irrigation extracted from rivers, lakes, and groundwater; timber extracted from forests; food and fiber derived from managed agricultural ecosystems and harvesting from more natural ecosystems; and medicinal and biochemical products produced by wetland biota. Intensive extraction and production of these provisioning services has a profound impact on the environment, particularly as driven by ever more efficient technologies focused narrowly on the maximization of single services to the exclusion of most others (UK National Ecosystem Assessment 2011).

A Millennium Ecosystem Assessment synthesis specifically considering global wetlands and water (Millennium Ecosystem Assessment 2005b) (including lakes, rivers, marshes, and coastal regions to a depth of 6 m at low tide but acknowledging that many wetland types were underrepresented) found that "*. . .more than 50% of specific types of wetlands in parts of North America, Europe, Australia and New Zealand were destroyed during the twentieth century, and many others in many parts of the world degraded.*" Nevertheless, wetlands produce a diversity of all categories of ecosystem services, including significant provisioning services such as fish, fiber, and water supply, comprising a wide range of both marketed and nonmarketed economic benefits. The total economic value of unconverted wetlands was considered often to be greater than that of converted wetlands, yet many of these values were ill-considered in contemporary decision-making. Consequently, "*. . .the* *degradation and loss of wetlands is more rapid than that of other ecosystems,*" a trend thought likely to be exacerbated by climate change. Land conversion, significantly for agricultural purposes (i.e., conversion to maximize a favored subset of artificially-grown provisioning services) was seen to be a major contributor to the global loss of wetland quantity and quality, reducing their capacity to contribute to human wellbeing.

The Ramsar Convention's "wise use" concept (www.ramsar.org/handbooks4/) recognizes the needs to balance the use of wetlands for the production of provisioning services and all other regulatory, cultural, and supporting services contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation.

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Agricultural Management and Wetlands: An Overview

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Mark Everard and Adrian Wood

Contents

Abstract

Agriculture is the major driver of wetland loss or degradation globally. However, wetland agriculture is also a major contribution to human well-being across the world and a critical contributor to livelihoods, poverty reduction, and climate change adaptation, especially in developing countries. Through wetland agriculture, hundreds of millions of people interact with wetlands. In addition to their partial use or wholesale conversion as agricultural systems, wetlands also play a diversity of additional roles in agricultural management. Effective management of wetlands for sustainable agricultural use is a global priority, both due to the impact of wetland management on the water cycle and value of wetland productivity for human security and development. A key consideration for their sustainable management is that all services of wetlands are considered, as all are pertinent to overall ecosystem characteristics and integrity, the balance of benefits that they confer on diverse beneficiaries,

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and hence the net benefit that they provide to society now and into the future. The role of wetlands in agricultural productivity are diverse, including both direct uses and the indirect supporting, regulatory, and cultural services they provide, adding to the resilience and functioning of wider landscapes. Progressive recognition and internalization of these wider benefits into the policy environment is essential if sustainable agriculture and wetland use are to be achieved for the net security of humanity.

Keywords
Optimisation · Nonprovisioning services · Systemic solutions · Land retirement · Conservation reserve Programme

Introduction

Agriculture is the major driver of wetland loss or degradation globally (Millennium Ecosystem Assessment 2005a). However, wetland agriculture is also a major contribution to human well-being across the world and a critical contributor to livelihoods, poverty reduction, and climate change adaptation, especially in developing countries (Wood et al. 2013; Howard et al. 2009; Kumar et al. 2011). Indeed, it is through wetland agriculture that hundreds of millions of people interact with wetlands. In addition to their partial use or wholesale conversion as agricultural systems, wetlands also play a diversity of additional roles in agricultural management.

Such is the priority of effective management of wetlands for sustainable agricultural use that Ramsar COP9 (The 9th Meeting of the Conference of the Contracting Parties, Kampala, Ramsar Convention 2005) agreed to the production of "GAWI – Guidelines on Agriculture and Wetland Interactions" (Wood and van Halsema 2008). The GAWI guidelines, summarized in this "Overview" chapter, built upon the findings of a prior international study, the Comprehensive Assessment of Water Management in Agriculture (2007), addressing the wider ramifications and trends of agricultural activities for the water cycle. Both the GAWI and Comprehensive Assessment studies were informed by the Millennium Ecosystem Assessment and a wider range of ecosystem service studies. Wetlands and agriculture were also the theme for World Wetlands Day in 2014, the focus of which was on wetlands and agriculture as partners for growth.

Wetlands also play a wider set of roles in agricultural management, reviewed below. A key consideration for their sustainable management is that all services of wetlands are considered, as all are pertinent to overall ecosystem characteristics and integrity, the balance of benefits that they confer on diverse beneficiaries, and hence the net benefit that they provide to society now and into the future.

Wetlands and Agricultural Management

The significance of agriculture to the water cycle is indicated by the global scale of water withdrawals to support it. Production of food and other agricultural products takes 70% of the freshwater withdrawals from rivers and groundwater, whereas industry takes 20% and municipalities only 10%. So the role of agriculture in all aspects of the water cycle is significant and these pressures are set to grow with increasing economic and demographic growth, which have been the major driving forces in the predominantly human-induced transformation of wetlands (Comprehensive Assessment of Water Management in Agriculture 2007).

The Millennium Ecosystem Assessment

The Millennium Ecosystem Assessment (MA) (2005a) study *Ecosystems & Human Well-being: Synthesis* is important from a wetlands and agriculture perspective for the way in which it recognizes the contribution of provisioning services to livelihoods and development. In particular, it identifies how the maintenance of these provisioning services is tightly interlinked with the other three categories of ecosystem services (regulatory, cultural, and supporting) and how, for instance, undermining supporting and regulating services can reduce the ability of the wetlands to provide provisioning services. Further, the MA argues that the overdevelopment or overuse of some ecosystem services, notably agricultural provisioning services, will undermine other ecosystem services (e.g., regulating) and so will make problematic the achievement of some related development goals (e.g., safe drinking water and sanitation). As a result, the MA suggests that balancing ecosystem services outcomes from wetland use and management is essential for their continuing integrity and for the achievement of development benefits.

The MA's approach is important because it takes forward the debate on wetland management and agriculture in several ways. Firstly, it emphasizes the links between wetlands and human well-being. Secondly, it raises the important issue of trade-offs among ecosystem services. Thirdly, it suggests that consideration of trade-offs should be central to the design of wetland uses in order to support maintenance of wetland ecological character and to achieve development goals. Finally, with respect to agriculture in wetlands, the MA states that:

> *In regions where agricultural expansion continues to be a large threat to wetlands, the development, assessment and diffusion of technologies that could increase the production of food per unit area sustainably, without harmful trade-offs related to excessive consumption of water or use of nutrients or pesticides, would significantly lessen pressure on wetlands.* (Millennium Ecosystem Assessment 2005b, p. 66)

Overall, the MA stresses that a rebalancing of ecosystem service outcomes is needed in order to sustain the productivity of these areas, with use of provisioning services needing to become more ecologically sensitive paying attention to agroecological opportunities, multiple cropping systems, and achieving diversity within agricultural landscapes.

The Comprehensive Assessment of Water Management in Agriculture

The Comprehensive Assessment of Water Management in Agriculture (2007) comprised a critical evaluation of the benefits, costs, and impacts of the past 50 years of water development. It identified the water management challenges communities face today and the solutions people have developed so far around the world. It addressed the overarching question: "*. . .how can water management in agriculture be developed and managed to help end poverty and hunger, ensure environmentally sustainable practices, and find the right balance between food and environmental security?*"

The Comprehensive Assessment was a multi-institute process, organized through the CGIAR's Systemwide Initiative on Water Management (SWIM), convened by the International Water Management Institute (www.iwmi.cgiar.org/assessment). Overall, it aimed at assessing the current state of knowledge and stimulating ideas on how to manage water resources to meet the growing needs for agricultural products, to help reduce poverty and food insecurity, and to contribute to environmental sustainability.

The scope of the Comprehensive Assessment spanned water management in agriculture, including fisheries and livestock, and the full spectrum of crop production from soil tillage through supplemental irrigation and water harvesting to full irrigation in a sustainable environment context. It considered both the importance of wetlands for agriculture and the adverse impact of agriculture on wetlands. The Comprehensive Assessment highlighted the fact that the expansion and intensification of agriculture has had many benefits for society but has also had adverse impacts on ecosystems globally. It highlighted the fact that agricultural systems depend fundamentally on ecological processes so, if ecosystems become degraded, not only are many direct ecosystem services lost but agricultural productivity itself may be undermined (Comprehensive Assessment of Water Management in Agriculture 2007). Involving the food and environment communities together has been an important step in finding sustainable agricultural solutions.

The primary focus of the Comprehensive Assessment with respect to wetlands was sustainable optimization of the livelihood benefits from these ecosystems. Hence, a key feature in the Comprehensive Assessment is recognition that managing ecosystems for livelihoods is going to have impacts on ecosystem services and these are not necessarily congruent with managing them for biodiversity goals. The Comprehensive Assessment further developed the idea of conflicts of interest in wetland use and the need for trade-offs between livelihood requirements and conservation that need skillful and innovative forms of management to balance outcomes. The objectives of addressing these trade-offs should not be to maximize values for conservation and poverty reduction simultaneously, but rather to produce the greatest overall net benefits for people while at the same time avoiding fundamental ecological threats and ensuring long-term sustainability of all ecosystem services (Sellamuttu et al. 2008). Thus, a pluralistic approach to wetlands and agriculture is proposed that provides opportunities to increase the overall productivity of agricultural systems while ensuring that all uses of ecosystem services are enhanced rather than harmed by agricultural development (Nguyen-Khoa et al. 2008).

The Comprehensive Assessment emphasized that drivers of wetland conversion for agriculture will intensify over the next three decades as populations rise and the demand for increased economic output and food production rises steeply. It also noted that these drivers of change will be most likely exacerbated by climate change and will be most severe in developing countries. In recognition of this, the Comprehensive Assessment stressed the need to identify:

- How the ecosystem services that contribute to agriculture can be enhanced
- How agricultural activities can be designed to contribute to ecosystem functioning (Comprehensive Assessment of Water Management in Agriculture 2007)

The Comprehensive Assessment found that, while it is possible to produce the food needed, it is probable that today's food production and environmental trends, if continued, will lead to crises in many parts of the world. Only if improvements are made in water use in agriculture, we will meet the acute freshwater challenges facing humankind over the coming 50 years. This should involve placing much greater emphasis on managing agricultural systems (whether in wetlands or not) as an integrated part of the landscape for multiple rather than single services (i.e., for the full suite of ecosystem services).

An important finding (under Policy Action 3 of the Comprehensive Assessment relating to the management of agriculture to enhance ecosystem services) was that good agricultural practice can enhance other ecosystem services in agroecosystems, rivers, and wetlands.

The GAWI Guidelines

The GAWI guidelines (Wood and van Halsema 2008) were developed by a group comprising the Ramsar Secretariat and a range of other intergovernmental bodies and NGOs with the aim, "*To promote synergies between agriculture, wetlands and water resources management, through the development and implementation of guidance on the joint management of agricultural and wetland systems for food production, poverty reduction, livelihoods support and environmental sustainability*". This built on a request from Ramsar COP8 to develop a framework for the dissemination of good agriculture-related practice, site-specific, and crop-specific information and policies that demonstrate sustainable use of wetlands for agriculture.

The GAWI framework document:

- Analyzed cases of agriculture-wetland interactions using the DPSIR (Drivers, Pressures, State change, Impacts, and Responses) model
- Identified the most pertinent issues affecting agriculture-wetlands interactions around the world

- Identified appropriate responses to these issues/challenges as a means to encourage "good practice"
- Illustrated through presentation/application of a set of cases that the issues are "real", i.e., that they are valid to a wide set of biophysical and socioeconomic settings

The diversity of wetlands and of agriculture-wetland interactions (AWIs) identified under GAWI was broken down into two major groups:

- Direct, in-situ interactions
- Indirect, basin wide interactions

These relationships span environmental, socioeconomic, and political spheres. Pressure points for change in various dimensions were identified.

GAWI stressed the need for wetlands to be seen as contributors to sustainable development, rather than merely as habitats to convert for immediate human utility, as has been a feature of much historic development. It identified the pressing need to enhance the functioning of wetlands as multiple-use resources, providing a range of ecosystem services (as for example addressed in ▶ Chap. 125, "Economics of Wetland Conservation Case Study: "Systemic Solutions" for Integrated Water Management" and by Everard and McInnes 2013). To achieve this, there is a need to understand the key actors involved in wetland use, the forces which are driving wetland use and their different socioeconomic contexts and impacts.

GAWI undertook a meta-analysis of 92 cases of AWI from all Ramsar regions using the DPSIR Model, with half the case studies coming from Africa and Asia and at least ten cases from each of the other regions. The major findings confirmed, perhaps unsurprisingly, that management for ecosystem services under AWI tended to be skewed towards overexploitation of provisioning services at the expense of regulating and supporting services. In 16% of cases, government policies seeking to encourage food production or to regulate the use of wetland and natural resources were substantive drivers. Drivers prioritizing climate change and climatic variability were conspicuously low or absent, except for Africa where these factors were mentioned in 32% of cases. Cumulatively, drivers of wetland change tended to increase agricultural activities such as agricultural expansion (32%), increased water use/depletion (31%), and agricultural intensification (30%). This in turn contributed to a general reported loss in biodiversity, loss of soil fertility and continuing soil erosion, and deteriorating water quality. These environmental changes relate to diverse socioeconomic impacts including the loss of subsistence agriculture off-set by substantial increases in market-oriented agriculture. However, a small but significant number of the case studies did show sustainable multiple use of wetlands, mostly, but not always, in developing continents and in situations with limited market involvement.

In addressing ways ahead, the GAWI Guidelines recognized the importance of rebalancing the use of ecosystem services as a key objective for achieving sustainability in agriculture-wetland interactions. This rebalancing of ecosystem services may involve a number of interventions which could include:

- Redirecting the drivers of change so that the specific needs of society (which lead to drivers) can be met in other ways, such as through trade, employment and/or nonwetland farming development, any of which could reduce the imbalances in ecosystem services and the negative state changes in wetlands or elsewhere in the river basin system
- Diversifying the wetland ecosystem services used beyond agriculture through the addition of fishing, crafts, ecotourism and payment for environmental services, etc., so as to still meet household needs while reducing the pressures from monoagriculture upon the wetlands and the negative state changes and impacts due to cultivation
- Diversifying the demands on wetlands for different ecosystem services so that nonprovisioning services can generate income, especially through payment for environmental services for regulatory or biodiversity conservation services
- Managing land at the basin level in ways to facilitate the maintenance of a balance of ecosystem services overall, with different ecosystem services provided at different points in the river/stream system
- Improving crop choices to select plants that require less alteration of the wetland ecosystem services, e.g., irrigated rice or taro (*Colocasia esculenta*) as opposed to drainage for cultivation of maize

From this work, it has become clear that there is need for a change in thinking about wetlands and agriculture across the globe. This should involve a move from a situation of competition amongst stakeholders who each seek to achieve singular ecosystem service uses of wetlands to meet their own specific interests – such as agriculture or biodiversity conservation – to a situation where stakeholders work together to achieve a mix of ecosystem services in wetlands, with mutually advantageous multiple benefits which help ensure the sustainability of all ecosystem services and wetlands in the long term. This will require the setting of priorities and policies for the different ecosystem services to be developed or maintained in different parts of a wetland or along a river/stream valley system, in order to accommodate the multiple demands made on wetland ecosystems. The DPSIR analysis helps achieve this by mapping out the socioeconomic demands for specific ecosystem services and the state changes and impacts which result from the development of these ecosystem services, along with their consequences for the balance of services in a wetland system. Overall, a congruent and harmonious functional management strategy must be developed that links each demand from society to a specific ecosystem service but also maintains a balance across the full spectrum of ecosystem services and thereby ensures long-term sustainability. Significant obstacles need to be overcome to improve coherence between scales, necessitating reform of a wide range of "levers" (markets and market-based instruments, regulation, common law, accepted protocols, consistent advice, etc.) to ensure that choices made locally by resource owners regarding land use and ecosystem service outcomes are consistent with higher-level aspirations for optimization across the full range of ecosystem services (Everard et al. 2014).

The GAWI analysis confirmed that, in order to sustain the benefits from wetlands and achieve the required rebalancing, all ecosystem services (provisioning and nonprovisioning) should be put to fruitful use in a wetland or across a wetland network in a stream/river basin, taking a landscape approach. Further, GAWI also identified the need to take account of political priorities and socioeconomic conditions in sustainable management of agriculture in wetlands.

Systemic Solutions for Agriculture-Wetland Interactions

A key emphasis of findings of the Comprehensive Assessment of Water Management in Agriculture (2007), the GAWI Guidelines (Wood and van Halsema 2008), and the Millennium Ecosystem Assessment (2005b) is the need in wetland use to achieve a balanced set of ecosystem services, not merely to maximize a selected set of provisioning services at net detriment to overall system integrity, functioning and contribution to human well-being. This is linked to reconceptualizing wetlands as multiuse resources for development, with the value of all ecosystem services recognized and valued. When that appropriate balance of ecosystem services is achieved, wetlands can provide enhanced benefits in a sustainable manner which will improve livelihoods and economies and the resilience of communities.

This transition demands a radically different approach to decision-making at all scales, especially by considering all ecosystem services and addressing political, economic, landowner, and community expectations, restrictions and reward systems. The scale of change required is substantial; in reality, we are yet to witness the transition required to address the magnitude of sustainability challenges facing humanity.

It is also important to match solutions to different scales. As we have seen in the findings of the GAWI analysis, there is a net transition from subsistence-scale to commercial-scale agricultural land use. Yet, for many communities, sustainable interaction with their supportive environment and particularly its wetland ecosystems operates most frequently at subsistence scale. Advice on sustainable uses of wetlands that support the development of provisioning services, such as agriculture, while retaining overall wetland character and functioning has been in development over many years (for example Everard et al. 1995). As detailed in ▶ Chap. 140, “Flood Recession Agriculture: Case Studies,” some forms of wetland land use for agriculture can work in sympathy with seasonal wetlands without necessarily degrading their characteristics or other associated services.

Furthermore, some traditional, enduring forms of wetland conversion and management have transformed these systems into not only productive but also culturally defining landscapes that have not only survived the rise and fall of civilizations but which provide a range of additional services including cultural cohesion and ecotourism. (A prime example of this is covered in detail in ► Chap. 142, “Rice Paddies”; the ► Chap. 197, “Landscape Aesthetics and Wetlands” also addresses how some wetland agriecosystems may be culturally enriching.) So not all cases of conversion of wetland systems for agriculture result in what may be perceived as a net negative set of associated consequences.

Key issues deduced in the above analysis are:

- Recognition of the contribution of wetland agriculture to a range of livelihoods and development goals
- Identification of the growing threats agriculture can present to the maintenance of wetlands and their full range of ecosystem services
- Recognition of the linkages amongst ecosystem services and the need to keep a balance or a mix of ecosystem services in order to maintain the functioning of wetlands, i.e., for their ecological sustainability
- The need to develop values for communities and wetland owners from the different ecosystem services, especially generating values for neglected ecosystem services such as regulatory and biodiversity services, sometimes through payment for environmental services
- Optimization of total benefits for society from the full range of wetland ecosystem services rather than maximizing the benefits from one or just a selected few
- Recognition of the need to explore different trade-offs between ecosystem services in wetlands, in conjunction with the wider landscape, so as to maximize overall benefits and maintain a mix of ecosystem services
- Use of trade-off methods to address the competing demands and conflicts which can develop between different interest groups with respect to wetland ecosystem services, so ensure sustainability
- Recognition of the role of people in the management of wetlands for ecosystem services maintenance, as wetland users managing agriculture and other provisioning services so as to optimize and sustain overall benefits

Above all, the implications of wetland use and management for all ecosystem services, their spectrum of beneficiaries (or potential victims), net economic consequences where these are all taken account of, and overall system resilience need to be raised as driving considerations. This is becoming progressively better recognized in policy and business environments, particularly through the concept of the “nexus” of interconnected food-water-energy issues (see ► Chap. 121, “Contribution of Wetlands to the Food-Water-Energy Nexus”).

Potential Roles of Wetlands in Reducing the Impacts of Agriculture

However, it is not merely as systems that may be partially used or converted wholesale as agricultural systems that wetlands play significant roles in agricultural management. For example, both natural and constructed wetland systems, including localized wetland features such as buffer zones and protected swales, may have a significant role to play in addressing some of the potentially damaging effects of agricultural activities on rivers and coastal water, other wetland systems and the wider landscape (as addressed in detail in ▶ Chap. 180, "Wetlands in the Management of Diffuse Agricultural Run-Off"). These challenges are exacerbated by a growing human population, increased concentration of populations in cities, and a changing climate.

There are increasing examples of areas of habitat, including wetland systems, being "retired" from narrow production of provisioning services as the importance of a wider set of ecosystem services are progressively becoming better recognized. Important services provided by wetlands which enhance the resilience of agriculture include supporting services such as provision of habitat, and particularly for species that have roles in the regulatory services of pollination, pest regulation, and disease regulation or which represent the provisioning services of genetic, biochemical, natural medicine, and pharmaceutical resources. Wetlands also play important beneficial roles in regulating water flows and natural hazards, erosion, water purification/waste treatment, salinity and fire all of which can be beneficial to agriculture or which can reduce the damage that may arise from agricultural activities. Wetland habitat may also provide valued cultural services in mixed farming landscapes and be a basis for community-building as well as an important educational resource. And the often overlooked supporting services of soil formation, primary production (other than that which is directly used as a farmed output), and the local recycling of water play important roles in supporting the integrity and function of landscapes and their resilience to future change. Prime examples of habitat conservation initiatives recognizing at least a subset of these wider benefits provided by wetlands include subsidized "land retirement" protecting a variety of public benefits under the US Conservation Reserve Programme (see ▶ Chap. 120, "Conservation Reserve Program (CRP): Example of Land Retirement").

Above all, it is important that all wetland-based and other solutions using natural processes focus broadly on optimization of a linked set of ecosystem service benefits rather than becoming narrowly focused on maximization of a single outcome, as to do so would merely substitute one narrow result (for example food production) for another (such as abatement of water pollution) while overlooking implications for the wider suite of benefits provided by ecosystems (habitat for wildlife, carbon storage, nutrient cycling, soil formation, etc.) This is addressed in detail in ▶ Chap. 125, "Economics of Wetland Conservation Case Study: "Systemic Solutions" for Integrated Water Management".

Future Challenges

The future challenges entailed in recognizing and optimizing the role of wetlands in agricultural management are massive. They cut to the heart of how landscapes and waterscapes are used to support not only primarily services of immediate human utility – food, water, biofuels, and so forth – but how they are managed to sustain the overall balance of ecosystem services essential to support humanity's evolving needs now and into the future. These challenges will only be compounded by population growth, continuing urbanization, and the consequences of a changing climate.

These challenges require a radical overhaul of our conceptualization of wetlands, and of established governance and economic systems so as to provide the necessary incentives and compulsions securing genuinely systemic management for optimal outcomes for these multiple-use economic resources.

The role of wetlands in agricultural productivity are diverse, including both direct uses and the indirect supporting, regulatory, and cultural services they provide, adding to the resilience and functioning of wider landscapes. Progressive recognition and internalization of these wider benefits into the policy environment is essential if sustainable agriculture and wetland use are to be achieved for the net security of humanity.

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Flood Recession Agriculture: Case Studies

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Mark Everard

Contents

Abstract

Flood recession agriculture is a common practice in many regions of the world on river floodplains, lake margins, and other wetlands where water levels rise and fall predictably. As water levels recede, wetted and nutrient-rich soils are exploited for their high agricultural productivity, with crops harvested before rains return inundating the landscape once again generally on a seasonal basis. The importance of flood recession agriculture may be substantial in arid and semi-arid areas with intermittent rains. It is significant in supporting subsistence needs, but may be compromised by damming and other schemes that perturb natural catchment hydrology. Flood recession agriculture reflects a 'knowledge' economy, local practices and crop choices closely linked to the unique topography, hydrology and other characteristics of flood recession areas.

Keywords

Senegal · Ethiopia · Awash catchment · Flood-retreating crops · Traditional knowledge · Conflict · Subsistence

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Introduction

Flood recession agriculture is a common practice in many regions of the world on river floodplains, lake margins, and other wetlands where water levels rise and fall predictably. As water levels recede, wetted and nutrient-rich soils are exploited for their high agricultural productivity, with crops harvested before rains return inundating the landscape once again generally on a seasonal basis.

This is a particularly important practice where large rivers cross otherwise arid landscapes, bringing moisture and fertility to soils as in the case of the three great river systems of Deccan India (the Cauvery, Krishna, and Godavari) and many of the greater rivers of Africa. Indeed, large rivers crossing the arid and semiarid parts of Africa have figuratively been called the "…lifeline of the impoverished nations" (Darkoh 1992).

However, this semipredictable pulse of wet and dry periods is threatened both by an increasingly unstable climate as well as by the damming of rivers. Damming not only potentially interferes with river flows but also generally obstructs the free flow and fertility of silt as well as contributing to soil salinization where year-round irrigation occurs (Everard 2013). Competition for water can also place pressures on water and wetland resources.

Case Study 1: The Senegal River Valley

Saarnak (2003) explored the system dynamics of the Senegal River valley in West Africa, including the importance and future of the low-yielding flood recession farming systems that are practiced there. Damming and the introduction of irrigation had caused changes to the hydrology of the Senegal River, exacerbated by periodic drought. The traditional flood recession farming system had persisted for many years, dependent upon natural irrigation and fertilization of the floodplain by the seasonal overspill of the Senegal River. This enabled local people to grow crops such as sorghum, beans, and melons, making use of residual soil moisture as water levels receded from yearly inundations of the river valley in August and September. This form of flood recession farming relied upon no controllable input other than land and labor, therefore providing a very high net return per unit of energy expenditure.

Nevertheless, labor productivity and net income from input-intensive irrigated farming systems enabled by damming of the river and rerouting of its flows far exceeded that of traditional flood recession farming, as well as perturbing river flows upon which the recession agriculture depended. Irrigated farming was introduced into the river valley during the 1980s, leading to an expansion of pump-based irrigated farming on the higher grounds close to the river that had become marginal for recession farming. Nevertheless, even where irrigated farming took place, flood recession farming remained important at the household level for subsistence needs, supporting household subsistence during months where other contributions to the household income were limited. There were conflicts between the two forms of farming, with manipulation of water releases in dams for irrigated farming interfering with flood recession agriculture affecting its yield and contribution to local livelihoods and security.

As farmers have no control over water flows to the recession fields, they lost incentives to invest in field development. For this reason, the future potential for recession agriculture is likely to decrease as population and subsistence demands increase and irrigated agriculture is further developed. The future of flood recession agriculture therefore depends heavily on the willingness of dam managers to emulate natural flood regimes, sustaining local livelihoods. The final choice of allocation of water resources is between increased production and secure supply of hydroelectricity to a large urban population and industry, compared to flood recession production that at present is of importance to the subsistence of a large agricultural population in the Senegal River Valley.

In this regard, the conflict is one of ideology between, on the one hand, prioritizing subsistence and sufficiency and, on the other, engineering-centered industrialization.

Case Study 2: The Upper Awash Catchment, Ethiopia

Nederveen (2010) undertook a study of flood recession farming in the Upper Awash river in Ethiopia, in the horn of Africa. Agriculture in Ethiopia is of high economic importance, accounting for almost 45% of the GDP and about 85% of the export (Factbook 2010). The areas in Ethiopia where flood recession farming is practiced all have their own history and agricultural methods adapted to local conditions with the principal differences comprising the types of crops cultivated, planting techniques, combination with other sources of income related with the flood (such as fisheries), and methods to manage risks. In some regions, rice is a preferred crop during the wet season as river levels rise, using small bunds to retain water. In some regions, flood recession agriculture is practiced with maize and sorghum grown on the residual moisture after the flooding period.

Flood recession farming is practiced at seasonal flooded areas in the Upper Awash Catchment. Some farmers also use pumps to draw water from rivers or wells to irrigate crops such as orange trees, mango trees, and other horticultural crops throughout the year. The hydrology of the Upper Awash River catchment and the topography of its surrounding land have profound effects on the productivity of flood recession agriculture in study sites in Ethiopia. As floods recede at the end of September, farmers plow the exposed soils before sewing a variety of crops according to different physical characteristics in each floodplain plot. Crops best suited to varying soil conditions produced by the local variations include chickpeas and lentils, maize, grass peas and haricot beans, as well as tomatoes, onions, and melons.

The differences between the timing and duration of inundations determine cropping strategies. Upstream areas that are also slightly elevated are inundated for shorter periods, whereas downstream and lower elevation areas remain inundated for longer, favoring different crops and husbandry.

Flood recession farming therefore relies on the traditional knowledge and expert judgment of farmers as to when to start plowing, crop selection, sowing, and harvesting.

Conclusions and Future Challenges

Flood recession agriculture is a "knowledge economy" using traditional knowledge and experience of the timing of natural flows of water and nutrients, of crop selection and ideal planting locations, and stewardship methods. This does not fit easily with modern technological manipulation of river flows for energy production or diversion of water to serve large-scale, commercial irrigation schemes. Excessive groundwater and surface water pumping can also perturb the hydrological regimes upon which flood recession agriculture depends.

Recognizing the value of flood recession agriculture and its consequent protection is important, as it supports the needs and livelihoods of many people, particularly across the developing world.

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Food from Wetlands

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Mark Everard

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Abstract

The world's diverse wetlands are vital sources of food for the global population. Adequate, good quality food is a prerequisite for healthy people, and wetlands are key contributors, supplying the global population with a broad range of wild and cultivated food sources such as fish (including shellfish); certain mammals; plants (rice, seaweeds, a range of leafy vegetables, fruits, and nuts, etc.); reptiles; amphibians; insects and other arthropods; snails, and a diversity of other organisms.

Harvesting fish resources from coastal and inland waters has been a source of sustenance and livelihood for millennia. Today one billion people, largely in developing countries, rely on fish as their main or sole source of animal protein and many more consume fish regularly. The fish we eat comes from both capture fisheries (62% in 2008) and aquaculture (38%) and both are heavily dependent on healthy coastal and inland wetlands. Achieving sustainability is a major challenge –75% of our commercially important marine stocks of fish were being overfished in 2008 and so too are many inland fish stocks; aquaculture, now the fastest-growing sector in the food production industry in the world, brings its own share of sustainability challenges through the side effects of pollution, habitat destruction (especially mangroves), escape of nonnative species, etc. While the loss or diminished availability

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of fish protein may be unwelcome for many of us, for most of the one billion people who are both poor and dependent on fish, failing fish harvests can have direct effects on their health through malnutrition leading to higher susceptibility to diseases.

Perhaps the most well-known wetland plant is rice, now largely a cultivated plant. Nearly 50% of the world's population, mainly in Asia, depends on rice as a staple food item. Other wetland plants, such as seaweeds (both naturally occurring and cultivated), also play a significant role in food supply, and many local communities rely upon a number of leafy plants in wetlands as a source of vegetables in their daily diets.

Keywords
Food · Fish · Shellfish · Rice · Paddy · Aquaculture

Overview of Wetland Food

The Ramsar Commission (2008) produced a summary document *Wetland food* marking World Wetlands Day 2008, from which the description in this chapter section is derived.

Rice provides an interesting wetland food example that is replicated in many wetlands. For many communities, particularly in Asia, rice is grown for livelihoods as well as for home consumption, but padi fields can provide much more than rice. In a study in Lao PDR (reported in Ramsar Commission 2008), at least 17 of the 26 fish species that were harvested as a main source of protein occurred in padi fields and six of them bred there. In addition, villagers consumed five amphibian species and at least two reptile, two prawn, four mollusk, and 10 aquatic plant species from the same source.

Our wetlands will continue to provide food to keep people healthy. However, many human actions negatively affect the ability of wetlands to continue to provide for human needs. Pollution, excessive water abstraction, poor sanitation, overharvesting and, of course, wetland destruction all degrade or destroy the capacity of wetlands to provide food for human consumption.

Wetland Food Webs

Wetlands are also important food sources for nonhuman organisms. As the US Environmental Protection Agency (US EPA n.d.) describe in communication materials, "Wetlands are among the most productive ecosystems in the world, comparable to rain forests and coral reefs... High Mountain Valley Wetland Wetlands can be thought of as 'biological supermarkets.' They provide great volumes of food that attract many animal species. These animals use wetlands for part of or all of their lifecycle. Dead plant leaves and stems break down in the water to form small particles of organic material called 'detritus.' This enriched material feeds many small aquatic insects, shellfish, and small fish that are food for larger predatory fish, reptiles, amphibians, birds, and mammals.” This description then highlights how wetlands make important contributions to food production in wider, connected habitats.

The productivity of freshwater wetland soils also means that many have been exploited through agricultural conversion, which is implicated as the most significant global driver of wetland loss and degradation (Millennium Ecosystem Assesment 2005).

Conclusions

Wetlands provide diverse forms of food for the global population for many of whom, mainly in the developing world, wetland sources are essential. Wetlands do so directly, for example, through capture fisheries or harvesting of plants and other organisms, and as a secondary contribution through supporting adjacent ecosystems as well as use for aquaculture and agricultural conversion.

Balancing exploitation with conservation, also reflected as Principle 10 of the Convention on Biological Diversity’s Ecosystem Approach (http://www.cbd.int/ecosystem/principles.shtml), is a pressing sustainable development challenge as global populations rise with increasing demands on ecosystems.

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Rice Paddies

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Abstract

Rice is the best-known wetland crop. Rice is also the most widely consumed staple food globally, a major part in the diet of more than half the world's population especially in Asia. In 2010, rice is also the grain with the second-highest worldwide production after maize. The importance of rice has been recognized for many centuries, as for example in India where it was once known as "dhanya" meaning "the sustainer of the human race".

The edible portion of rice comprises seeds from two species of wetland grasses, *Oryza sativa* (Asian rice) and *Oryza glaberrima* (African rice). Asian rice is the dominant crop grown for subsistence and commercial purposes globally, with many cultivated strains across two principal subspecies: the sticky, short-grained variety (japonica or sinica); and the non-sticky, long-grained variety (indica).

Paddy systems have persisted for over six millennia, their efficient productivity and retention of water, soil and nutrients leading to pervasive implementation across much of the tropical and subtropical world. Their central importance to communities means that they can be an organizing principle, also attracting spiritual and tourism

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importance. Greenhouse gas emissions from paddy fields may be significant, so improved stewardship systems are required to reduce associated emissions.

Keywords
Rice · Oryza · Paddy · Padi · Buffalo · Banaue Rice Terraces · Traditional knowledge · Polyculture · Methane · Social infrastructure

Introduction

Rice is perhaps the best-known wetland crop. Rice is also the most widely consumed staple food globally, a major part in the diet of more than half the world's population especially in Asia. The importance of rice has been recognized for many centuries, as for example in India where it was once known as "dhanya" meaning "the sustainer of the human race" (Kew n.d.).

Based on data published in 2010, rice is also the grain with the second-highest worldwide production, after maize (FAO 2014). Over 400 million tonnes of milled rice is produced globally each year (Kew n.d.).

Rice basically comprises the seeds from two species of wetland grasses, *Oryza sativa* (Asian rice) and *Oryza glaberrima* (African rice). Of these, Asian rice is the dominant crop grown for subsistence and commercial purposes globally, with many cultivated strains across two principal subspecies: the sticky, short-grained variety (*japonica* or *sinica*); and the non-sticky, long-grained variety (*indica*). Optimal cultivation of these strains varies, with *japonica* usually cultivated in dry fields in temperate East Asia, upland areas of Southeast Asia, and high elevations in South Asia, while *indica* varieties are mainly grown semisubmerged in water in lowlands throughout tropical Asia and many other parts of the world. By contrast, African rice (*Oryza glaberrima*) is believed to have been domesticated from a wild African ancestor, *Oryza barthii*, in the inland delta of the Upper Niger River in modern-day Mali, West Africa. African rice is today a staple food throughout West Africa and is also prized for its delicate, nutty taste. It grows best in the alluvial soils that remain the principal habitat of its progenitor wild species, and it is also more resilient than Asian rice in its growing requirements in terms of fluctuations in water level, iron toxicity, soil infertility, severe climatic conditions, and human neglect.

Rice Paddy

Rice paddy, or padi, is derived from the Malay word "padi", meaning "rice plant". Rice paddy comprises flooded parcels of arable land used for the growing of rice. Archeological evidence suggests that wet-field cultivation originated in China, with the earliest paddy field dated to 6280 years before present based on carbon dating of grains of rice and soil organic matter (Cao et al. 2010). So efficient a system of production is paddy field farming that it has not only endured, but is now a widespread global production system. Today, it occurs in Cambodia, Bangladesh, China, Taiwan, India, Indonesia, Iran, Japan, North Korea, South Korea, Malaysia, Myanmar, Nepal, Pakistan, the Philippines, Sri Lanka, Thailand, Vietnam, Laos, and parts of Italy, the Camargue in France, Haiti, and California (USA).

Paddies can be built into steep hillsides as terraces, sometimes engineered over centuries by dependent communities as an efficient means to sustain themselves while conserving water, soil, and nutrients. Paddy can also be found in depressed or steeply sloped landscape features, such as river floodplains or marshes. Rice is also a common crop grown in the flooded period of flood recession agriculture. Rice paddy systems require considerable labor to create and maintain; a great deal of it effected by hand although the wetland-adapted water buffalo is important in paddy field farming throughout Asia.

Some extensive paddy systems built as cascades of paddy fields in steep landscapes, such as the Banaue Rice Terraces in the Philipines, are as much as 2000 years old, carved by successive generations of local people into the mountain sides. Similar extensive systems are found across the Far East, the Indian subcontinent, some hill slopes of Africa, and in some landscapes with similar topography in South America. So imposing are some systems, including Philippine's Banaue Rice Terraces, that they are considered wonders of the world, nowadays with considerable tourism value.

In Asia, the overwhelming majority of rice is still produced within walking distance of where it is eaten (Codrington 2005) using centuries-old traditional knowledge to intercept natural flows of water through landscapes. Though "low tech" in western industrial terms, the terraced cultivation systems common throughout Asia are remarkably efficient, conserving not only water but also soil and nutrients. Polyculture is also commonly practiced in rice paddy, small fish in particular introduced when the paddy is flooded and then harvested as a source of protein as fields are drained adding to their value and efficiency. Rice straw is also a valuable dry weather source of fodder as well as bedding both for stock and people.

Notwithstanding their many wider sustainability credentials, paddy fields are also a major source of atmospheric methane, estimated globally to contribute in the range 50–100 million tonnes of the gas per annum (GHG online n.d.; Tampa Bay Times 2007).

Rice as an Organizing Principle in Cultures and Livelihoods

Rice cultivation is the principal activity and source of income for about 100 million households in Asia and Africa (Umadevi et al. 2012). Built largely by hand, maintained over the centuries by the constant attention and labour of communities shaped by the need to tend these systems for mutual food security, this type of collaborative activity also binds communities throughout the tropical world and across time. Human as well as technological infrastructure is vital for the continued functioning and productivity of these paddy systems. However, this common stewardship shapes communities and livelihoods, influencing their timing in response to monsoons and other weather systems. Often, rice and traditional rice-growing systems also have notable spiritual significance, stemming for the bonding of people around common stewardship of nature's productive capacities.

This form of localized communal approach to the tapping of environmental flows of water, both through soils and in the form of capture of rainfall to meet local needs, has underpinned great civilizations upon which empires have been built, even if the vital technologies supporting the livelihoods of most of their people have been almost entirely overlooked. Cultures may come and go, but the system persists (Pearce 2004).

Conclusions and Further Challenges

Rice is the globally dominant form of wetland food, supporting both the subsistence needs of billions of people but also valuable trade. Paddy systems have persisted for over six millennia, their efficient productivity and retention of water, soil and nutrients leading to pervasive implementation across much of the tropical and subtropical world. Greenhouse gas emissions from paddy fields may be significant, so improved stewardship systems are required to reduce associated emissions.

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Lake Bed Cropping: Wetland Products (Australia)

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Sue Briggs

Contents

Abstract

Lakebed cropping is cropping on the bed of a lake while it is dry. The following types of lakebed cropping are undertaken in the Murray-Darling Basin in inland south-eastern Australia. They are: (i) cropping once only following recession of floodwaters; (ii) cropping once following rainfall; (iii) cropping following recession of floodwater or rainfall, followed by cultivation for a second crop; (iv) cropping between floods, with repeated cultivation to keep the ground bare between crops; (v) cropping following occasional release of water; and (vi) cropping following regular, usually annual water releases. The impacts of lakebed cropping on soil and on dryland and wetland biota depend on the type of lakebed cropping. Compared with other forms of cropping, most types of lakebed

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cropping in inland south-eastern Australia have low or moderate ecological impacts. The chapter concludes with recommendations for managing lakebed cropping.

Keywords
Lakebed cropping · Murray-Darling Basin · Soil cracks

Introduction

Lakebed cropping is cropping on the bed of a lake while it is dry. Lakebed cropping has historical origins and is practised in several countries (Siemens 1983; Scoones 1991; Briggs and Jenkins 1997; Nagabhatla et al. 2010; Turyahabwe et al. 2013). This chapter is about lakebed cropping in the Murray-Darling Basin (Fig. 1) in inland southeastern Australia.

Most lakes in inland southeastern Australia flood and dry out periodically. They flood for 1–3 years and are then dry for a decade or longer. Sometimes two floods occur in quick succession. The dual wet-dry nature of the lakes provides habitat for dryland biota and for wetland biota as well as opportunities for periodic cropping. Cropped lakes in the Western Division of New South Wales in inland southeastern Australia are usually large (over 1,000 hectares) and are intermittently inundated by freshwater from a river (Seddon and Briggs 1998). Most lakebed cropping in inland southeastern Australia (hereafter southeastern Australia) commenced in the late 1970s after the big floods of that decade receded.

Most lakebed cropping is sporadic (Briggs and Jenkins 1997). Crops are planted opportunistically when soil moisture levels are sufficiently high (either from receding floodwaters or rainfall), rather than every year or in a particular season. Lake soils are naturally fertile, and lakebed cropping usually has low inputs. Crops are commonly grown organically (Briggs and Jenkins 1997), without fertilizers, pesticides, or herbicides. Crops planted on lakebeds include wheat, barley, oats, forage and grain sorghum, safflower, canola, mustard, sunflowers, chickpeas, and dryland cotton.

Types of Lakebed Cropping

There are several types of lakebed cropping depending on frequency of cultivation and whether crops are sown once following flood recession, sown on rainfall, sown following water releases, or a combination of these. Six types of lakebed cropping are practised in southeastern Australia (Briggs and Jenkins 1997). They are: (i) cropping once only following recession of floodwaters; (ii) cropping once following rainfall; (iii) cropping following recession of floodwater or rainfall, followed by cultivation for a second crop; (iv) cropping between floods, with repeated cultivation to keep the ground bare between crops; (v) cropping following occasional release of water; and (vi) cropping following regular, usually annual water releases.

Fig. 1 Map of the Murray-Darling Basin in southeastern Australia. Most of the lakes in the Murray-Darling Basin flood and dry intermittently. The Murray-Darling basin contains many more lakes than are shown on the map (see Seddon and Briggs 1998; Kingsford et al. 2004). (Image credit: Murray-Darling Basin Authority © copyright remains with the author)

Two further types of cropping are sometimes, erroneously, called lakebed cropping. They are irrigated cropping on lakebeds and cropping on lakebeds from which water has been permanently or near permanently excluded.

Ecological Impacts of Lakebed Cropping

Ecological impacts of lakebed cropping on soil and on dryland and wetland biota depend on the type of lakebed cropping (Briggs and Jenkins 1997; Seddon and Briggs 1998) (Table 1). Frequent cultivation of lakebeds reduces soil structure and

Table 1 Levels of impacts from different types of lakebed cropping. Impacts assume lakebed cropping is organic (no fertilizers or herbicides), and no perennial vegetation, including lignum *Muehlenbeckia florulenta*, is removed. These assumptions do not apply to irrigated cropping or to dryland cropping on lakebeds from which floodwater has been permanently excluded. The assumption that lignum is not removed does not necessarily apply to lakebed cropping following regular water releases and sometimes not to the other higher impact forms of lakebed cropping (Adapted from Briggs and Jenkins (1997) and Seddon and Briggs (1998))

Type of cropping	Frequency of cultivation/ cropping	Effects on soil	Ground vegetated between crops	Soil cracks reform between crops	Impact on dry lake wildlife – small mammals and reptiles	Impact on flooded lake wildlife and invertebrates	Impact on water regime	Overall ecological impact
Cropping once only following recession of floodwaters	Low	Low	Yes	Yes	Low	Low[a]	Not altered[b]	Low
Cropping once following rainfall; cropping can occur relatively frequently in higher rainfall areas	Low; moderate in higher rainfall areas	Low; moderate in higher rainfall areas	Yes	Yes	Medium; high in higher rainfall areas	Low[a]	Not altered[b]	Medium; high in higher rainfall areas
Cropping following rainfall or recession of floodwaters, with cultivation for a second crop	Low; moderate in higher rainfall areas	Low; moderate in higher rainfall areas	Sometimes	Sometimes	Medium	Low[a]	Not altered[b]	Medium
Cropping between floods, with repeated cultivation to keep the ground bare between crops	High	High	No	No	High	Low[a]	Not altered[b]	High

Cropping following occasional release of water[c]	Low	Probably low	Yes	Yes	Medium	Low[a]	Moderate	Medium
Cropping following regular, usually annual water releases[d]	High	Probably moderate	No	Not usually	High	Medium; high when water regime resembles controlled irrigation	High	High
Irrigated cropping[e]	High	High	No	No	High	High	Very high	Very high
Dryland cropping on lakebeds from which floodwater water has been permanently excluded[e]	High, unless minimum or no till practices are used	High, unless minimum or no till practices are used	Not usually	Not usually	High	High	Very high	Very high

[a]Cultivating lakebed soils reduces the number of rotifers that emerge from the soil after flooding but does not reduce the number of crustaceans that emerge after flooding (Jenkins 1995). Lakebed cropping without clearing perennial vegetation and without altering water regimes is likely to have low effects on waterbirds because their habitat and food sources are not affected

[b]Water is sometimes temporarily excluded from lakes by temporary soil banks, until the crop is harvested

[c]Cropping occurs after natural floods as well as following water releases

[d]Can have impacts similar to irrigated cropping when water releases resemble controlled irrigation

[e]Included for comparison only. This chapter is not about irrigated cropping on lakebeds or dryland cropping on lakebeds from which water is permanently excluded. These types of cropping are not lakebed cropping. Information on impacts of these types of cropping is in McKenzie et al. (1991), Chan et al. (1995), Kingsford (2003), Lemly et al. (2000), and Arthur et al. (2011)

levels of organic carbon, whereas once only cultivation and cropping following recession of floodwaters does not affect these soil properties (Briggs 1994; Jenkins and Briggs 1995; Briggs and Jenkins 1997; Seddon and Briggs 1998). Briggs (1996) found that abundance and diversity of small mammals and reptiles were lower on parts of lakebeds which had been cropped and cultivated frequently, compared with uncropped areas of the same lakebeds. Small mammals reinvaded dry lakes following cropping, providing the ground was not cultivated after the crop was harvested (Briggs 1997). The following material on ecological impacts of lakebed cropping is taken from Briggs and Jenkins (1997), except where otherwise stated. Impacts of lakebed cropping are summarized in Table 1.

Cropping Once Following Recession of Floodwaters

In this type of lakebed cropping, the ground is cultivated once and a crop is sown once on lakes as they dry following natural flooding. The ground is not plowed after the crop is harvested, and no further crop is sown until the lake floods and redries. The crop is sown onto bare soil with no cracks or holes (since the soil is still moist from the floodwaters) and no vegetation is cleared to sow it. The soil cracks and holes reform during the crop phase, as the lake soil dries. These cracks and holes remain intact during harvest. The normal vegetation of the lake regrows after the crop is harvested, and small mammals and reptiles reinvade the dry lake (Briggs 1997). Cropping once following recession of floodwaters has low ecological impacts.

Cropping Once Following Rainfall

In this type of lakebed cropping, the ground is plowed and a crop is sown once following rain. The ground is not plowed following harvest of the crop, and no further crop is sown until the next major rainfall event. Annual vegetation is usually removed when cultivating to sow the crop. Depending on how much rain falls, soil cracks and holes are usually present in the lakebed prior to cultivation; these cracks and holes are plowed over. Soil cracks and holes in dry lakes provide habitat for small mammals and reptiles and invertebrates (Read 1987; Briggs et al. 2000; S. Briggs, personal observation). Cropping once following rainfall has higher impacts than cropping once following floodwater recession, because native vegetation is usually removed, and cracks and holes are filled in by plowing. This type of lakebed cropping has fewer ecological impacts than the more intense forms of lakebed cropping below, except when practised in higher rainfall areas where crops can be sown relatively frequently following rainfall. Repeated cropping on lakebeds following rain has relatively high impacts (similar to dryland cropping on lakebeds except floodwaters are not excluded).

Cropping Following Rainfall or Recession of Floodwater, Followed by Cultivation for a Second Crop

In this type of lakebed cropping, the ground is plowed following harvest of the first crop (sown after rain or on recession of floodwaters) in the expectation that a second crop will be planted. The success of the second crop depends on soil moisture, which depends on rainfall. The normal lake vegetation usually regrows after the second crop is harvested. This type of lakebed cropping has higher ecological impacts than the forms of lakebed cropping above because the ground is plowed twice, rather than once, and the return to native vegetation and recolonization by native small mammals and reptiles is delayed by two crops with cultivation between the crops. Sometimes the ground is cultivated following harvest of the first crop, but a second crop is not planted due to lack of rain.

Cropping Between Floods, with Repeated Cultivation to Keep the Ground Bare Between Crops

The first crop in this type of lakebed cropping is usually sown following receding floodwaters. The lakebed is then repeatedly cultivated to keep the soil bare and cropped following rainfall. Native vegetation does not recolonize the lakebed, as the ground is kept bare between crops. As with the above forms of lakebed cropping, the lake continues to flood and dry naturally or semi-naturally. This type of lakebed cropping has higher ecological impacts than the above types of lakebed cropping because the ground is plowed repeatedly, and native vegetation and holes and cracks do not reform in the lakebed soil between crops. This type of lakebed cropping only occurs where rainfall is sufficient to support repeated cropping.

Cropping Following Occasional Release of Water

Some lakes are cultivated and cropped following occasional water releases as well as following recession of floodwaters and sometimes following rainfall. Cropping following occasional water releases is a rare form of lakebed cropping. The ecological impacts of this type of lakebed cropping depend on how frequently it occurs. Cropping following occasional releases of water is likely to have low ecological impacts. Cropping following frequent releases of water will have the same ecological impacts as cropping following regular, usually annual water releases.

Cropping Following Regular, Usually Annual Water Releases

This is a highly managed form of lakebed cropping. It has high ecological impacts compared with other forms of lakebed cropping (except for repeated cropping interspersed with cultivation to keep the ground bare, which also has high impacts). Cropping following regular, usually annual water releases has ecological impacts from the changed water regime of regular flooding and from the infrastructure required to achieve this (Arthur et al. 2011; Kingsford 2003). This type of cropping occurs more frequently on floodplains than on lakebeds.

Two other types of cropping are sometimes, erroneously, called lakebed cropping. The first of these is irrigated cropping on lakebeds, which has similar high ecological impacts as irrigated cropping elsewhere (ee McKenzie et al. 1991; Lemly et al. 2000; Arthur et al. 2011). Irrigated cropping on lakebeds should not be referred to as lakebed cropping, because such cropping is the same as all other irrigated cropping. The second type of cropping, sometimes also erroneously referred to as lakebed cropping, is cropping on lakes from which floodwaters have been permanently or near permanently excluded. Cropping on lakes which have been drained or from which floodwaters have been excluded has similar impacts as dryland cropping elsewhere, including impacts on soil when repeatedly cultivated and cropped (Chan et al. 1995), loss of habitat for small mammals and reptiles, loss of bird habitat if perennial vegetation is cleared, and impacts on habitats of waterbirds, fish, and other aquatic species from preventing natural flooding.

Benefits of Lakebed Cropping

Compared with other forms of cropping, most types of lakebed cropping in inland southeastern Australia have low or moderate ecological impacts. Although sometimes financially risky (due to unexpected second floods or unanticipated dry weather), lakebed cropping provides considerable economic benefits to farmers and graziers and to the rural and regional areas where it occurs. Lakebed cropping is an important form of diversification in inland southeastern Australia. Most types of lakebed cropping are consistent with the goals in the Brundtland Report for sustainable development: “Meeting the needs of the present without compromising the ability of future generations to meet their own needs” (United Nations 1987). No economic analyses have been undertaken on financial benefits of lakebed cropping for rural communities. The ratios of economic benefits to adverse ecological impacts from lakebed cropping are likely to be high. Economic analyses of the direct and indirect financial benefits of the different types of lakebed cropping would be a fruitful area for research.

Recommendations for Managing Lakebed Cropping

Summarized recommendations from Briggs and Jenkins (1997) and Seddon and Briggs (1998) for managing lakebed cropping are:

(i) Management of lakebed cropping needs to take into account which type of lakebed cropping is being undertaken.

(ii) At least 15% of the area of lakes which are cropped only once after flooding should be left uncultivated and uncropped. This 15% should comprise a band of uncultivated and uncropped land around the perimeter of the lake adjacent to the surrounding woodland, usually black box (*Eucalyptus largiflorens*), coolibah (*Eucalyptus coolabah*), or river red gum (*Eucalyptus camaldulensis*) and an uncultivated and uncropped area near the center of the lake. Vehicles should not be driven in either area nor should they be heavily grazed.

(iii) At least 25% of the area of lakes which are repeatedly cropped and cultivated between crops should be left uncultivated and uncropped. This 25% should comprise a wider perimeter band than on lakes cropped and cultivated once only following floods plus an uncultivated and uncropped area near the center of the lake. Vehicles should not be driven in either area nor should they be heavily grazed.

(iv) Some landholders allow a sharefarmer to cultivate and crop their lake, with the proceeds from the crop split between the landholder and the sharefarmer. Landholders should include environmental conditions for cropping and cultivating their lakes in the financial contract with their sharefarmer.

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Swamp Wetlands: Provisioning Services 144

Mark Everard

Contents

Abstract

Swamps have been a traditional source of food throughout human history. Food sources range from plants such as lotus and the fruits of many other species including blueberries (shrubs of the genus *Vaccinium*), in addition to a diversity of fishes, amphibians and reptiles, waterfowl, mammals, and invertebrates. However, wetlands also produce a diversity of other provisioning services ranging from building materials to medicinal and ornamental plants, animals, and minerals.

Keywords

Food · Building materials · Medicines · Fuel · Ecosystem services

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Introduction

Swamps have been a traditional source of food throughout human history. Food sources range from plants such as lotus and the fruits of many other species including blueberries (shrubs of the genus *Vaccinium*), in addition to a diversity of fishes, amphibians and reptiles, waterfowl, mammals, and invertebrates. However, wetlands also produce a diversity of other provisioning services ranging from building materials to medicinal and ornamental plants, animals, and minerals.

Provisioning Services

The Millennium Ecosystem Assessment (2005) classification of ecosystem services lists a range of types of provisioning services: things that can be extracted from ecosystems to support human needs. (A more detailed definition of "Provisioning Services" appears as a separate chapter in this Wetland Book.) These types of provisioning services are listed below, together with some of the products of swamp wetlands associated with them.

Fresh Water

Swamp wetlands can serve as important fresh water sources for a range of human uses: domestic, industrial, and for irrigation and stock watering. Although physico-chemical purification of water and water flow regulation are themselves regulatory services, and water retention and recycling is a supporting service, swamp wetlands perform these processes efficiently maintaining the fresh water resource for productive human uses as well as the benefit of wider connected ecosystems.

Food

Food production from swamp wetlands can be significant, as outlined above and discussed in far greater depth in the Wetland Book ▶ Chap. 141, "Food from Wetlands." Many communities are wholly or largely dependent on the productivity of swamp wetlands, particularly in tropical and developing nations. Swamp wetlands also sustain species used as food resources in connected habitats.

Fiber and Fuel

The production of fiber such as wood, *Papyrus*, kapok, the fur of wetland fauna, and a range of other sources serves a multiplicity of human needs as diverse as clothing and materials used for construction and the manufacture of crafts and musical instruments. Plant fiber too, particularly timber but also peat, is an important fuel source for many communities. There is also increasing interest in algal biofuels, as well as widespread use of wetland tree species for bioenergy production such as short-rotation coppiced willow species and palm oil plantations. Other fibers, such as the "fur and feather" of swamp wetland species, may be appreciated by diverse users such as anglers tying flies to fool the fish they pursue.

Genetic Resources

The genetic resources provided by swamp wetlands include species and genetic strains that may be exploited for crop and stock breeding. For example, modern rice strains derive from wild species of wetland grasses (as discussed in ▶ Chap. 142, "Rice Paddies"), and modern cattle are largely derived from aurochs that used to graze floodplains prior to domestication and selective breeding. Further species and strains of both plants and animals and potentially also of fungi may come into domestication in future, particularly where they are adapted to local conditions. Enhancement of strains of crops and stock may also be achieved by crossing with wild species possessing desirable traits, such as resistance to diseases and saline soils. Transfers of genetic material from wild strains may also be achieved through biotechnology.

Biochemicals, Natural Medicines, and Pharmaceuticals

Many modern medicines derive from wild species of plants and animals. Many are still extensively used and relied upon across the developing world in traditional medicines. As a storehouse of valuable pharmacological resources, many under-researched, but potentially of great value, swamp wetlands warrant further exploration and protection. (▶ Chap. 151, "Medicinal Plants in Wetlands" are addressed in more detail as a chapter in this Wetland Book.)

Ornamental Resources

Swamp wetlands provide various products and materials that may have decorative or other symbolic value. These include shells, seeds, flowers, shaped wood and aggregates, plant stems, and animal remains and products, among others. Arguably, fish, terrapins, and other swamp wetland species collected and reared as pets are a form of ornamental provisioning service, as indeed are plants brought into horticulture.

Additional Provisioning Services

Although not explicitly listed in the Millennium Ecosystem Assessment classification, some other substances potentially extracted from swamp wetlands constitute provisioning services. These include, for example, aggregates of various types; water systems sorting and laying down silt, gravel, cobbles, and rocks all of which may be extracted for a diversity of human uses.

Conclusions and Future Challenges

Swamp wetlands provide a broad diversity of provisioning services addressing a range of human needs. The key challenges entailed in their future management are to control extraction of these services; to avert conflicts between their use and the production of other services (including other provisioning services but also regulatory, cultural, and supporting services); and to manage swamp wetlands to optimize benefits including system integrity and resilience.

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145 Wetland Management for Sustainable Fisheries: Overview

Mark Everard

Contents

Abstract

Fisheries support the protein needs and livelihoods of more than a billion people globally, particularly in the developing world, as well as serving often lucrative international markets. Fisheries and aquaculture are also, whether directly or indirectly, a source of livelihood for over 500 million people, mostly in developing countries. Fisheries comprise both capture fisheries and aquaculture, and include a diversity of fin-fish and shellfish species. All are dependent on the vitality of a range of wetland types. Sustainable fishery management is a complex socioeconomic and ecological undertaking, requiring a systemic, participatory, and adaptive approach.

Keywords

Sustainable management · Participation · Maximum sustainable yield · Participatory planning · Conflicts

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Introduction

Fisheries support the protein needs and livelihoods of more than a billion people globally, particularly in the developing world, as well as serving often lucrative international markets. Fisheries and aquaculture are also, whether directly or indirectly, a source of livelihood for over 500 million people, mostly in developing countries (FAO 2009). Fisheries comprise both capture fisheries and aquaculture, and include a diversity of fin-fish and shellfish species. All are dependent on the vitality of a range of wetland types.

The Concept of a Sustainable Fishery

The technical concept of a sustainable fishery is relatively straightforward to define. In essence, it is one that is harvested at a sustainable rate such that the fish population does not decline over time because of fishing practices.

However, practical realization of a sustainable fishery is a far more complex matter than adherence simply to the foundational science of fish population dynamics and the identification of thresholds beyond which stocks are likely to decline in quantity and quality. It has also to encompass diverse societal needs, and the formation of multistakeholder agreements including about curtailing destructive, illegal and excessive fishing practices, and the legal and informal protocols that ensure resource conservation. Also, of course, fish are just part of wider ecosystems, the conservation of which is intimately interconnected with the sustainability both of fish stocks and the livelihoods that may depend upon them.

Sustainable fisheries thus depend on a network of informal agreements, laws, and policies, including potentially the establishment of protected areas, suitable economic drivers and market-based interventions, education and consensus-building amongst stakeholders and the wider public, and innovative measures such as the development of independent certification programs.

Sustainable fishery management then is a complex socioeconomic and ecological undertaking, requiring a systemic, participatory, and adaptive approach.

Integrating Nature Conservation and Fisheries

There is an apparent dichotomy between the harvesting of fish and the conservation of the ecosystems that sustain them. At the Fourth World Fisheries Congress in 2004, Pauly (2004) asked, "*How can fisheries science and conservation biology achieve a reconciliation?*" then answering the question with "*By accepting each other's essentials: that fishing should remain a viable occupation; and that aquatic ecosystems and their biodiversity are allowed to persist.*" In reality, sustainable fish stocks depend on productive and resilient ecosystems. The key challenge then is in how to manage wetlands of various types, from coral reefs and shallow coastal waters to estuaries and flowing and standing freshwater systems, such that exploitation of fishery stocks does not undermine system integrity and with it the livelihoods of dependent communities.

A key element of integration is undertaking a participatory approach, such that fishery activities are not guided narrowly by assessments of stock yields of selected target species. Rather, the perspectives of multiple stakeholders need to be taken into account, consistent with the principles of the Convention on Biological Diversity's Ecosystem Approach (http://www.cbd.int/ecosystem/principles.shtml). At the very least, this should include such interests as:

- Wetland owners and managers
- Statutory organizations with management responsibilities or interests in selected outcomes
- Local authorities
- Both commercial and small-scale fishers
- Nature conservation interests
- Ecotourism interests
- Navigation, watersports, flood management, and other related interests identified on a context-specific basis
- Farming and other interests in catchments feeding into the wetland

Only when different interests in a wetland have a voice can constructive dialogue ensue to identify potential conflicts, potential synergies, and any necessary management measures to avoid the former and promote the latter.

An example of a synergistic measure is where a "no take" fishing zone can be identified, in which areas of the greatest value for nature conservation can also serve valuable fishery stock recruitment purposes. Zoning of watersports to minimize interference with wildlife, fishery activities, and other interests is an example of a management measure in which inherent conflicts are averted. In Europe, Marine Protected Areas are providing a means to establish areas of particular ecological and cultural importance from which some forms of fishing may be excluded, or else zoned, but which in turn have roles in the regeneration of commercial and recreational stocks. Other management measures may include agreements on limiting landings from the fishery, restrictions on destructive fishing methods, fishery closed seasons at vulnerable times, and other restrictions aimed at cooperation – be that informal or legally enforced – to ensure the viability of the fishery ecosystem.

As an example of a management measure based on fishery exploitation, Pacific salmon (a generic term covering a range of migratory species of fish in the family Salmonidae) are generally managed on the western American seaboard by determining how many spawning salmon, known as the "escapement," are required to ensure fishery regeneration to maximize the harvestable surplus (Hilborn 2005). A similar approach is taken in the UK though for basic conservation reasons, the Government including in its suite of indicators of sustainable development the status of Atlantic salmon (*Salmo salar*) as both a priority species (it is scheduled under the

EU Habitats Directive) and an indicator of ecosystem health. To effect this, England's Department of Environment, Food, and Rural Affairs (Defra) includes the "*Number of rivers in England with sustainable salmon stocks*" as one of the nine headline biodiversity indicators, with the objective of protecting and enhancing Atlantic salmon stocks to ensure sustainable exploitation by fisheries as well as conservation of genetic diversity (Defra 2013). The indicator relates to the number of rivers in which Atlantic salmon spawning levels have met or exceeded agreed Conservation Limit standards for sustaining the native salmon stock. The Conservation Limit is set at a stock size below which further reductions in spawning numbers are likely to result in significant reductions in the numbers of juvenile fish produced in the next generation, and hence returning adults from that year class. It also recognizes that, as each river has a genetically distinct Atlantic salmon stock, once the population of salmon is reduced below the number required to maintain a sustainable population, there is a significant risk of losing that genetic diversity. A target of 27 qualifying rivers, out of 40 rivers assessed annually, was set in 1999. Between 1997 and 2004, the number of rivers with sustainable salmon stocks varied between 13 and 25. Despite a significant biannual fluctuation pattern, more than 20 rivers have qualified between 2004 and 2008, and the target was exceeded in 2008 with 28 qualifying rivers identified. Despite this increase between 2004 and 2008, the number of qualifying rivers fell sharply in 2009 to 16, similar to numbers seen in 2001 and 2003. The overall trend in recent years is therefore tentatively upwards, taking account of the biannual fluctuation pattern. This is welcome, though no excuse for complacency, as 40 rivers meeting their Conservation Limit targets out of 40 surveyed is surely the only logical target if we seek a sustainable future in which critical aquatic and other environmental resources are not being systematically undermined by the cumulative pressures of contemporary lifestyles.

In terms of destructive fishing methods, dynamiting and poisoning for immediate gain remain common in Indian rivers, destroying potential brood fish stocks as well as a range of other species and associated livelihoods, though redirection of revenues from recreational angling and associated ecotourism to benefit local people is proving to be a lever for community self-policing to stop such practices (Everard and Kataria 2011). In coastal seas, destructive trawling is more problematic, degrading the sea bed and evading effective regulation let alone policing for compliance.

Conflicts Between Capture Fisheries and Aquaculture

As wild capture fisheries continue to decline globally in productivity, aquaculture is increasingly providing a greater proportion of fish for human uses. Including open seas resources, it is commonly agreed that two-thirds of oceanic fisheries are now being fished at or beyond their sustainable yield and, as a direct consequence, many are collapsing or have already done so (FAO 2000). It is also estimated that aquaculture will provide humanity with more fish than are harvested from the world's oceans by 2030 (FAO 2000). Furthermore, a greater proportion of feed for aquaculture, particularly for predatory species such as salmonids and also coastal prawn farming in tropical areas, derives from harvesting of smaller species (such as sand eels) and juvenile, otherwise unsalable fish derived from coastal seas. This practice not only threatens fish stocks but also the viability and breeding success of seabirds and other elements of marine and estuarine ecosystems. Understanding and managing all of these associated risks and pressures is therefore a pressing priority.

For example, displacement of fish nursery habitat by coastal fish and prawn farms in the tropics is having impacts on the recruitment of wild fish stocks in addition to compromising natural coastal defenses from storm surges and erosion. Equally, inland, displacement of small-scale local fishers from riparian areas by prawn farms is observed as a source of conflict in ► Chap. 147, "Lake Chilika: Sustainable Fisheries Management Case Study".

All of these pressures need to be managed within an encompassing participatory and adaptive wetland management plan if conflicts, competitive exploitation and net degradation of the wetland system, and with it the interests of all, are to be averted.

Wider Pressures on Fisheries

Fisheries are also vulnerable to a wider range of pressures on the environment. For example, coastal fishing communities in Bangladesh are vulnerable to flooding from sea-level rises (Sarwar 2005), and the loss of coral reefs around the world is a cause of major concern for the interconnected conservation of nature, fisheries, and livelihoods. These are just some amongst many manifestations of how climate change is threatening the integrity of wetlands, fisheries, and human livelihoods globally (FAO 2009).

Increasing siltation and eutrophication of wetland by unsympathetic farming and other practices in upstream catchments also threatens the vitality of many wetland ecosystem including their fisheries and associated livelihoods.

Challenges Remaining

The technical issues associated with wetland management for sustainable fisheries are far from straightforward. However, setting them in the context of nature conservation and other human use priorities in "real world", wetland systems raises formidable challenges. Sustainable management is therefore only possible through a systemic, participatory, and adaptive approach, bringing all interested stakeholders into dialogue and consensus around common management objectives. Long-lasting, successful examples are scare, but include the ► Chap. 147, "Lake Chilika: Sustainable Fisheries Management Case Study".

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Sustainable Fisheries Management Case Study (Africa)

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Randall Brummett

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Abstract

The inland fisheries of Africa are highly diverse, both in terms of species richness and endemism. There are over 3350 species of fish in Africa, approximately 20% of which are endemics within 92 "ecoregions". Most African inland fisheries are small-scale and seasonal, and most fishers are also small-scale farmers. While reported total capture of fish from African inland waters has been rising steadily, there has been a decline in the trophic level of the species taken, indicating biological over-fishing and a reduction in the value of the catch. These trends are reflected by the steadily increasing number of fishers and reduction of CPUE. Serious and concerted effort is needed to correct the downward trajectory of these natural resource use systems for both environmental and social reasons.

Keywords

CPUE · Overfishing · Trophic level

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Inland Fisheries of Africa

The inland fisheries of Africa are highly diverse, both in terms of species richness and endemism. There are over 3,350 species of fish in Africa (Paugy et al. 2011), approximately 20% of which are endemics within the 92 “ecoregions” that Thieme et al. (2005) use to refine the major African ichthyological provinces originally described by Roberts (1975; Fig. 1). The most important of these in terms of inland fisheries (97% of total reported catches) are the Nilo-Sudan, the Congo Basin, the Great Lakes of Eastern Africa, and the Zambezi River.

Inland Fisheries Across the African Continent

The vast extent of the Nilo-Sudan fauna reflects the extent of the humid Sahara that prevailed between about 10,000 and 5,000 BP (Beadle 1981). The main fisheries are in the Nile, Senegal, Niger and Volta Rivers, Lake Chad, and the large reservoir behind the Aswan Dam on the Nile, which together produced some 1.2 million tons in 2012 (FAO 2014), of which 35–50% are cichlids (Lévêque and Paugy 1999).

The topography of the Congo Basin is at least 600 million years old and the aquatic biodiversity within it represents remnants of an older fish fauna that once covered most of the continent. Although the fish biodiversity of the Congo is extraordinary, 686 reported species, the antiquity of the soils and the large extent of forest cover result in low overall productivity. Pervasive and persistent political

Fig. 1 Ichthyological provinces of Africa, based on Roberts (1975) as modified by Lévêque (1997) and redrawn according to new hydrological basin mapping published by FAO (2000). *1* = Maghreb, *2* = Nilo-Sudan, *3* = Upper Guinea, *4* = Lower Guinea, *5* = Congo Basin, *6* = Quanza, *7* = Zambezi, *8* = East Coast, *9* = Southern, *10* = Malagasy (Image credit: R Brummett © copyright remains with the author)

instability in the Congo Basin during the last part of the twentieth and early part of the twenty-first centuries has both constrained fisheries and made data collection impossible, but it can be reasonably assumed that the real, or at least potential, catch is much higher than the 284,000 T reported to FAO (2014).

The Great Lakes of East Africa (especially Victoria, Tanganyika, Malawi, Turkana, Albert, Kivu, and Edward) were created by volcanic and rifting activity during the Miocene. Total catch from these lakes was about 1 million tons in 2012 (FAO 2014). Although remnants of the ancient fauna persist, these lakes are now ecologically dominated by haplochromine cichlid species flocks that have rapidly evolved to fill most ecological niches. Extremely important to the overall fish catches in some of the Great Lakes are small pelagic clupeids of the genera *Engraulicypris* (Malawi), *Stolothrissa* (Tanganyika), and *Limnothrissa* (Kivu), and in Lake Victoria the small pelagic cyprinid *Rastrineobola argentea*. The introduction of Nile Perch (*Lates niloticus*) had serious negative consequences for the indigenous haplochromines of Lake Victoria, but has generated a profitable fishery of its own, comprising some 240,000 tons in 2012 (FAO 2014).

The Zambezi River system, dominated by the large lakes created by the Kariba and Cahora Basa dams, produced some 150,000 T of fish catches in 2012, of which *Limnothrissa* introduced from Lake Tanganyika accounted for about 15%. The *Limnothrissa* fishery is one of the few on the continent that can be considered at least semiindustrialized, with motorized vessels equipped with large winch-operated lift-nets fishing at night under bright lights that attract fish.

The Value and Characteristics of African Inland Fisheries

The inland fisheries of Africa generated some $6.3 billion in 2012 (FAO 2014) representing an estimated 0.33% of continental GDP, of which $4.7 billion was the direct value of the catch. An estimated 3.1 million fishers ply the inland waters of Africa, almost 7% of whom are female (de Graaf and Garibaldi 2014). The small scale and dispersed nature of inland fisheries globally probably means that the reported catches are significant under-estimates (Lévêque and Paugy 1999).

Most African inland fisheries are small-scale and seasonal, and most fishers are also small-scale farmers (Jul-Larsen et al. 2003). Seasonality is driven by the Intertropical Convergence Zone (ITCZ), which fluctuates north and south centered over the equator bringing rain that drives flooding onto the savannas to the north and west, and into forests in central Africa. In riverine fisheries, flooding stimulates fish spawning migrations that can be intercepted by the traditional gears that dominate African inland fisheries. In lakes and floodplains, fish concentrated by decreasing water levels over the course of the dry season are targeted and in parts of Western Africa represent important seasonal feasts (Paugy et al. 2011). Whedoes (fish-holes) are a traditional fishing system originating from floodplain fisheries in the Nilo-Sudan, particularly Benin and Togo. As waters receded at the end of the rainy season, fish are trapped in whedoes from whence they can be harvested as the need arises.

Few African inland fisheries are what could be called “industrialized” (e.g., using large, steel boats powered by inboard diesel motors and using winch drawn nets). Most fishers operate gill-nets, various traps, weirs, hook-lines, and/or hand-operated seines, either from shore or from small wooden boats either rowed or powered with outboard motors. Hand-woven basket traps manufactured from local materials come in a dazzling array of sizes and styles. Supplementing these gears are various types of fish attracting devices. Acadjas (brush parks) are indigenous to West African lakes and lagoons; bésolos are the equivalent in the Congo Basin. These are piles of woody material heaped up in shallow water where they decompose, creating habitat and food resources that attract fish, which can then be relatively easily captured by surrounding the pile with a net and removing the brush.

While the reported total capture of fish from African inland waters steadily rose from 1950 through 2012, there has been a worrisome decline in the trophic level of the species taken, indicating possible biological overfishing of higher value carnivores (Fig. 2) and certainly a reduction in the value of the catch.

These trends are reflected by the steadily increasing number of fishers, illustrated by data from some of the few fisheries for which the time-series information is available (Jul-Larsen et al. 2003; van Zweiten et al. 2011; LVFO 2013).

- Lake Kyoga (Uganda) – 3500 (1978) to 8400 (2008)
- Lake Mweru-Luapula River (Zambia, DR Congo) – 2300 (1965) to 12,000 (1997)
- Lake Victoria (Uganda, Kenya, Tanzania) – 129,305 (2000) to 205,249 (2012)
- Volta Lake (Ghana) – 18,358 (1970) to 71,861 (1998)

Together, the declining trophic level, value of the catch, and increasing numbers of fishers add up to reductions in the catch per unit of effort and the profitability of the fishery for individual fishers. A key driver of these declines is the “tragedy of the commons,” wherein fishers compete with each other for catch share with no regard for the status of the fish stock. Already, fishing communities are among the poorest

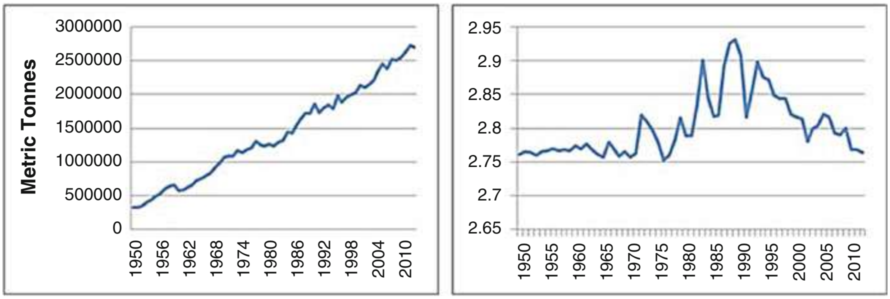

Fig. 2 Tonnes (*left*) and average trophic level (*right*) of inland fisheries catches reported to FAO (2014). (This file is licensed under the Creative Commons Attribution-Share Alike 3.0 Unported license)

of the poor. Serious and concerted effort is needed to correct the downward trajectory of these natural resource use systems for both environmental and social reasons.

Challenges

- Accurate stock assessment and catch data for small-scale fisheries
- Management systems that create incentives for wise management of fish stocks
- Alternative livelihoods that can help to reduce excess fishing effort

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Lake Chilika: Sustainable Fisheries Management Case Study

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Ritesh Kumar

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Abstract

The diverse and dynamic assemblage of fish, invertebrate, and crustacean species found in Lake Chilika, located in India's Odisha (formerly Orissa) State, provides the basis of rich fisheries which support livelihoods of over 0.2 million fishers and generate nearly 9% of Odisha's foreign revenue from marine products. Sustainable management of fisheries forms an important component of integrated approaches adopted for ensuring "wise use" of Chilika as a Ramsar Site (Wetland of International Importance under Ramsar Convention). Ecological restoration, management of the lake basin, inventory and assessment, and promoting community-managed fisheries are key measures that are being implemented to sustain fisheries and maintain the overall ecological health of Lake Chilika.

Keywords

Fish · Lake Chilika · India · Odisha · Orissa · Livelihoods · Marine products · Sustainable management · Fisheries · Ramsar Site · Ramsar Convention · Ecological restoration

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Introduction

The diverse and dynamic assemblage of fish, invertebrate, and crustacean species found in Lake Chilika (Fig. 1) provides the basis of rich fisheries which support livelihoods of over 0.2 million fishers and generate nearly 9% of Odisha (formerly Orissa) State's foreign revenue from marine products. Sustainable management of fisheries forms an important component of integrated approaches adopted for ensuring "wise use" of Chilika as a Ramsar Site (Wetland of International Importance under Ramsar Convention). Ecological restoration, management of the lake basin, inventory and assessment, and promoting community-managed fisheries are key measures that are being implemented to sustain fisheries and maintain the overall ecological health of Lake Chilika.

Ecological Restoration for Rejuvenating Fisheries and Biodiversity

Hydrological connectivity of Lake Chilika with the Bay of Bengal, tributaries of the River Mahanadi and a variety of streams of western catchments establishes a unique salinity gradient necessary for maintenance of this rich diversity of fish (Fig. 2) and other aquatic flora and fauna. Nearly 86% of the fish species found in the wetland are migratory and hence are dependent on riverine and marine habitats for a part of their life cycle. Chilika underwent a phase of rapid degradation between 1950 and 2000 owing to increasing sediment loads from the catchments and reduced connectivity with the sea. The lake fisheries underwent a major decline, invasive macrophytes proliferated and the wetland shrank in area and volume.

Introduction of shrimp culture added further pressure on lagoon ecology and ultimately led to significant disruption of traditional community governance of lake fisheries (Dujovny 2009). This formed the background for inclusion of Lake Chilika into the Montreux Record in 1993.

In 1991, the Government of Odisha instituted the Chilika Development Authority (CDA) under the aegis of the Department of Forest and Environment to undertake ecosystem restoration. In September 2000, a major hydrological intervention was carried out by opening a new mouth to the Bay of Bengal, which helped improve salinity levels enhancing fish landings, decreasing invasive species, and contributing to an overall improvement of lake water quality. The annual fish catch during 2001–2011 was 11,961.37 MT as compared to a decline from 8,861 MT to 1,734.9 MT between 1986 and 1999 (Mohapatra et al. 2007). The mouth connecting Lake Chilika with the Bay of Bengal is maintained through periodic dredging, with extensive monitoring to ensure that the connection to the sea is maintained. A 22.6 km lead channel has also been dredged in the northern sector to ensure sediments received from the Mahanadi River are flushed out from the wetland.

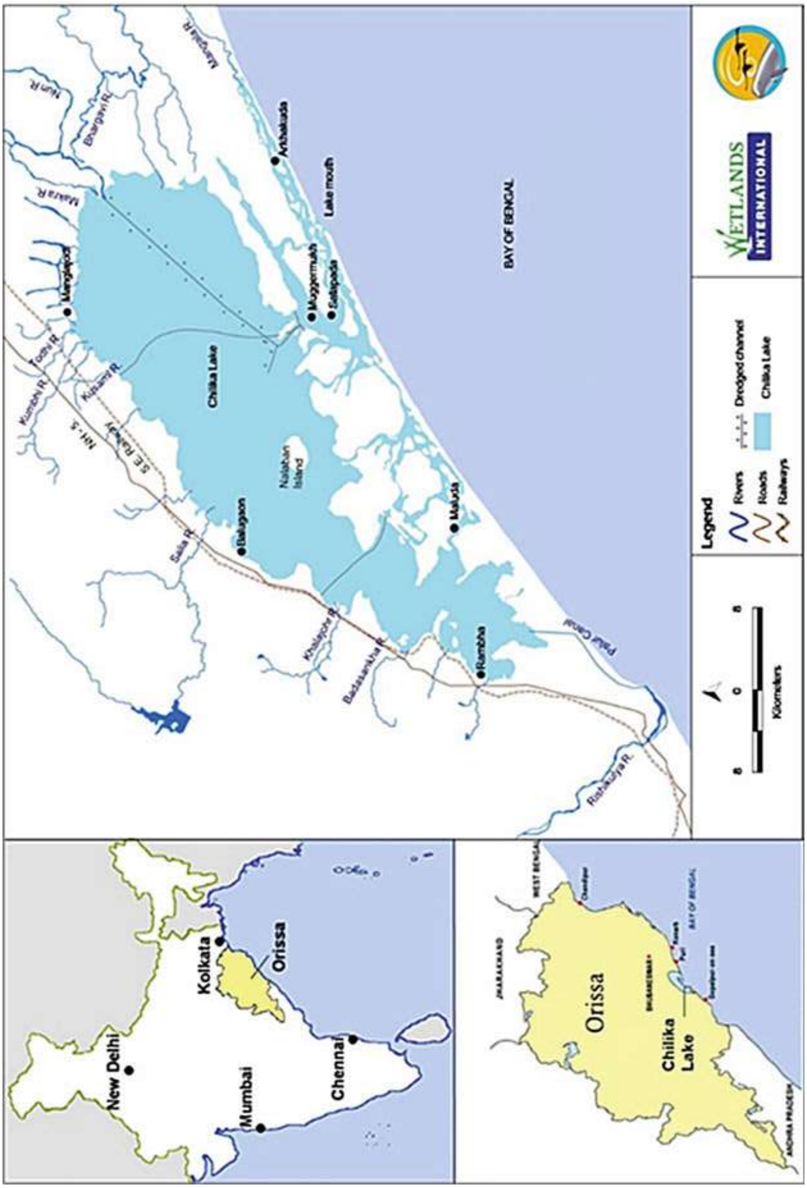

Fig. 1 Location of Lake Chilika

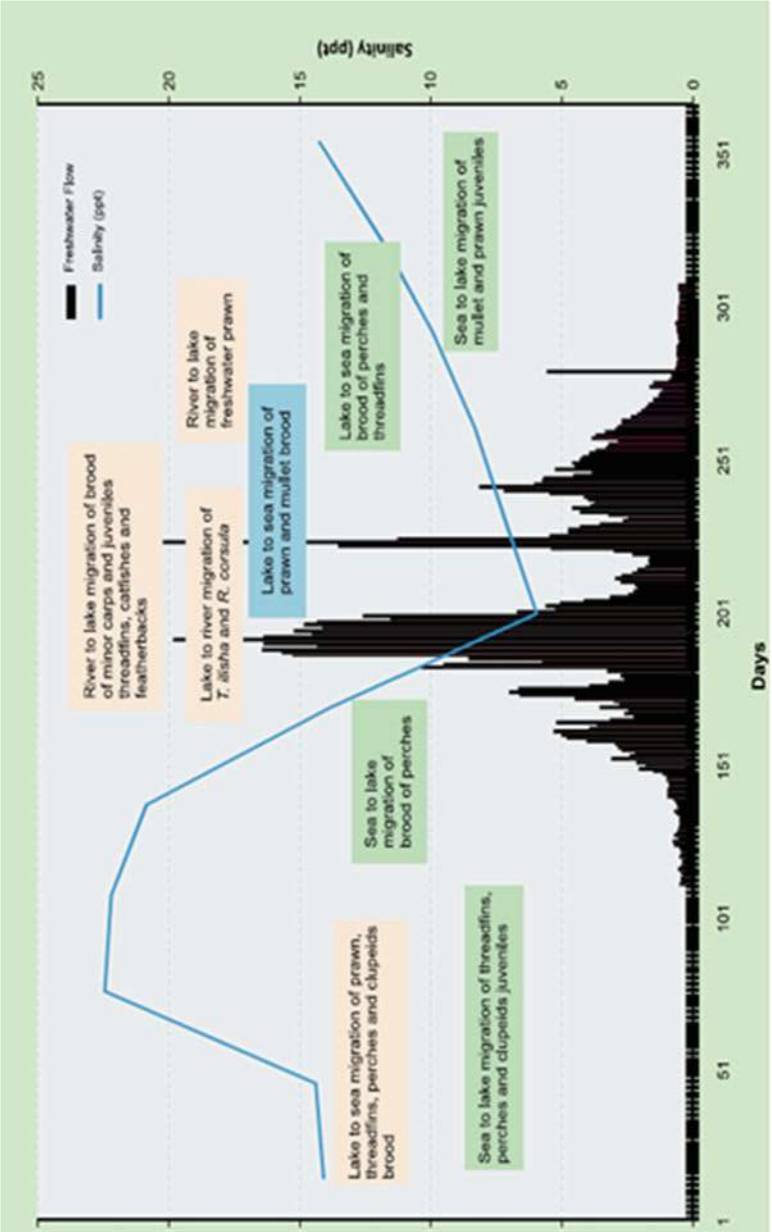

Fig. 2 Water regimes and fish migration in Lake Chilika

Lake Basin Management

Sustenance of fisheries in Lake Chilika is closely linked to ecological processes influenced by water, sediment, and nutrient exchange with the lake basin and the coastal zone. The CDA initiated a massive participatory watershed management project in the western catchments to restore their vegetative cover, improve soil moisture, and enhance resources for community livelihoods. Through dedicated capacity-building, conflict resolution, and trust-building with local stakeholders, the CDA enabled formulation of watershed management plans and also provided resources for their implementation. The overall forest cover in the basin, which had declined from 1,255.43 km^2 to 1,099.46 km^2 between 1972 and 1990, was observed to have increased to 1,267.27 km^2 in 2011 (Kumar and Pattnaik 2012).

Awareness Generation

The CDA launched intensive awareness generation campaigns concerning the values and functions of the wetland system, particularly among villages in and around Lake Chilika and with a focus on school children. A visitor center at Satapada serves as the hub of these activities. This center is open to local communities and visiting tourists throughout the year, providing information on the wetland through exhibits, dioramas, and models. Telescopes are placed in the Visitor’s Gallery for dolphin and waterbird-watching. An education kit for school children has also been developed.

Events as World Wetland Day and World Environment Day are celebrated each year within the local schools and community centers to promote awareness and seek participation from communities in wetland management. A newsletter “Chilika” is produced in both English and “Chilika Darpana” in Oriya, published by the CDA as an important means for communicating with stakeholders about programs and policies.

Inventory, Assessment, and Monitoring

Management of Lake Chilika is supported through a state-of-the-art ecological monitoring system using a network of 47 monitoring stations within the lake basin and 30 stations within the lake. Fisheries biodiversity and productivity is primarily monitored through intensive catch sampling, put in place since the hydrological intervention of 2000. A fish tagging study on the extent of seaward breeding migration of commercially important mullet species (*Mugil cephalus* and *Liza microlepis*) has been initiated with the active participation of local fishers.

A regional coastal processes study is also being implemented to develop baseline information on the sediment shelf using sediment transport and hydrodynamic models, the outputs of which will be used to develop a shoreline management plan. An assessment of response options and means to reduce climate change-related risks has also been initiated based on modeling of biophysical and social vulnerabilities, and coping and adapting capacities.

The Ecosystem Health Assessment program uses a report card system for communicating a set of water quality and biotic indices. These assessments provide the requisite information to adapt lake management in the face of dynamic drivers and pressures acting at multiple spatial, temporal, and political scales on the wetland.

Promoting Community-led Fisheries

While the hydrological intervention of 2000 was able to restore the necessary ecological conditions for rejuvenation of fisheries, the key to its sustenance lies in the design of institutional arrangements and mechanisms through which various stakeholders gain access and control over the resource base. Fishing in Chilika was historically managed by community institutions. For generations, Chilika fishers evolved a complex system of resource partitioning, wherein access to each fisher group was determined on the basis of the species they catch. The norms include setting spatial limits (what places are fished), temporal limits (seasonality), gear restrictions (what harvesting gear may be used), and physical limits (what sizes may be fished). However, weak capacities and economic nonviability led to the gradual decline of community fisheries institutions, with the fishers falling into debt traps at the hands of unscrupulous moneylenders (Nayak 2014). In 2010, the CDA, through technical collaboration with the Japan International Cooperation Agency (JICA), formulated a Fisheries Resource Management Plan (FRMP) based on over 3 years of resource survey assessing: the biology and ecology of eight commercially important high-value fish, prawn, and mud crab species; modeling of various conservation and management options; a wide range of stakeholder consultations; and ratification by an expert committee. The FRMP entails convergence in fisheries governance to ensure sustainable fish production through wise use of fisheries resources as well as to secure the livelihoods of fishers. The plan recommends a comanagement strategy, with the active participation of fishers.

In July 2010, the State Government of Odisha established a new Central Fishermen Cooperative Society known as the Chilika Fishermen Central Cooperative Society (CFCCS) Ltd as the apex agency for managing the Chilika fisheries. Availability of credit on equitable terms plays an important role in the economic viability of the PFCS. Under a pilot initiative, the CDA, through the Fisheries and Animal Resources Development Department, provides Rs. 7 lakh (Rs.700,000 or $US11,200 at an August 2014 conversion rate) as a revolving fund to PFCSs to revive the institution and ensure fair access to credit to its member fishers. Several PFCSs have managed to become economically viable and functional with this financial and capacity-building support.

Lack of appropriate storage facilities force the fishers to sell their catch to the middlemen who exploit their vulnerability by paying lower prices and manipulating weights. The CDA, through support from the Marine Products Export Development Authority (MPEDA), has launched an initiative to provide ice boxes to the fishers so that the catch can be maintained for a longer time, enabling fishers to choose their preferred point of sale. A 70 liter box costs Rs.2,200, of which 50% is subsidized by the MPEDA, 30% by the CDA, and the rest is borne by the fisher. This scheme has been very warmly received and, thus far, 1,000 boxes have already been distributed with fishers reporting at least a 30% increase in sale proceeds.

Integrated Management Planning

The fisheries values of Lake Chilika coexist with high biodiversity. The lake is a natural habitat for 224 species of waterbirds (including 97 intercontinental migrants) and regularly hosts over one million wintering migratory birds. It is also one of the two lagoons in the world that supports the Irrawaddy Dolphin (*Orcaella brevirostris*). Designation of Lake Chilika as a Ramsar Site commits the Government of India and Government of Odisha to take actions to ensure wise use of the wetland ecosystem.

Wise use is the longest-established example amongst intergovernmental processes of the implementation of what is now known as the Ecosystem Approach for conservation and sustainable development of natural resources, including wetlands. The wise use approach identifies the critical linkages that exist between people and the sustainable development of wetlands, and encourages community engagement and transparency in negotiating trade-offs and determining equitable outcomes for conservation. An integrated management planning framework for conservation and wise use of Lake Chilika has been formulated with extensive community consultation and is a reference point for the implementation of annual action plans (Kumar and Pattnaik 2012).

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Tonle Sap: Fisheries Management Case Study

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Gareth Johnstone and Mak Sithirith

Contents

Abstract

The Tonle Sap Lake in Cambodia is considered the "heart" of the Mekong River Basin, helping to pulse the flood waters in and out during the high and low flood seasons and saving large areas in the Mekong Region from flooding and drought. The inflow and outflow of the water between the Mekong River and the Tonle Sap Lake enriches catchment biodiversity and fisheries; the lake itself also an important location for biodiversity and fish. The Tonle Sap Great Lake is connected to the Mekong River through the 100 km long Tonle Sap River, close to 60% of its water originating from the Mekong with the rest coming from local tributaries and rainfall over the lake. The Great Lake covers an area of 250,000–300,000 ha in the dry season and 1.0–1.3 million ha in the wet season. The Tonle Sap Great Lake and floodplain is characterized by high fishery and agricultural productivity, with close to 300 fish species found in the Lake. The Tonle Sap fisheries account for about 60% of the national fish catch. Hydrological fluctuation and dispersed resources around the lake shape the ways in which people settle on the lake geographically, ecologically, and socially with a human population in the Tonle Sap Basin of more than four million.

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Keywords
Lake · Hydrology · Cambodia · Fishery · Mekong · Productivity

Introduction

The Tonle Sap Great Lake and floodplain is characterized by high fishery and agricultural productivity. The Great Lake covers an area of 250,000–300,000 ha in the dry season (Somony and Schmidt 2004) and 1.0–1.3 million ha in the wet season (Nikula 2004; CNMC and NEDECO 1998; Asian Development Bank (ADB) et al. 2003; MRC 2003). The Tonle Sap Lake is considered the "heart" of the Mekong River Basin, helping to pulse the flood waters in and out during the high and low flood seasons, and saving large areas in the Mekong Region from flooding and drought. The inflow and outflow of the water between the Mekong River and the Tonle Sap Lake enriches catchment biodiversity and fisheries, the lake itself also an important location for biodiversity and fish. The Tonle Sap Great Lake is connected to the Mekong River through the 100-km long Tonle Sap River, close to 60% of its water originating from the Mekong with the rest coming from local tributaries and rainfall over the lake (Baran et al. 2007).

The lake is home to close to 300 fish species (Wright et al. 2004) and its productivity is one of the highest in the world. The Tonle Sap fisheries account for about 60% of the national fish catch (Baran 2005) estimated at 390,000 tons annually (FiA 2009). In the wet season, the water from the Mekong River flows into the Tonle Sap Lake, the lake size increases 5–6 times, and the water level increases from 1.5 m to 9–10 m. In the dry season, the water from the Tonle Sap reverses its flow into the Mekong River, shrinking the lake's surface area and reducing its water level to 1.5–2 m.

The hydrological fluctuation and dispersed resources around the lake shape the way in which people settle on the lake geographically, ecologically, and socially. The human population of the Tonle Sap Basin is more than 4 million, with 1.5 million people dependent directly on the lake's resources for livelihoods based on fishing and farming practices (NIS 2008). A recent analysis shows that fishing dependency (a combination of fishermen density and poverty index) is the highest around the Tonle Sap Lake and Mekong River (Nasielski et al. 2013) (Fig. 1). The Tonle Sap Lake and floodplain also shape the way of life for several million Cambodians, who have adapted livelihoods to coincide with the rise and fall of the water between wet and dry seasons, and where fish resources from the lake and rice fields continue to make significant contributions to incomes, food and nutrition security (Joffre 2012; Navy et al. 2006).

Governance of the Tonle Sap Lake and Floodplain

Governance of the Tonle Sap is determined by an array of international, national, and local actors, institutions, laws, and regulations. The governance system has emerged over time. Prior to 2000, the management of fisheries around the Tonle Sap was

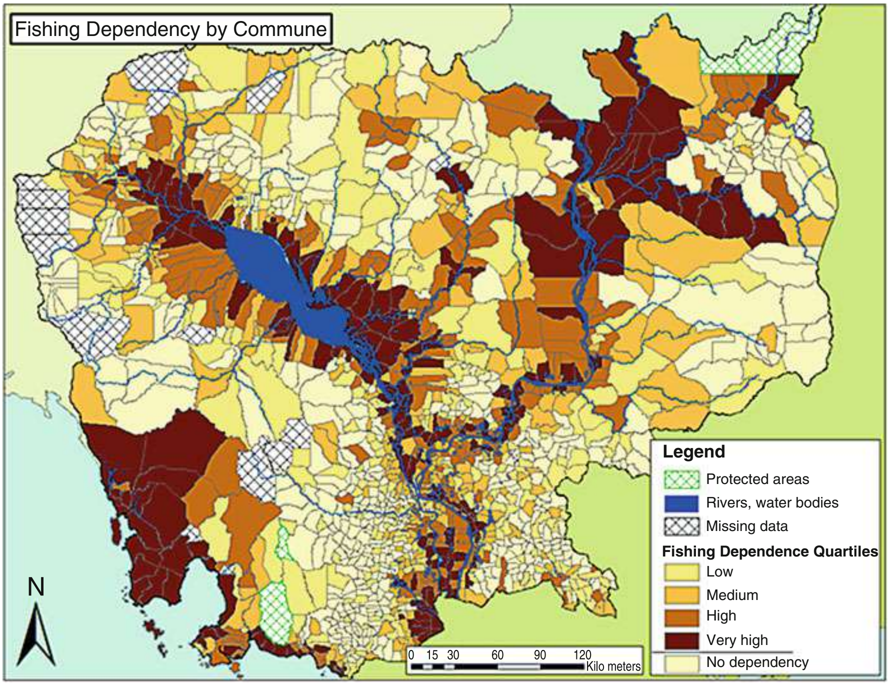

Fig. 1 Fishing dependency in Cambodia (Image credit: G Johnstone and Mr Nalseilski © copyright remains with the authors)

equated with management of the Tonle Sap Lake. The Fisheries Administration (FiA) under the Ministry of Agriculture, Forestry, and Fisheries (MAFF) was the key actor in managing fisheries, and the whole lake was managed based on private leasehold fishing lots, open access and conservation areas. The MAFF also mandates agriculture, reflecting the importance the lake and floodplain to rice production.

Internationally, the Tonle Sap is also governed for its biodiversity value, recognized as a part of Cambodia's international commitment to conservation through ratification of the Convention on Biological Diversity in 1995, the UNESCO agreement on Man and Biosphere Reserves (1997), and the Ramsar Convention (1999). The Tonle Sap Biosphere Reserve was formally designated by Royal Decree in 2001 and is overseen by the Tonle Sap Biosphere Reserves (TSBR) Secretariat. The designation of the Tonle Sap Lake as a Biosphere Reserves in 1997, with the establishment of the TSBR Secretariat within the Cambodian National Mekong Committee (CNMC) in 2001, elevated the profile of a wider range of criteria for environmental governance of the lake, extending beyond a narrower former fisheries management focus. This led to the zoning of the Lake into three zones – (i) the transitional zone; (ii) buffer zone; and (iii) core zone – for biodiversity conservation and to a new structure of governance for biodiversity conservation (see Fig. 2).

Regionally, the Tonle Sap plays an important role in flood management in the Mekong River basin. The four governments in the Lower Mekong River Basin

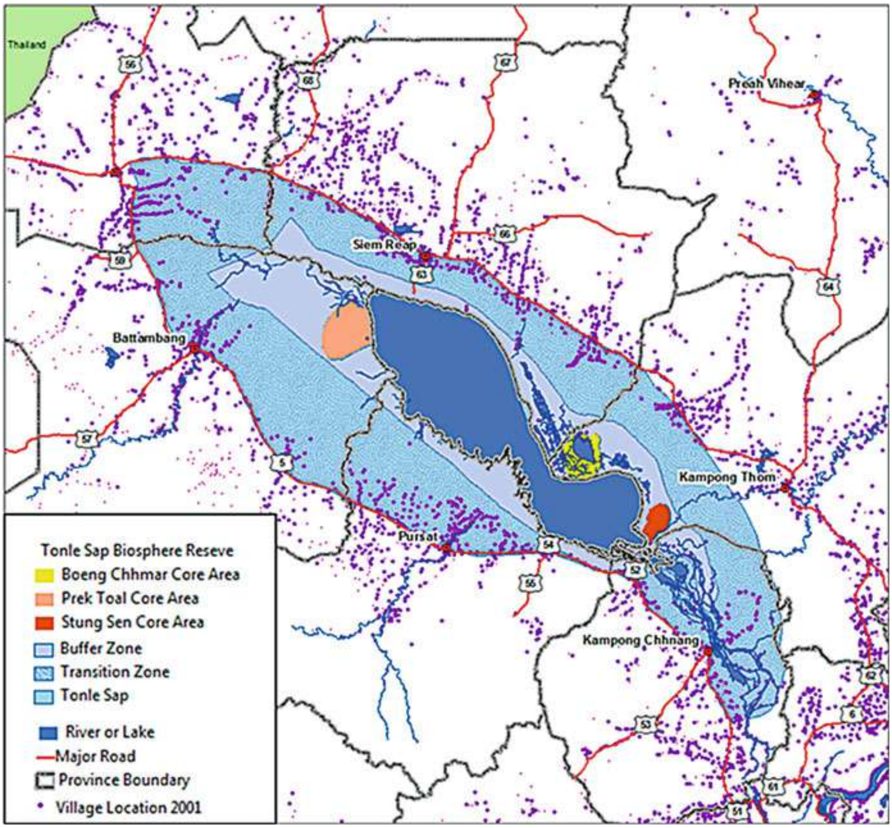

Fig. 2 Tonle Sap situation map (Map created with the GIS version of the 2006 Atlas of Cambodia published by Save Cambodia's Wildlife 2006)

(Laos, Thailand, Cambodia, and Vietnam), which formed the Mekong River Commission (MRC) in 1995, have agreed to protect the "flood pulse" function of the Tone Sap (Sithirith 2011). The TSBR, established within the Cambodian National Mekong Committee (CNMC), coordinates ten Ministries and is responsible overall for biodiversity conservation. In 2007, The Tonle Sap Authority (TSA) was established by a Royal Decree, aiming to improve the coordination, conservation, and development in the Tonle Sap. The TSA is chaired by the Ministry of Water Resources and Metrology (MOWRAM) in order to coordinate government and nongovernment agency activities affecting Tonle Sap.

Nationally, the contribution of fisheries and agriculture to the economy is significant and estimated to be 33% of GDP, with fisheries and fish trade alone accounting for 12% of GDP (FAO 2011). Without water, agriculture and fisheries cannot flourish, and water management has also become of equal importance to fisheries, agriculture, and conservation the management of Tonle Sap. The system of governance involves the deconcentrated and decentralized federated government system that includes Village Chiefs, Commune Councils, District and Provincial Governments. Technical Working Groups (TWGs) have been established at national level, aiming to improve agricultural productivity, diversification, and fisheries. Within MAFF, there are two TWGs respectively addressing Agriculture and Water and also Fisheries. At the provincial level, the Department of Agriculture is responsible for implementing agricultural programs. Under separate arrangements, Provincial Fishery Cantonments are responsible for implementing fishery programs.

Fisheries Management

The system of fishery management in the Tonle Sap has been dominated by the use of commercial fishing lots, a form of private leasehold that has provided income nationally from fisheries for more than 100 years. Continued conflicts between competing lot owners and communities resulting in over-use, as well as issues relating to access and management of fish resources has contributed to the reform of the sector.

In 2000, the Government reduced by 56% the area of commercial fishing lots. The decommissioned areas were transferred to small-scale local fishers, progressively organized into Community Fisheries (CFis) with the help of the Fisheries Administration (Kurien et al. 2006). All remaining fishing lots were abolished in 2012 (with the exception of the “Dai” fishery) with further transfers to Community Fisheries or creation of fishery conservation zones. In total, more than 1 million hectares of private concessions were transferred to CFis, constituting a shift in management from private ownership to community-based management (Ratner 2006).

In 2010, the FiA adopted the Strategic Planning Framework for Fisheries (2010–2019), detailing the main objectives and targets for fisheries development. The targets for capture fisheries includes: 470 community fisheries officially registered and operating effectively by the end of 2019; and at least 80% of Great Lake fish sanctuaries improved through boundary demarcation, protection, and public awareness by the end of 2019.

As well as the measures taken by the FiA to support of decentralization of management responsibility to CFis and for conservation, the government has also delegated authority for management of fisheries to other institutions. Notably, the Tonle Sap Inter-Ministerial Committee has been formed to combat illegal fishing and, in partnership with an Inter-Provincial Committee and the TSA, has conducted a series of campaigns to crack down on illegal fishing (TSA 2013). However, community CFis have not been involved in resolving fishing conflicts. Determining access to fisheries and decisions about compliance remain centralized and continue to create tensions with fishing communities.

Management Challenges and Opportunities

Coordination is the main challenge facing governance of the Tonle Sap Lake and floodplains. The establishment of the Tonle Sap Authority has gone some way to coordinate between institutions, but further revisions are required. A case in point is the multi-institutional structures and regulations relating to protecting the biodiversity of the Tonle Sap involving international, national, and local level bodies.

The cancellation of fishing lots was a rapid process that raised questions about how open access areas would be managed, a challenge reinforced by increasing evidence of uncontrolled use and a "tragedy of the commons" for fisheries and habitats. The Fishery Law limits fishing practices in the CFis to small-scale and subsistence fishing technologies. This constrains the capacity of fishing communities to make an income, often leading to noncompliance and increased illegal fishing practices. This is further compounded by inconsistencies in knowledge about and enforcement of relevant laws by Inter-Ministerial and Inter-Provincial Committees that vary from province to province (Johnstone et al. 2013).

In order to support CFis, the government has formulated policies and a regulatory framework, set up institutional mechanisms, promoted capacity building, and has contributed to stock enhancement initiatives operated by CFis (for example the establishment of conservation zones). CFis have achieved a lot but, according to SCW (2014), they function best when supported by external organizations or donors; without such support, they are weak. This is mainly due to the limited capacity of CFis members, the lack of sources of income for CFis to carry out their activities, and the lack of personal incentives for CFi members to participate in management activities.

Conclusion

Governance of the Tonle Sap has changed from a narrow focus on fishery management, focusing on privatized rights for commercial use of fishery resources, to community-based fishery management and conservation of biodiversity that involves many organizations, institutions, and communities. While this is a new approach for Cambodia, a primary concern is how to ensure collaboration and coordination under the complex institutional and legal arrangement to ensure appropriate use, access, and management of the fishery and supporting natural resources.

Information on the overall governance framework for the Tonle Sap is available at national and international levels. However, a more sustainable approach to cancellation of fishing lots and the zoning of the lake and floodplains requires more data and knowledge about the consequences on management of the lake and floodplain and, in particular, how local governance systems at village and community levels function.

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Recreational Fishery Case Study (UK) 149

Mark Everard

Contents

Abstract

The value of recreational angling is substantial. In the UK, diverse branches of the sport – principally broken down into freshwater coarse fishing, game fishing, and sea fishing – contribute significant economic and social value as well as bolstering of the case for protection of fish stocks and wetland health. The contribution of angling to the economy, as a form of informal monitoring, as a contribution to activity and health, and to social cohesion and inclusion has to date been substantially under-represented. So too has been the value of fit and functional wetland ecosystems sustaining populations of fish of recreational interest. Recreational angling interests also provide a political voice to campaign for protection of fish stocks and wetland environments. Through multiple dimensions, recreational angling makes a substantial contribution to human well-being and strengthens the case for better stewardship of diverse wetland ecosystems, both in the UK and globally.

Keywords

Recreation · Cultural service · Angling · Game angling · sea angling · Course angling · Indicator · Conservation · Reinvestment

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Introduction

The value of recreational angling is substantial. In the UK, diverse branches of the sport – principally broken down into freshwater coarse fishing, game fishing, and sea fishing – contribute significant economic and social value as well as bolstering of the case for protection of fish stocks and wetland health.

The Value of Recreational Sea Angling in England

A 2012 government study in England highlighted that recreational sea angling in England supported over £2 billion (pounds sterling) in total spend, supporting over 23,000 jobs (Defra 2012). Furthermore, the Defra (2012) study estimated that there are 890,000 recreational sea anglers in England, representing approximately 2% of the country's total adult population, undertaking nearly four million days of sea angling. The vast majority of English sea angling took place from the shore, followed by private boats and then charter boats. Other principal motivations for going sea angling include being active, being outdoors, and also relaxing, though many other social, environmental, and health benefits accrue to individuals and communities from sea angling.

The Value of Recreational Game Fisheries to the UK

The direct economic benefits stemming from Britain's recreational game fisheries have been reasonably well studied and understood, and can be locally substantial. Formerly, the value of fish may have been expressed occasionally through the direct and generally detrimental exploitation of these natural resources. However, understanding of the value of game fisheries, as for other resources, is now expanding to recognition that intact and functional ecosystems providing the series of linked habitats that sustain the life cycles of game fishes, and particularly migratory game species such as Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) that have marine adult life stages, confer many dimensions of human well-being. Game fisheries thus not only have direct exploitative values but also serve as indicators and conservation goals for the vitality of connected wetland systems. The multiple values of these connected systems have historically been almost complete taken for granted yet which, if lost, would have potentially serious negative consequences for those relying upon or enjoying them.

Fisheries supporting Atlantic salmon, brown and sea trout, Arctic charr (*Salvelinus alpinus*) and grayling (*Thymallus thymallus*) provide food, though most recreational angling for wild fish is now undertaken in the UK on a catch-and-release basis to preserve stocks, but also associated livelihoods which may be of considerable economic importance for local communities. Everard and Knight

(2013) observe that, if one takes a commercially caught wild salmon of 3 kg in weight and assigns a generous price of £15 per kilogram to the commercial fisherman, then the netted fish is worth £45 at point of capture. Also, of course, the fish has to die to capture this financial return, and so is lost to the stock and the spawning escapement. However, if this same fish is caught by rod and line in the UK, it has at least a 60% chance of being returned alive to the water – and to spawn – while contributing anything from £500 to £2,000 to local economies, depending on the region in which it was caught.

Further economic value accrues though the investment of fishery owners into river systems and fish conservation projects. A survey in 2005 on Hampshire's Rivers Test and Itchen showed that anglers spent £3.25 million to fish that year, of which £3 million was reinvested into management of those rivers by fishery owners supporting 120 full- and part-time jobs in the process (Beville 2005). At least some of the other £0.25 million was spent in habitat rehabilitation projects by individual proprietors. As an overall investment in the aquatic environment, these impressive values are not exceeded by any other sector of society. And this does not include the money spent by those anglers in local pubs, hotels, bed-and-breakfasts, and shops, which added greatly to those economies in areas where other employment was scarce.

The Scottish Executive's 2001 Green Paper, *Scotland's Freshwater Fish and Fisheries: Securing Their Future*, highlighted the lack of useful data quantifying the economic position of freshwater angling. This in turn led to the commissioning of a study of the impact of angler expenditure on output, income, and employment on both a Scotland-wide and a regional basis. The outcome of this was the 2004 research report *The Economic Impact of Game and Coarse Angling in Scotland* that estimated that freshwater fishing was worth £113 million annually to the Scottish economy, with salmon and sea trout anglers accounting for over 65% (£73 million) of this total (Scottish Executive 2004). In addition, freshwater angling produced over £100 million worth of annual output for the Scottish economy, generating nearly £50 million in wages and self-employment income to Scottish households and supporting around 2,800 jobs.

Placing Values on Wetlands Supporting Recreational Angling

In England, two studies published by the Environment Agency in 2010 exploring the wide range of economic values resulting respectively from sea trout habitat restoration on the River Glaven in North Norfolk (Everard 2010) and the installation of a "buffer zone" to protect a formerly severely cattle-trampled field margin on the upper Bristol Avon in North Wiltshire (Everard and Jevons 2010). Both revealed significant benefits to society and the wider economy. Though ostensibly focused on fish and angling interests, the River Glaven study recorded that angling benefits from the sea trout restoration project accounted for less than 1% of the total benefit of the project which returned a substantial 325:1 benefit-to-cost ratio for the modest investment, with many benefits accruing from building social relations, stimulating ecotourism and regulating the environment, including a significant contribution to flood risk in the catchment. Likewise, the upper Bristol Avon buffer zone, though promoted by an angling club in collaboration with fishery officers of the Environment Agency, saw angling interests account for only 9.6% of the benefits of the scheme (still returning a favorable benefit-to-cost ratio of 3:1 to fishery interests) with the remaining benefits (which including angling benefits returned a benefit-to-cost ratio of 31:1) accruing to wider society, including improved amenity, social relations, costs averted from nature conservation priorities, and erosion regulation.

Further studies exploring surrogate markets to begin to reflect the many broader, nonmarketed services provided by the natural environment include the report *Revealing the Value of the Natural Environment in England* prepared for the UK government in 2004 (GHK Consulting Ltd 2004). This report assessed the annual value of the tourist trade attracted by a high quality natural environment in the UK as in the order of £5 billion in 2003, supporting the equivalent of 92,000 full-time equivalent jobs. The study and its resultant figures were not specific to wetland, but do at least give a sense of the magnitude of often overlooked values provided by ecosystems. However, even these figures are far from adequate to account for the less tangible but no less important spiritual, regulatory, sense-of-place, and other values of such treasured landscapes. The maintenance of wetlands in a condition suitable to support recreational fisheries supports many of these ecosystem services in addition to their direct recreational values.

Sometimes, the water requirements necessary to support ecosystems, such as ensuring minimum flow requirements to support different life stages of salmon, trout or other fish and organisms of conservation importance, are perceived as competing with the needs and demands of people for water. So too, the costs of conservation measures to retain environmental quality and diversity suitable to sustain the vitality of populations of species such as fishes have often been framed as a constraint of legitimate economic progress. However, it is a fallacy to view ecosystems in this blinkered way as representing a net cost or constraint upon human economic advancement. Wetlands and other ecosystems and their constituent biodiversity, including charismatic fish fauna, really matter both for their inherent worth and more tangibly in the many ways that they support the various dimensions of human well-being, from sustaining our basic biophysical needs through to providing economic resources and contributing to many less tangible aspects of our quality of life, achievement of potential, and future security.

From this post-industrial, ecosystem-informed view of the world, what have stocks of fishes of recreational interest ever done for us? The answer to this question is emphatic. The reality, through consideration of the multiple ecosystem services provided by the fish themselves and the ecosystems of which they are part, is that they confer significant and often quantifiable benefits to all people, ranging from reliable yields of fresh water, valued landscapes, purification of waterborne wastes, and the natural fertilization and formation of soils.

Conclusions

The contribution of angling to the economy, as a form of informal monitoring, as a contribution to activity and health, and to social cohesion and inclusion has to date been substantially under-represented. So too has been the value of fit and functional wetland ecosystems sustaining populations of fish of recreational interest.

Recreational angling interests also find a political voice through organizations such as the Angling Trust (www.anglingtrust.net) which campaigns for the protection of fish stocks and wetland environments.

Through multiple dimensions, recreational angling makes a substantial contribution to human well-being and strengthens the case for better stewardship of diverse wetland ecosystems, both in the UK and globally.

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Products from Wetlands: Overview

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Seb Buckton

Contents

Abstract

Wetland Products are those things produced in wetlands that can be used by people as food, fuel, building materials, fibre, and fodder for livestock. They equate to Provisioning Ecosystem Services derived from Wetlands. They may be used in their raw state or be processed in some way and can be exploited at a range of scales and intensities. Wetland products have been an important resource for human development, and are important components of livelihoods, particularly in developing countries. Economically important wetland products include rice and fish, whilst others may provide vital support for small-scale local livelihoods. Wetland products are of great and increasing importance in wetland management considerations, providing both opportunities for and constraints to conservation management.

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Wetland product · Provisioning ecosystem services · Livelihoods · Development · Wetland management · Conservation · Economics

Introduction

In the broadest sense, wetland products are materials produced by a wetland that are used by people. The term correlates largely with Provisioning Ecosystem Services derived from wetlands (Millennium Ecosystem Assessment 2005). These are subdivided into food, fresh water, fiber and fuel, biochemical products, and genetic materials.

A narrower definition would describe wetland products as those things produced in wetland environments that can be used by people as food, fuel, building materials, fiber, and fodder for livestock. They may be used in their raw state or be processed in some way. Generally, fresh water provided by a wetland for drinking and irrigation is not considered a wetland product.

Historically, wetland products have been an important resource for human development and have contributed to the development of human settlements around the world. More recently, widespread direct use of wetland products as components of local livelihoods has been largely confined to developing countries.

Wetland products can be exploited at a range of scales and intensities, from intensive large-scale cultivation of fish or rice, to extensive, small-scale subsistence harvesting of wetland plants and animals. The economic significance varies accordingly, from providing significant national income to providing vital support for small-scale local livelihoods of the poorest in society.

Wetland products are of great and increasing importance in wetland management considerations, providing both opportunities for and constraints to conservation management.

Types of Wetland Products

Wetland products can be categorized according to the uses to which they are put. They can be used for human food (both plant and animal), as fuel, building materials, for fiber production, and as fodder for livestock. Although generally biological in origin, wetland products can also include mineral products from wetlands (e.g. mud, clay, or sand used as a building material).

Food

Wetland animals used as human food include fish, frogs, snails, and freshwater and marine shellfish, the production of which may be managed or unmanaged. Plants include both cultivated plants grown in wetlands, such as rice, and naturally occurring plants that may be foraged for subsistence use.

Sometimes, naturally occurring plants are semicultivated within seminatural wetlands. In China and other parts of Asia, beds of native lotus and water chestnut are cultivated for harvesting as a crop to sell. Plant material may also be processed to produce other products such as edible starch, sugar, and vegetable oils from various palms. Honey is harvested from bees in wetlands, particularly mangrove and *Melaleuca* forests.

Wetland plants also provide genetic material that can be used to develop strains of plants for use as food. Rice *Oryza sativa japonica* first originated from a population of wild rice *O. rufipogon* from the Pearl River area of southern China and was crossed with local wild rices to create *Oryza sativa indica* (Xuehui Huang et al. 2012). Populations of wild rice are still providing genetic material to improve rice cultivars (Barona-Edra 2012)

Other Consumable Wetland Products

Wetland provide habitat for a wide variety of medicinal products, including animals such as the medicinal leech, and many plants (Ramsar 2008). Many of these are used in ethnomedicinal preparations, but the origins of some modern pharmaceutical preparations come from wetland plants, most notably aspirin, based on salicylic acid, the precursor molecule that originates from willow *Salix* trees (Mahdi et al. 2006).

Salt is another consumable wetland product that is produced on a large-scale commercially. Sea water is allowed to evaporate in salt pans, leaving deposits that are subsequently collected and processed into salt.

Fuel

Wetlands provide fuelwood in the form of timber, which is widely collected in the developing world for domestic purposes. Peat deposits, formed from decayed wetland vegetation over thousands of years, are harvested to be used as fuel in certain parts of the world. There is also increasing interest in using plants to produce fuel. Sugar cane, soy, and oil palm are all grown to produce biofuels and thrive in wetland habitats. Willow and poplar are grown on a short-rotation coppice to produce biomass for use in heat and power, with potential for both domestic and industrial scale production.

Building Materials

Wetlands can provide materials used for structural work (e.g. timber) and also as roofing or plastering material. Wetland grasses are used as thatch, whilst muds and clays from wetlands can be used as walling materials.

Fiber

Rushes and grasses are used for weaving a range of products, such as mats, eating utensils (plates and bowls), hats, and basketry.

Fodder

A wide variety of wetland plants are harvested and used as fodder for livestock. Fodder from wetland plants is particularly important in dry areas where there is insufficient forage outside of wetlands to support livestock (Westlake et al. 2009).

Economic Importance

Products from wetlands are of significant economic importance to people all over the world. Fish is a significant source of protein for three billion people, whilst rice provides about a fifth of global calorie consumption (Ramsar undated). There is scope for developing other wetland products as sustainable sources of materials or nutrition. For example, Sago palm, which requires seasonal inundation to flourish, has the potential for development as a commercial crop, chiefly for high-quality starch for use as an industrial feedstock (Flach 1997).

Many of the other industrial scale uses outlined above have great economic significance. However, wetland products are also vital components of livelihoods of those living near wetlands, particularly amongst rural communities in developing countries (Silvius et al. 2000). Wetland products can provide an income by being traded or can be components of wider sustainable livelihood strategies through providing food or materials used in the household.

Relationship with Wetland Management

Wetland products are a valuable provisioning ecosystem service that people benefit from at a variety of scales. Exploitation of wetland products can impact on wetland habitats and conservation depending on the scale of exploitation. There are concerns over the direct impact of unsustainable extraction on specific wetland products such as fish or timber. Exploitation of some wetland products can also have indirect impacts on wetland habitats and their capacity to deliver other ecosystem services. Salt extraction, biofuel production, rice cultivation, and peat extraction can all compromise ecological character of wetlands.

Small-scale exploitation for subsistence or local commercial activity are more likely to be carried out sustainably and have less impact on wetland ecosystems. However, the importance of wetland products can both be a barrier and incentive to wetland conservation. Where a product is of high economic value, exploitation of it can become unsustainable, leading to deterioration in the ecological character of the wetland. In other circumstances, wetland products can highlight the importance of wetland conservation in supporting livelihoods, especially of the poorest in the community. As such, wetland products can be important management tools in wetland conservation.

Wetland Products as Management Tools

Sustainable management of wetlands to allow benefits to accrue to local people through controlled exploitation of wetland products is often a key management objective. Win-win situations can occur where exploitation of a wetland product provides both benefits to local people and benefits to biodiversity. For instance, at Koshi Tappu in Nepal, local people developed the use of water hyacinth, an invasive nonnative aquatic weed, to produce compost for adjacent farmland, reducing the need to use chemical-based fertilizers that are pollutants of the adjacent Ramsar site. Another nonnative weed that grows in damp ditches (Ipomoea) is used to make smokeless charcoal briquettes, reducing the need to use animal dung as fuel. As a result, more animal dung is available for fertilizing farmland, and human health is improved by reducing the smoke in houses from cooking using animal dung (Buckton et al. 2010).

Wetland products can also be used to promote wise use of wetlands by providing economic incentives to local people. The Wildlife Conservation Society (WCS) Cambodia Programme and Sansom Mlup Prey have developed the concept of "Ibis Rice." The rice is marketed as Certified Wildlife Friendly™ and organic, contributing to protecting critically endangered bird species (including the giant ibis *Pseudibis gigantea*) whilst also promoting improved incomes for farmers that engage in conservation. See http://www.youtube.com/watch?v=QLIsiTp5hZI and http://collaborations.wcs.org/smp/About/WildlifeFriendlyIbisRiceProject.aspx

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Medicinal Plants in Wetlands

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Danna Leaman

Contents

Abstract

All human societies use plants as medicines. Many important medicines are derived from plants with a long history of traditional use, and new medicinal applications for plants as medicines continue to be discovered. Plants with known medicinal uses and properties occur in most habitats to which people have access. Most medicinal plants are wild-harvested rather than cultivated. Studies have shown some habitats to be richer in medicinal species than others. In terrestrial habitats, for example, secondary forests are richer in the plant secondary compounds useful in medicine than are primary forests. While animals, fungi, bacteria, and algae associated with wetlands are proving to be successful targets for discovery of new natural products with medicinal properties, some flowering plant taxa typical of wetlands have been valued as sources of medicine since ancient times.

Keywords

Medicines · Medicinal · Conservation · Sustainable use · Health · Livelihood

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Introduction

All human societies use plants as medicines. Many important medicines are derived from plants with a long history of traditional use, and new medicinal applications for plants as medicines continue to be discovered (Miller 2011). Plants with known medicinal uses and properties occur in most habitats to which people have access. Most medicinal plants are wild-harvested rather than cultivated (Schippmann et al. 2006).

Studies have shown some habitats to be richer in medicinal species than others. In terrestrial habitats, for example, secondary forests are richer in the plant secondary compounds useful in medicine than are primary forests (e.g., Leaman 2006). While animals, fungi, bacteria, and algae associated with wetlands are proving to be successful targets for discovery of new natural products with medicinal properties, some flowering plant taxa typical of wetlands have been valued as sources of medicine since ancient times (Horwitz et al. 2012). These include the sundews (*Drosera* / *Droseraceae*), lotus (*Nelumbo*/*Nelumbonaceae*), reeds and grasses (*Phragmites*/*Poaceae*), and bulrushes (*Typha*/*Typhaceae*).

Contribution of Wetlands to Plant-Sourced Medicines

A comprehensive global survey of the distribution of medicinally used plants in various habitats does not yet exist, without which it is not possible to evaluate the relative importance of wetlands as a source of medicinal plants. Therefore, most descriptions of medicinal plants in wetlands rely on well-known taxa (e.g., Horwitz et al. 2012). However, local and regional surveys of biodiversity that include socioeconomically important plants can provide insights into the importance of wetlands as sources of medicinal plants, their contribution to health and livelihood benefits, and their conservation challenges.

For example, a survey of freshwater biodiversity in Northern Africa included 518 plant species, of which 43% have known social and economic value (Juffe-Bignoli et al. 2012). Among the documented uses (structural material, paper, other chemicals, household, fuel, aromatic, horticulture, aquarium, handicrafts, animal feed, ornamental, food, medicinal), the largest number of species have medicinal use. The sedge (*Cyperaceae*), mint (*Lamiaceae*), knotweed (*Polygonaceae*), and daisy (*Asteraceae*) families provided the largest numbers of medicinal plants from the wetland habitats included in the survey. These are large families with cosmopolitan distributions, although the strong representation of species in the mint family may reflect its regional center of diversity in the Mediterranean–Central Asia (Mabberley 1997).

The representation of plant taxa in the medicinal flora of freshwater habitats likely varies with the taxonomic composition of the regional flora. For example, in the Apo Kayan communities at the headwaters of the Kayan River, East Kalimantan (Indonesian Borneo), the ginger family (*Zingiberaceae*) provides the largest number of medicinal plant species associated with freshwater habitats, followed by the grass (*Poaceae*), aroid (*Araceae*), and mint (*Lamiaceae*) families (Leaman 2006), reflecting the relatively large representation of these families in the Southeast Asian/ Malesian flora (Mabberley 1997).

High numbers of plant species with medicinal use are common in developing countries, where traditional systems of medicine are more accessible than modern alternative treatments. However, the use of herbal remedies is increasing worldwide, as governments and health practitioners accept and integrate traditional medicine and other complementary and alternative practices (World Health Organization 2013), based on a growing scientific capacity to evaluate the medicinal properties, safety, and efficacy of traditional and new applications (e.g., Betz and Hardy 2014). The economic value of trade in wetland plant species is not currently possible to assess as a distinct component of the global sales of herbal products, which is itself difficult to evaluate even on a national scale, since the substantial value of local use and trade is not captured in most market data. In China alone, the market value of Chinese *Materia Medica* was estimated to be US$83.1 billion in 2013 (World Health Organization 2013).

The great majority of freshwater medicinal plants in the northern Africa region is collected from the wild (Juffe-Bignoli et al. 2012), reflecting the global situation for wild-sourcing of medicinal plants (Schippmann et al. 2006). In northern Africa, 20% of the wetland plant species identified as economically important are threatened with extinction. The main threat is habitat loss and degradation (affecting 95% of species), although nonsustainable level of harvest is identified as a major threat to some species (Juffe-Bignoli et al. 2012).

Future Challenges

So little is known about medicinal plants in wetlands that there is clearly a need for better documentation of their presence and value in this habitat. While it may be possible to use existing data sets more effectively to identify wetland medicinal plants, extending global, regional, and local surveys of wetland biodiversity to include the socioeconomic value of wetland plants will greatly contribute to a more comprehensive understanding of the global importance of these species to health and wellbeing, their conservation status and threats, and the conservation actions needed to support their long-term sustainable use.

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Traditional Medicines from Wetlands

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Donovan Kotze

Contents

Abstract

Traditional medicine refers to the knowledge, skills, and practices based on the theories, beliefs, and experiences indigenous to different cultures, used in the maintenance of health and in the prevention, diagnosis, improvement, or treatment of physical and mental illness (WHO 2004). The majority of wetland species used are plants, which are from a variety of different growth forms and taxa, e.g. the red mangrove (*Rhizophora mucronata*) the sedge *Cyperus articulatus*, the grass *Vetiveria zizanoides* and the herbaceous dicotylenous species *Alepidea amatymbica* (Ikhathazo. The most well-known wetland animal used for medicinal purposes is the leech (*Hirudo medicinalis*). Crocodiles and turtles are widely used as traditional medicines, especially in Asia and Africa. The use of wild populations of both plants and animals is likely to continue into the foreseeable future, requiring that the sustainability of harvesting be urgently addressed.

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Keywords
Traditional medicine · Wetlands · Plants · Chemical defences · Animals · Vulnerability to over-harvesting.

Introduction

Traditional medicine refers to the knowledge, skills, and practices based on the theories, beliefs, and experiences indigenous to different cultures, used in the maintenance of health and in the prevention, diagnosis, improvement, or treatment of physical and mental illness (WHO 2004). The focus of this contribution is on traditional medicines which are harvested from wetlands.

People have been using wetland plants and animals for medicinal purposes for millennia. For example, the medicinal properties of lotus (*Nelumbo nucifera*) are documented in Chinese writings over 2400 years old (Guo 2009). The bulk of the species used in traditional medicine are plants. Although no estimate exists for wetlands specifically, the number of plant species used for medicinal purposes globally is more than 50,000 (Schippmann et al. 2003).

Approximately 70–80% of people worldwide rely chiefly on traditional, largely herbal, medicine to meet their primary healthcare needs, and the global demand for these medicines is increasing (Hamilton 2004). Although some species are cultivated, most medicinal species are collected from the wild (Hamilton 2004) and this is likely to be so for wetlands.

Some Key Wetland Species Used in Traditional Medicine

Plants with medicinal properties are concentrated within certain families, which reflect particular ecological adaptations such as chemical defenses against herbivores, fungi, and pathogens (Horwitz et al. 2012). Medicinal plants harvested specifically from wetlands belong to a variety of different growth forms and taxa.

Trees and Shrubs

The flooded forests and swamp forests of the African, Asian, and South American lowland tropics contain a high diversity of medicinal trees and shrubs in the Apocynaceae (*Rauvolfia, Tabernaemontana*), Clusiaceae (*Clusia, Garcinia*), Rubiaceae (*Genipa*), and Euphorbiaceae (*Phyllanthus*) families (Cunningham, personal observations in Horwitz et al. 2012). In coastal wetlands throughout the tropics, the red mangrove (*Rhizophora mucronata*) is used in a variety of remedies. In temperate wetlands in the northern hemisphere, an important tree species is the White willow (*Salix alba*), which is the original source of salicylic acid, the precursor of aspirin, and is also used in skin care products (Ramsar Convention 2008).

Sedges, Rushes and Grasses (*Cyperaceae, Junaceae, Poaceae*)

These taxa, which epitomize wetlands, do not include many medicinal plants. Some exceptions are *Cyperus articulatus*, the Chinese water chestnut (*Eleocharis dulcis*) and the aromatic grass *Vetiveria zizanoides*. The rhizome of *Cyperus articulatus* is used in several African countries and Amazonia in the treatment of a wide variety of complaints, including headaches, migraines, and stomach aches, and has been demonstrated to have sedative properties (Rakotonirina et al. 2001). *Eleocharis dulcis* corms, widely favored in Chinese cooking for their sweet apple flavor and crisp texture, have antibacterial properties and are used to make a tonic and treat a number of ailments, including abdominal pain, amenorrhea, hernia, and liver problems (Parr et al. 1996). The rhizome of *Vetiveria zizanoides* is used to treat rheumatism, and vetiver oil is used as a carminative in flatulence, colic, and obstinate vomiting (Swapna et al. 2011) and has also demonstrated antibiotic properties (Gangrade et al. 1990).

Bulrushes/cattails (*Typhaceae*)

Several different plant parts from *Typha* spp. are used to treat a host of different conditions. Female flowers have been traditionally used in North America, the Middle East, India, and China for external treatment to promote the healing of wounds and burns, and experimental investigation on *Typha domingensis* by Akkola et al. (2011) confirmed this remarkable wound-healing activity. The rhizome is stated to be detersive (acts like a detergent), astringent, and stimulant, and is used as a diuretic in cases of retarded or painful urination, and even as a remedy for dropsy, dysentery, gonorrhea, and the measles (Morton 1975). In China, *Typha* pollen is administered as an astringent and as a diuretic and is an important component of traditional Chinese medicine.

Herbaceous Dicotyledonous Plants

Some wetlands, particularly bogs and mountain seeps, may support a diversity of dicotyledonous plants used for medicinal purposes (Ramsar Convention 2008; Horwitz et al. 2012). Examples include:

- Bogbean (*Menyanthes trifoliata*), a wetland species occurring at high latitudes in bogs and shallow water, is used traditionally for digestive ailments and is a commercially approved treatment in Germany for dyspeptic discomfort and loss of appetite.

- Sundews (*Drosera* spp.), insectivorous herbs found in acidic bogs, have proved useful in homeopathic and pharmaceutical treatments for bronchial complaints.
- Labrador tea (*Ledum palustre*), a widespread bog bush, is used against many illnesses and also as a mosquito repellent.
- River pumpkin (*Gunnera perpensa*), widespread in montane and foothill seepages and marshes of southern Africa, is used widely to ease childbirth and promote the expulsion of the afterbirth in both humans and livestock, and the effect of *G. perpensa* rhizome extract on the contraction of uterine smooth muscle has been well demonstrated (Khan et al. 2004).
- Ikhathazo (*Alepidea amatymbica*), widespread in montane and foothill seepages and damp grasslands of southern Africa, is used as a tonic and to treat coughs and chest complaints.

Most plants harvested from the wild for traditional medicine are native species, but some are introduced. An interesting wetland example is calamus, also known as sweet flag (*Acorus calamus*). Calamus is thought to have originated in Central Asia or India, from where it was introduced to Europe. European settlers, who grew it for medicinal purposes, then introduced it to North America, where it spread into the wild and was taken up by local indigenous populations, becoming one of their most commonly used medicinal plants. The wonderfully aromatic rhizomes are considered to possess antispasmodic, antihelminthic, anti-inflammatory, and antioxidant properties, and are widely used for oral hygiene, the treatment of digestive complaints, and as a stimulating tonic.

Animals

Probably the most famous wetland animal to be used for medicinal purposes is the leech (*Hirudo medicinalis*). The first recorded use of medicinal leeches is from Ancient Egyptian wall paintings over 3300 years old (Whitaker et al. 2004). During feeding, leeches secrete a complex mixture of substances into the wound, including inhibitors of blood coagulation, anti-inflammatories, and vasodilators (Singh 2010). Leeches are used in traditional medicine and in modern western medicine to assist in the treatment of abscesses, arthritis, glaucoma, myasthenia gravis, thrombosis, and some venous disorders and in plastic surgery, for example to aid in the salvage of replaced digits, ears, and lips. Many of the medicinal benefits of leeches have been well substantiated through research (Whitaker et al. 2004; Singh 2010), and leeches are still used routinely for certain mainstream medicine procedures by the National Health Service (NHS) in the UK.

Reptiles are among the animals most favored in traditional medicine globally (da Nóbrega Alves et al. 2008); turtles and crocodiles are the reptiles most strongly associated with wetlands. In China, turtle carapace is widely used in traditional medicine, for example to treat diseases such as night sweats and amenorrhea (the absence of a menstrual period in women of reproductive age). Crocodile products, including meat, skin, fat, and bile, are used to treat a variety of different conditions in both Asia and Africa.

Future Challenges

Although much research is still required, the scientific validity of many plant species medicines has been positively demonstrated. In contrast, there is a lack of studies to substantiate the effectiveness of many animal products used for traditional medicines, except for leeches. Nonetheless, the use of wild populations of both plants and animals is likely to continue into the foreseeable future, requiring that the sustainability of harvesting be urgently addressed.

Plant species vary greatly in terms of their inherent vulnerability to overharvesting, with the most susceptible generally being habitat-specific, slow-growing and those which are destructively harvested (Schippman et al. 2003). Animals used for medicinal purposes also face growing pressure. More than half of all turtle species from southeast and eastern Asia, for example, are currently endangered or critically endangered, largely because of their overcollection by the food and traditional medicine industries (da Nóbrega Alves et al. 2008).

WHO et al. (1993) provide a framework for the conservation and sustainable use of plants in medicine. Specific recommendations are given for each country to prepare a national strategy for the conservation and sustainable use of its medicinal plants. This framework highlights how no single sector can undertake the conservation of medicinal plants alone, but rather the task requires a team effort, involving a wide range of disciplines and institutions.

With increased awareness of the consequences of overharvesting, cultivation of wild species is being promoted. However, it is recognized that cultivation is not a panacea for overexploitation and may, for example, lead to the loss of incentives to conserve wild populations (Schippmann et al. 2003). Harvesting is often one of several impacts affecting the population of a particular species, and the fate of the species may be affected by several interacting socioeconomic factors. These need to be understood, and all key role-players need to be engaged in order that an appropriate mix of conservation in the wild and conservation through cultivation is developed. Practical guidelines are available such as those of Diederichs (2006) which assist in understanding the key socioeconomic drivers and the dynamics of the medicinal plant industry, the regulatory framework for harvesting and trade in medicinal plants, organizational development and sustainable harvesting, and propagation technologies. Workable conservation solutions have also to address the fact that a large proportion of the medicinal use of plants and animals may occur outside of the formal economy. In addition, ongoing action research is required to guide and strengthen initiatives to promote more sustainable harvesting of species from wetlands and other ecosystems.

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Reed Products from Lake Burullus, Egypt 153

Kamal Hussien Shaltout

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Abstract

Lake Burullus is a shallow, brackish, Mediterranean coastal lagoon situated between the two branches of the Nile that form its Delta. It is one of the Protected Areas of Egypt, registered as a Ramsar site and Important Bird Area. It has an area of 410 km^2, maximum length and width of 47 and 14 km. Its depth varies between 0.4 and 2 m. Common reed *Phragmites australis* (Cav.) Trin. ex Steud in Lake Burullus offers provisioning service products as it is high-quality livestock forage and silage during its early growth stages, while at maturity it becomes tough and unpalatable. It was an important source of matting in ancient Egypt and is still widely used for this purpose, used as a soil binder to prevent erosion, valued resource for thatching and construction of windbreaks for crop protection and nets for fishing and capture of birds. It is reported as a folk

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medicine for treating leukemia, bronchitis, cholera, diabetes, dropsy, gout, rheumatism, typhoid, antiemetic for acute arthritis, jaundice and food poisoning. Potential additional uses include production of paper pulp; some success has been achieved in Sweden in growing and harvesting *P. australis* for energy production. In South America, the natives have used its stems to make arrow shafts, prayer sticks, weaving rods, pipe stems, screens, and nets. Unfortunately, the area of common reed in Lake Burullus decreased by about one third in one decade (1988 - 1998). However, its management in Lake Burullus should include periodical partial removing between the islets in order to avoid lake fragmentation into four disconnected basins. Anyhow, reed harvesting in Lake Burullus can hold back successional processes.

Keywords
Mediterranean wetland · Nile delta · Common reed · Reed products · Livestock forage · Folk medicine

Introduction

Common reed *Phragmites australis* (Cav.) Trin. ex Steud. is believed to be the most widely distributed of all angiosperms. Although originating initially in the old world tropics, it is remarkable for being equally at home in the countries of the northern temperate zone and Australasia. It is common throughout the swamps of the Nile. Although excessive growth of common reed can impair other uses of waterways, pastures, and arable fields, it has many beneficial uses such as shelter, wind break, thatch, forage and refuge for animals, fuel, fertilizer, for making craft items, mats and baskets, and as a raw material for the paper industry (Holm et al. 1977). The rhizome is reported as useful in folk medicine (Boulos 1983). The common reed also plays an important environmental role in the remediation of the polluted water. The shores and islets of Lake Burullus comprise one of the most important reedbeds in the Mediterranean region, where this type of habitat is becoming rare and threatened (Shaltout and Al-Sodany 2008). Wintering and migrant birds are strongly dependent on this habitat for foraging, refuge, and breeding. The reedbeds also create a suitable shelter for the fishes of this lake, particularly fry and juveniles.

Site Characteristics

Lake Burullus is situated between the two principal branches of the Nile that form the Delta (Fig. 1). It is one of a network of Protected Areas throughout Egypt and registered as a Ramsar site. BirdLife International has identified it as an Important Bird Area (IBA). It is a shallow, brackish, Mediterranean coastal lagoon with an area

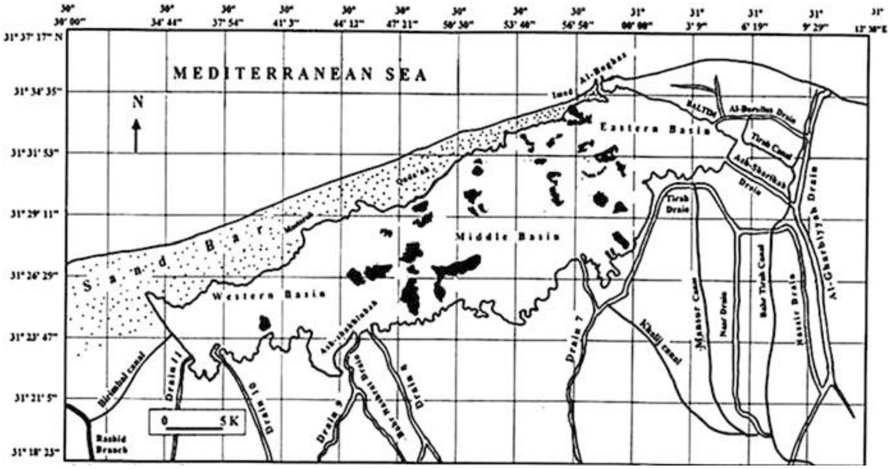

Fig. 1 Map of Lake Burullus along the Deltaic Mediterranean coast of Egypt. The black areas represent the islets scattered inside the lake. (Image credit: KH Shaltout © copyright remains with the author)

of 410 km^2, a maximum length of 47 km and a maximum width of 14 km. Its depth varies between 0.4 and 2 m. Biodiversity includes numbers of rare, endemic, and threatened species. This includes diverse plankton, higher plants, and fauna including birds. Fisheries provide the principal life-support system for the local inhabitants: production approximates 51,000 ton year^{-1}. Seventeen thousand licensed fishermen and their families depend on these resources for their living. Other resource uses include: agriculture, livestock farming, fish farming (155,000 ton year^{-1}), reed harvesting, bird hunting, tourism, and recreation (Shaltout and Khalil 2005).

Provisioning Service Products

Fodder for Livestock and Wildlife

Common reed is high-quality livestock forage during its early growth stages, while at maturity it becomes tough and unpalatable. Consequently, livestock should be fed a protein supplement when grazing mature reed. In the Burullus wetland, it is readily eaten by buffalos, cows, sheep, and goats (Fig. 2). Common reed is also used as silage, with the leaves and stems cut into small pieces and placed in silos (large receptacles with airtight sides and bottoms). This enables fermentation to take place, producing a more palatable food source for cattle.

Fig. 2 Buffalo and ducks feed, drink, and swim in Lake Burullus. (Photo credit: KH Shaltout © copyright remains with the author)

Fencing, Thatching, and Matting

Common reed was an important source of matting in ancient Egypt and is still widely used for this purpose. Common reed is also used in the Lake Burullus region as a soil binder to prevent erosion. It is also a valued resource for thatching and the construction of windbreaks for crop protection and nets for fishing and capture of birds. Villagers also use early stages of reed growth to pasture their stock animals, and reeds reaching maturity are cut and sold at a market price of about LE 0.20–0.60 per bundle. Although the sale of reeds is not of itself economically feasible, use of reeds as animal fodder throughout the year is significant (Shaltout and Khalil 2005). However, intensive use of common reed can adversely affect the density of reed stands.

Folk Medicine

P. australis is reported to be used in a wide range of folk remedies including, for example, leukemia, bronchitis, cholera, diabetes, dropsy, gout, rheumatism, and typhoid (Hartwell 1982) and as an antiemetic for acute arthritis, jaundice, and food poisoning (Boulos 1983).

Potential Additional Uses

Production of Paper Pulp

Egypt suffers from a shortage of wood for the production of paper pulp. Common reed can partially reduce this shortage. In other parts of the world including the Danube Delta (Romania), common reed is converted into pulp for the production of printing paper (Holm et al. 1977).

Energy Production

Some success has been achieved in Sweden in growing and harvesting *P. australis* for energy production, providing an evidence base about propagation methods, nutrient requirements, and harvesting technology (Hansson and Fredriksson 2004). The biomass observed along the watercourses in Nile Delta and Lake Burullus is 29–33 ton ha^{-1} (El-Kady 2000, Shaltout et al. 2004), which could in theory be harvested as an energy source.

Other Potential Products and Uses

Endemic people in South America have used the stems of common reed to make arrow shafts, prayer sticks, weaving rods, pipe stems, screens, and nets. It is also used for lattices and in the construction of adobe h (Kiviat and Hamilton 2001) and plaited into sandals with the culms carved into writing pens.

Future Challenges

Based on LandSat TM data, the area of common reed in Lake Burullus decreased by about one third in the decade between 1988 and 1998 (Fig. 3). If this rate of decline continues, reeds around Lake Burullus would be eliminated within few decades (Shaltout et al. 2004).

As a converse trend, the management of common reed in Lake Burullus should include periodical partial removal between the islets in order to avoid fragmentation of the lake into four disconnected basins.

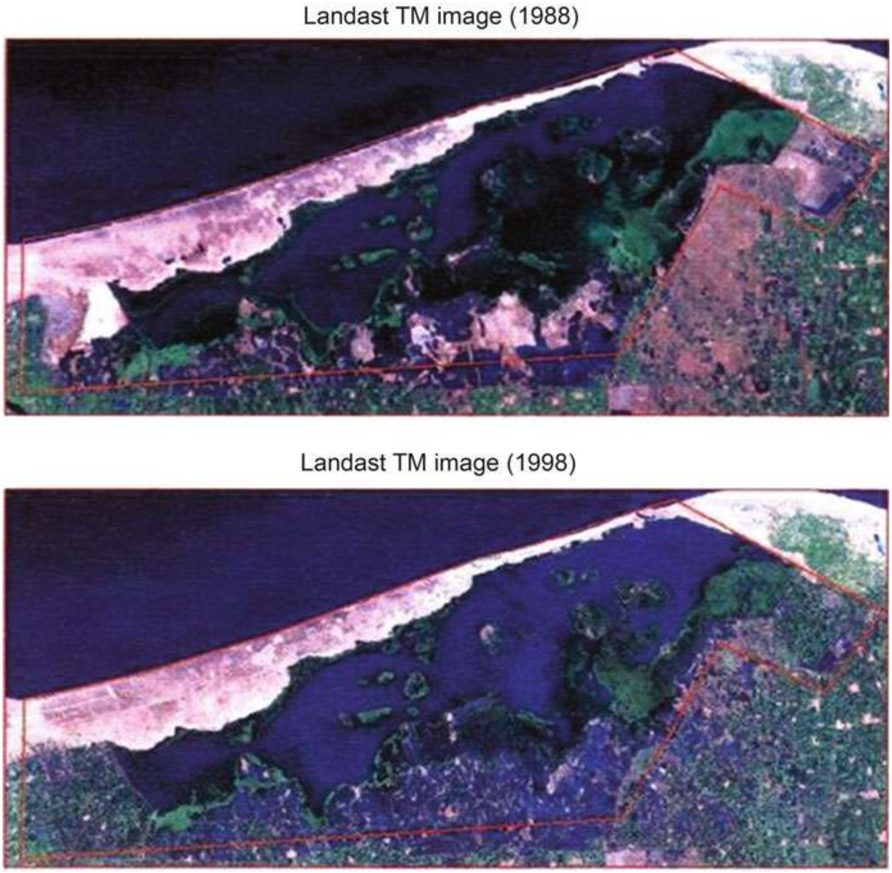

Fig. 3 LandSat TM images of Lake Burullus indicating the changes in the areas of the lake and common reed (*Phragmites australis*) during 1988 and 1998

Reed harvesting for commercial purposes in the south-eastern part of Lake Burullus can hold back successional processes.

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Salt production from Secovlje Salina Nature Park, Slovenia

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Andrej Sovinc

Contents

Abstract

The Secovlje Salina extends over 7 km^2. It is located in the north-eastern part of the Adriatic Sea, Slovenia. The area is one of very few salt-pans in the Mediterranean where salt is still produced in accordance with several-centuries-old method. Sustainable production of salt based on evaporation of the sea water and favourable biotic and abiotic factors in Secovlje Salina offers suitable conditions for a very diverse network of habitats and rich flora and fauna. The area is designated as Nature Park, Natura 2000 and Ramsar site. Salina has been assigned the status of extraordinary cultural heritage and landscape.

The Government of the Republic of Slovenia made an agreement for management of the protected area with the private company, which also held the concession for traditional salt-making process. Business model for the company Soline is a combination of commercial activities compatible with the conservation objectives for the area.

Despite the resurrection of the traditional salt-making process with typical wetland products and the increased number of visitors to the park, the number and

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distribution of selected indicator bird species nesting in the salt-pans has been increasing as a result of the implementation of the protective regime and the concept of conation.

Keywords
Secovlje Salina Nature Park · Artisanal salt production · Business model · Wetland products

Introduction

Salt production is one of the oldest industries known to man. In the past, several tens of traditional salt-pans have been located along the river estuaries in the Mediterranean. The Secovlje Salina extends over 7 km^2. It is located in the north-eastern part of the Adriatic Sea, in the east side of the Trieste Bay (Fig. 1). Together with adjacent smaller Strunjan Salina, these are the last remaining salt-pans in Slovenia and one of very few salt-pans in the Mediterranean where salt is still produced in accordance with several-centuries-old method.

Combination of the sustainable use of natural resources (production of salt based on evaporation of the sea water, passing through the extensive network of basins

Fig. 1 Geographical position of the Secovlje Salina Nature Park (Image credit: M Everard © copyright remains with the author)

with shallow water), use of traditional hand-work methods with limited human disturbance, and favourable biotic and abiotic factors in Secovlje Salina offers suitable conditions for a variety of wildlife species. Biodiversity values of the area are especially pronounced at the southern part of the Secovlje salt-pans where salt production was abandoned in 1960s and the level of human disturbance is the lowest. Botanically, the area is characterized by halophytes; more than 40 plant species from the national list of endangered plants can be found there (Kaligaric and Tratnik 1981). Important habitat types include mud- and sandflats, Mediterranean salt meadows and salt marshes, *Spartina* swards and estuaries (www.kpss.si). Although Secovlje Salina is home to several animal species, characteristic for saline environment, the area is best known for its birds. Around 300 species have been recorded, including nationally most important breeding populations of *Sterna albifrons*, *Charadrius alexandrinus*, *Recurvirostra avosetta*, *Himantopus himantopus*, and others; the Salina is also important stopover area for migrating birds (Skornik 2012).

The area is designated as Nature Park (IUCN protected area category V), Natura 2000 site, and Ramsar wetland of international importance. Moreover, the Secovlje Salina has been assigned the status of extraordinary ethnological, technological, historical, and architectural heritage and landscape.

Historical Development of the Site

Artisanal Mediterranean salt-pans had extremely important role in the local economy and for the founding of the culture. The town of Piran, adjacent to the salt-pans, was "built on salt" due to visible links between salt production, trade, and development of the local community. People from this town spent warmer part of the year in the Secovlje Salina, producing precious sea-salt and returned to the town for the winter.

The situation on the European market of salt dramatically changed after the WWI, when the salt produced with low costs along the North African coast and in European salt mines and "salt factories" caused the decline of several dozens of traditional salt-pans in the northern Mediterranean. Several artisanal salinas were abandoned or transferred into tourism resorts, aquaculture areas, or harbours. At the northern part of the Secovlje Salina salt-production was never abandoned, while the southern part of the area was left mainly to the processes of natural succession as the salt-making process was abandoned in the 1960s.

Institutional and Management Agreements

The Government of the Republic of Slovenia designated the area of Secovlje Salina as the state protected area due to its high biodiversity values in the year 2001, but no management authority has been put in place. Salt-pans are man-made areas and require active management, especially control over the water regimes, if they are to deliver their production and conservation functions. Secovlje Salina Nature Park was on the way to become just another “paper park” if a new and innovative management agreements would not be established.

The Government of the Republic of Slovenia made an agreement for management of the Nature Park with the private company “Soline d.o.o.,” which also held the concession for traditional salt-making process in the area in the year 2003.

The combination of performing the public service of the park management by a private company and commercial activities (artisanal salt production) resulted in a special form of public-private partnership. The company Soline was given a concession for management of the area according to the management plan which is approved by the Ministry of Environment of the Republic of Slovenia. All the land and infrastructure in the Secovlje Salina Nature Park remains state-owned property.

Business Model and Wetland Products

The Decree on the establishment of the Secovlje Salina Nature Park (Uredba 2001) recognizes not only the compatibility between the maintenance of the traditional salt-making and conservation but also requires that artisanal salt-making activities should be performed in order to secure favourable ecological conditions for specific species and habitats, highly dependent of the salty environment.

Traditional salt production process starts when sea water pours into the evaporation areas by the high tide. The water flows through the evaporation areas are regulated partly by the free-fall and partly by using water-pumps. Salt crystals in the traditional salt-pans are segregated during hot summer months by natural evaporation of the sea water in the crystallization pans. This process is stimulated by daily harvesting of the salt, which is done by salters, using traditional wooden tools. The bottom of the crystallization pans (cavedini) is covered with a thin layer of “petola.” The petola is composed of gypsum, carbonate minerals, and blue-green algae, and it is only used in the salt-pans of Secovlje and Strunjan (www.kpss.si).

The company Soline was bought by a major telecommunication company which provided substantial initial funding for the reconstruction of the traditional salt-making process and invested into the infrastructure for visitation of the area in the years after signing the concession contract. The main business interest of the telecommunication company was to obtain the profile of environmentally responsible company, investing part of the incomes into conservation of nature and maintenance of the cultural heritage and thus attracting interest of new customers. However, high intrinsic interest of the leadership of the company for the preservation of this national heritage asset could not be neglected.

Business model for the company Soline is a combination of commercial activities compatible with the conservation objectives for the area. Geographical conditions in Secovlje Salina do not allow competition with other producers of salt in the quantity of salt, so the company’s orientations are in production of high quality salt, rich with

Fig. 2 Natural wetland products from Secovlje Salina Nature Park: “Fleur du Sal” and the Piran salt

minerals and produced on 700 hundred years old method, without using any chemicals, bleaching, or additives. This approach proved to become commercially viable and is at the same time supportive to the maintenance of the ecological character of the coastal wetland ecosystem. In addition, cosmetics products, based on hypersaline water and mud, have been developed. Salt and cosmetics from Secovlje Salina are developed under separate trademark programmes, promoted as natural products and available on the national market and in over twenty countries worldwide (Fig. 2).

The annual quantity of the salt, harvested in the salt-pans of Secovlje and Strunjan, depends on the weather and reaches up to 5,000 tons per year.

Part of the company, which is responsible for the park management, generates incomes from three major sources: the contribution from the state, park own incomes, and international projects. As the government is not providing funding to the amount agreed in the management plan (approximately two thirds of the yearly park’s management and financial plan), the company covered the financial gap in the last years. But this proved to be unsustainable and new organizational and financial schemes are currently under discussion.

Number of visitors to the park has been increasing owing to a variety of programmes offered to visitors and tourists and park’s own incomes are reaching more than one third of the yearly budget.

Implementation of the business vision based on production of the traditional, entirely natural substance – Piran salt – as well as on active management of the nature park resulted in increased number of employees and visitors to the area. From 2002 (when the park saw the introduction of active management and the resurrection of traditional salt-making) to 2012, the numbers of employees increased from 15 to more than 90, while the number of visitors in the same period grew from approximately 8,000 to almost 50,000 visitors per year (Sovinc 2012).

Despite the resurrection of the traditional salt-making process with typical salt-pan products and the increased number of visitors to the park, the number and distribution of selected indicator bird species nesting in the salt-pans (Fig. 3) has

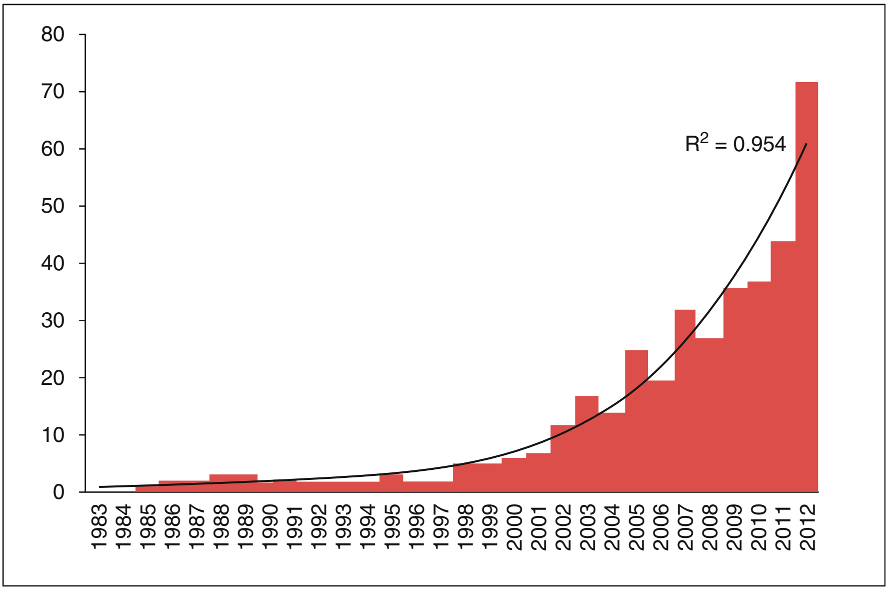

Fig. 3 Increase in number of breeding pairs of the little tern (*Sterna albifrons*) in Secovlje Salina Nature Park (Skornik 2012)

been increasing as a result of the implementation of the protective regime and the concept of zonation according to which park visitation and salt-making areas are limited to specially designated zones.

Future Challenges

Lack of funding for even basic functioning of protected areas, including wetlands, is a common issue at the national (Kus Veenvliet and Sovinc 2009) as well as at the global level (Flores and Rivero 2008). Resources available are often hardly sufficient for covering basic costs of the park's management and maintenance. Even if wetland products are recognized, there is often not enough funding available for their development, marketing, and branding. Traditional funding lines, including contributions from the national and regional authorities responsible for management of protected areas, need to be complemented with new and innovative forms.

In case of the Secovlje Salina Nature Park, commercial experiences and initial funding for development and marketing of the wetland products were sought from the company with the concession for both use of natural resources of the area and its management for conservation. Final wetland product, traditionally produced salt, was developed jointly with marketing campaign. The company Soline is determined to continue with producing and selling Piran salt. In achieving this goal, the company will maintain the saline ecosystem and its diverse habitats that are sustaining the characteristic flora and fauna and safeguard the traditional production process and technology of salt-making.

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Sustainable Use of Papyrus from Lake Victoria, Kenya

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Anne A. van Dam and Julius Kipkemboi

Contents

Abstract

Cyperus papyrus-dominated wetlands in eastern and southern Africa are important for millions of people because of their provisioning ecosystem services (food, water, materials, medicines) but also because of regulating services (e.g., water and nutrient retention, climate regulation), cultural services (heritage of wetland communities, importance for science and tourism), and biodiversity. Papyrus wetlands are under pressure from agricultural and urban development. Freshwater and food production are important, but sustainable management strategies are needed to protect regulating services and biodiversity. This chapter summarizes current uses of papyrus wetlands in the East African region and identifies natural and human-induced factors affecting these. To achieve sustainable management,

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more awareness of regulating ecosystem services and more quantitative methods for assessing them are needed. This will allow better estimation of the value of papyrus ecosystem services and better evaluation of trade-offs with conversion to other uses.

Introduction

Wetland degradation poses a threat to biodiversity, ecosystem integrity, and human well-being. The important ecosystem functions of wetlands support not only the health and biodiversity of terrestrial and aquatic ecosystems but also the livelihoods of large numbers of people (Millennium Ecosystem Assessment 2005; Russi et al. 2013). The *Cyperus papyrus* (L.)-dominated wetlands in eastern and southern Africa, commonly referred to as papyrus marshes, are good examples of this. It is estimated that millions of people depend on these wetlands for diverse products such as food, water, medicine, handicrafts, utensils, building, and other materials (Gaudet 2014).

Despite the importance of these marsh systems, African wetland area and aquatic biodiversity are decreasing at alarming rates (Owino and Ryan 2007; Rebelo et al. 2010; Darwall et al. 2011). The average rate of loss of papyrus wetlands over the last 50 years was estimated at 3% per year (van Dam et al. 2014). In Africa, the rural communities living adjacent to wetlands depend heavily on their ecosystem services, so changes in the ecosystem can have direct implications for people's lives. Developing sustainable management strategies for papyrus wetlands is therefore important and urgent. This contribution provides an overview of the different uses of papyrus wetlands and discusses options for their sustainable use in the context of the ecosystem services framework.

Uses of Papyrus Wetlands: Ecosystem Services

Table 1 summarizes a number of uses of papyrus wetlands using the ecosystem services (ES) framework of "The Economics of Ecosystems and Biodiversity" study (TEEB 2010; Russi et al. 2013) with four ES categories: provisioning, regulating, cultural, and biodiversity/habitat services.

Provisioning Ecosystem Services

The most visible and direct use of papyrus wetlands is the harvesting of materials (food, fiber, minerals) from the natural wetland ecosystem (Fig. 1; Rongoei et al. 2013). Vegetation is harvested for many purposes (Fig. 2). Virtually every part of the papyrus plant, from rhizomes to umbel, can be utilized. The rhizomes and dry culms can be used as fuel. Culms are used for a wide variety of crafts, material for mat making and house construction (e.g., rope making and thatching), and fishing traps.

Table 1 Common uses of papyrus marshes in sub-Saharan Africa and link to ecosystem services (ES) category according to TEEB (Russi et al. 2013). Based on van Dam et al. 2014 and references therein

Use/activity	Examples of use
Provisioning services	
Vegetation harvesting	- Construction: house construction and thatching, boats and rafts/floating islands - Crafts, tools, and utensil-making: chairs and other furniture, baskets, wall hangings and other decorations, ropes, brooms, mats, paper, fishing traps, and baskets - Fuel/energy: dry rhizomes and culms used as firewood; briquette production - Food: harvesting of wetland plants for edible roots and other plant parts; medicinal plants - Fodder: young papyrus umbels are consumed by livestock; wetland grasses and other vegetation serve as fodder
Clay and sand mining	- House walls smoothing; brick production; pottery
Hunting/fishing	- Fish: various wetland species - Game: swamp antelopes (lechwe, kobs), hippopotamus
Crop production	- Seasonal subsistence farming: fertile soils rich in organic matter used for production of rice, maize and other cereals, cocoyam, vegetables, and cotton; umbels used to shade vegetable seedings after transplanting - Commercial farming: rice, sugarcane, and other crops often following permanent conversion to cropland by drainage; construction of irrigation infrastructure
Water source	- Drinking water: shallow wells at wetland edges, particularly during the dry season; larger scale for drinking water production (e.g., Nabajjuzi wetland, Uganda); livestock watering - Irrigation: use of water for crop production
Regulating services	
Water quality improvement	- Natural wetlands: uptake of nutrients and other compounds by vegetation; binding to soil. Examples: Nakivubo and Namiiro wetlands (Uganda); Kahawa swamp (Nairobi, Kenya) - Constructed wetlands: wastewater treatment by papyrus vegetation
Water quantity regulation	- Water storage; groundwater recharge; evaporation
Climate regulation	- Evaporative cooling; carbon sequestration
Cultural services	
Culture and tradition of wetland people	- Use as sacred places for worship and as a source of myths and traditional wisdom (e.g., the Luo people in Uganda, Kenya, and Tanzania, and the Dinka and Nuer people in South Sudan). Clay is used in traditional ceremonies among the Luhya and Kalenjin communities in western Kenya
Scientific research and education	- Papyrus wetlands have been the subject of scientific research since the early 20th century leading to insights on wetland biogeochemistry, hydrology, ecology, and management. Papyrus wetlands are important for BSc and MSc programs in various universities in Africa and Europe

(*continued*)

Table 1 (continued)

Use/activity	Examples of use
Ecotourism	- Wildlife habitat, tourism, bird watching, and other wild game observation areas
Biodiversity and habitat services	
Biodiversity and habitat	- Habitat for various endemic fish and bird species, swamp antelopes, amphibians, aquatic invertebrates, reptiles, and hippopotamus. Several papyrus wetlands are Ramsar sites (e.g. the Sudd wetland in South Sudan, the Okavango delta in Botswana, and Lake Naivasha, Kenya) or Important Bird Areas. There is also some evidence for papyrus marshes playing important roles as repositories for cichlid biodiversity in Lake Victoria

Fig. 1 Vegetation harvesting in papyrus wetlands, with papyrus umbels for broom making (**a**) and papyrus culms (**b**) drying in the field after harvesting, a bundle of papyrus culms (**c**), and harvested grasses (**d**) (Photo credit (**a**): J. Kipkemboi © copyright remains with the author; Photo credit (**b-d**): A.A. van Dam © copyright remains with the author)

Young tender umbels form good fodder for livestock while mature umbels are used by some communities for making brooms. Besides papyrus, other wetland plant species are harvested for construction and craft making (e.g., *Phragmites* sp., *Eichhornia crassipes*, wetland grasses and trees), for human consumption (herbs and medicinal plants), for fuel/firewood, or as fodder for livestock kept in the homesteads.

Fig. 2 Papyrus products, with mats for sale along the roadside (**a**), example of artwork produced with papyrus (**b**), and brooms (**c**) and fish traps (**d**) made from papyrus fibres (Photo credit: A.A. van Dam © copyright remains with the author)

Besides harvesting of naturally produced vegetation, papyrus wetlands are also used for crop cultivation. Both small-scale, low-input, often seasonal agriculture and large-scale commercial agriculture can be found in papyrus wetlands (Fig. 3). During the dry season, dramatic changes to papyrus wetlands can be observed in floodplains with large areas of vegetation converted to sugar cane, maize, rice, and vegetable farms. With the conversion of natural wetlands for agricultural production come changes to the geomorphology and hydrology of the wetland, such as building dikes for flood protection, channelization for crop irrigation and for exclusion of wildlife which feed on crops (such as hippos), and soil desiccation through clearance of above-ground vegetation and cultivation. Burning of wetlands is also common in many areas to reduce the amount of senesced biomass and to allow fresh biomass to regenerate as pasture for livestock grazing, or to clear wetland margins for seasonal agriculture or drive out wild game during hunting.

Wetland fauna is also harvested as a source of protein. Papyrus wetlands are important for fish production in two ways: directly, by providing habitat for fish species that are caught in the wetland using a variety of fishing gears (Fig. 4); and indirectly, by providing a breeding environment and shelter for fish that subsequently recruit to adjacent river and lake fisheries. Generally speaking, fish catches

Fig. 3 Agriculture in papyrus wetlands, with livestock grazing in Nyando wetland, Kenya (**a**); tractors plowing a papyrus wetland in Bugesera, Rwanda (**b**); commercial rice cultivation in Namatala wetland, Uganda (**c**); and small-scale agriculture in Nyando wetland, Kenya (**d**) (Photo credit (**a**-**c**): A.A. van Dam © copyright remains with the author; Photo credit (**d**): M.M Rahman © copyright remains with the author)

from papyrus wetlands are poorly documented and it is hard to estimate fish production. A variety of fishing techniques is used, ranging from gill nets and small basket traps to elaborate structures to capture fish (Fig. 4c). Some research has been done to enhance fish production using seasonal fishponds ("fingerponds"). Despite the high potential demonstrated from this research, wider adoption of this practice is still limited (Kipkemboi et al. 2010). Besides fish, other animals that are harvested include marsh antelopes, hippos and insects.

Another important provisioning service of papyrus wetlands is water for drinking and for irrigation. Shallow wells at the edge of the permanent wetland are important for wetland communities, particularly in the dry season. In some places, papyrus wetlands are the source for production of municipal drinking water (e.g., in Nabajjuzi wetland near Masaka, Uganda).

Regulating, Cultural and Habitat Ecosystem Services

Besides the provisioning of food and materials, the importance of papyrus wetlands for their regulating services is increasingly recognized. The high growth rate and

Fig. 4 Fishing in papyrus wetlands, with a fisherman showing a lungfish (*Protopterus aethiopicus*) in Nyando wetland, Kenya (**a**); hook-and-line fishing with a papyrus culm as float in Namatala wetland, Uganda (**b**); and a fish trap in Nyando wetland, Kenya (**c**) (Photo credit: A.A. van Dam © copyright remains with the author)

biomass of papyrus ensure rapid uptake and storage of nutrients. Nitrogen, phosphorus, and other nutrients are taken up by plants and stored in their biomass (Muthuri and Jones 1997). Phosphorus is also bound to the sediment (Kelderman et al. 2007). Sediments from rivers are deposited in the wetland, and papyrus fringe vegetation reduces the entry of sediment particles into the adjoining water bodies (Boar and Harper 2002; Cohen et al. 2006). However, nutrient retention is probably strongly dependent on water flows between rivers/lakes and wetlands, with flushing of dissolved and particulate nutrients into the lake during high flow events. Papyrus wetlands are probably a source of DOC for Lake Victoria (Mwanuzi et al. 2003; Loiselle et al. 2008).

Nitrogen retention was estimated in model studies at around 22 g/m^2/yr (Hes et al. 2014). The nutrient uptake and storage capacity of papyrus can be utilized in constructed wetlands for wastewater treatment. Nutrient removal from wastewater by papyrus wetlands can be up to 70–80% for ammonium and orthophosphate (Chale 1985, Kansiime and Nalubega 1999).

Carbon storage in papyrus wetlands was estimated at over 700 tonnes of C per ha, mostly in the peat layer beneath the surface. In this sense, papyrus wetlands may be a significant carbon sink, but one that is vulnerable to hydrological drawdown and human activities (Saunders et al. 2014).

Papyrus wetlands also play a role in moisture circulation, influencing local and regional climate. Daily vapor flux through the canopy of a papyrus wetland at the Lake Victoria shoreline was found to be approximately 4.75 kg of water per m^2 per day, about 25% higher than water loss through evaporation from open water (Saunders et al. 2007). Estimates of annual evaporation from the Sudd wetland range from 1460 to 2100 mm (Sutcliffe and Parks 1989; Mohamed 2005). Although from the point of view of downstream water users evaporation is often considered a loss, papyrus wetlands contribute positively to local and regional precipitation (Zaroug et al. 2012).

With respect to cultural ecosystem services, papyrus wetlands play an important role in the history and culture of wetland communities like the Luo people in southern Uganda and western Kenya and Tanzania, and the Nuer and Dinka people in South Sudan. Traditionally, many papyrus wetlands harbored sacred places, although this practice is gradually disappearing (Kibwage et al. 2008). Some papyrus wetlands are important recreational and (eco)tourism destinations, particularly due their role as Important Bird Areas (IBAs). Papyrus wetlands form useful research and education objects, and some wetlands have been studied extensively by universities, government departments and NGOs (van Dam et al. 2014).

The biodiversity and habitat ecosystem services of papyrus wetlands are also significant, with several species of endemic birds, marsh antelopes and hippopotamus (see "Papyrus Wetlands" by Kipkemboi and van Dam, Vol. 4 for a more detailed description).

Factors Affecting Human Use of Papyrus Wetlands

Changes in the status and area of papyrus wetlands are influenced directly by two strongly interrelated factors: water and human activity. Seasonal weather dynamics (one or two rainy seasons) produce distinct permanent and seasonally inundated zones, leading to zonation in soil moisture and vegetation. In the permanently flooded zone, papyrus is extremely productive and can outcompete most other aquatic plant species, resulting in almost monotypic stands of *C. papyrus*. In the seasonally flooded zone, the papyrus competes with plant species that are more adapted to dry conditions (Rongoei et al. 2014). In the longer term, hydrological change can lead to the more or less permanent disappearance or creation of wetlands. For example, many fringing wetlands of Lake Victoria were drowned during the Uhuru rains in the early 1960s when the lake water level rose by about 2 m. New papyrus wetlands were formed in the newly flooded zones (Thompson 1976).

Seasonal hydrological dynamics have strong impact on human activities in the wetland. Flooding and waterlogging excludes most human activities in the wetlands. However, dry conditions make wetlands vulnerable to human activities, including livestock grazing. In wetlands with seasonal agriculture, vegetation can be removed and crops planted as soon as the flood has retreated. Vegetation harvesting is also affected by flooding, as flooding restricts access to harvestable stands. Sometimes, harvesting is done by boat. Harvesting frequency in papyrus stands determines the regeneration potential and consequently biomass production (Osumba et al. 2010). Water depths of more than 20–30 cm also impede livestock herding. By contrast, fishing is enhanced during the rainy season.

Indirectly, human population growth and economic development, climate change as well as formal and informal institutions and policy interventions are important determinants of wetland use. In East Africa, high population density combined with limited employment opportunities result in low incomes from non-wetland activities. Recent changes in weather patterns, leading to prolonged dry spells, can influence the vulnerability of the wetland to livelihood activities. Changes in rainfall patterns can cause crop failure in traditional rain-fed upland agriculture and lead to migration towards wetlands, where seasonal agriculture and livestock herding remain possible. Traditionally, there were many informal institutions (traditions, customary land tenure arrangements) that regulated wetland resource use. In some areas, these traditions are now changing. Formal institutions and instruments (e.g., regulatory agencies and wetland policies) are being established and developed in many countries, but implementation and enforcement of sustainable management practices remain a challenge (Kibwage et al. 2008).

Sustainability and Future Challenges

The declining trend in wetland area and biodiversity raise concerns about the sustainability of current wetland management practices. The legitimate need for economic growth and food security of African nations will lead to further degradation and loss of wetlands through agricultural and urban development. Sustainability is defined in terms of the wise use of wetlands as “the maintenance of their ecological character, ...within the context of sustainable development” (Finlayson 2012). With increasing numbers of people relying on the productivity of papyrus wetlands, intensity of use has been increasing leading to permanent changes to the papyrus ecosystem and its ecological character.

More intense use comes with changes in water supply and drainage of the wetlands. Upstream water abstraction and damming of rivers leads to reduced surface or groundwater flows to papyrus wetlands. Removal of papyrus rhizomes and construction of drainage channels prevent recovery of the papyrus vegetation during the rainy season. The ensuing drying and mineralization of the underlying peat layer and release of the stored nutrients (including carbon) create conditions for plant species that are more adapted to dry conditions. More intense tillage increases sediment erosion. With time, the lower natural accretion of sediment and nutrients requires the import of nutrients in the form of fertilizers to maintain crop productivity, which increases nutrient discharge to downstream areas. Application of chemicals to control pests and diseases in arable crops introduced to wetlands through agricultural practices may affect biodiversity. Burning may become more frequent, disrupting the papyrus climax community and providing opportunity for invasive species (Terer et al. 2012). Urbanization also leads to permanent changes in papyrus wetlands due to conversion, water abstraction and pollution. The results of these changes are increasingly observed, for example, in a decline in papyrus surface area as well as in bird populations (Maclean et al. 2014).

Conversion of papyrus wetlands to crop production enhances provisioning services, but the implications for regulating, cultural and habitat services are often not considered. This is partly due to the difficulty of quantifying these other ecosystem services, both in terms of material processes and economic value. A trade-off analysis as part of a management strategy is therefore not made (Zsuffa et al. 2014). As a result, African countries are losing the benefits and values of these important ecosystems. This also has implications for the equity of economic development, as poorer members of wetland communities depend particularly on highly diverse wetland livelihood activities whereas benefits from converted wetlands often accrue to fewer individuals or corporations. If sustainability is defined as "the long-term use of one or several of the wetland's ecosystem services without degrading or losing other ecosystem services" (van Dam et al. 2014), agriculture and other wetland uses should be developed in such a way that water quality functions and biodiversity values are taken into account and protected as much as possible.

To increase awareness of the importance of sustainable management of their papyrus wetlands, countries containing these systems need to assess the areal extent of land occupied by *C. papyrus* and the ecosystem services they provide, as a basis for monitoring change and identifying causes of degradation. There is a need to allocate adequate resources towards the implementation of wetland policies, where they exist, or to fast-tracking policy development where they do not exist. Management plans should be aimed at a better balance between agricultural and urban development and the maintenance of a wider suite of beneficial ecosystem services.

Research is required to provide better estimates of the value of regulating and cultural ecosystem services, and to develop methods for sustainable resource harvesting and crop and fish production that do not endanger the nutrient and water regulation functions of these wetlands. Experiences with attempts to improve wetland management in Africa have shown that stakeholder participation and capacity development are vital for the success of sustainable wetland management.

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Section XI

Management of Regulating Services

Robert J. McInnes

Management of Regulating Services: Overview

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Robert J. McInnes

Contents

Abstract

Regulating services are the ecosystem processes that affect climate, floods, disease, wastes, and water quality. In simple terms, regulating services can be thought to represent the services that "maintain" desired environmental conditions, such as a stable coastline or steady water supply. Cumulatively, these regulating services are essential to moderate climate, environmental hazards, surface and groundwater hydrology, and pests and to purify air, land, and water. Wetlands play a crucial role in providing regulating ecosystem services to human society.

Keywords

Hazards · Climate regulation · Water supply · Pest control · Disease control · Pollination

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Introduction

The concept of ecosystem services, describing the diverse benefits that the natural world provides to people, has been emerging since the late 1980s, but the recognition that nature is important to human society goes back to the beginnings of civilization (Mooney and Ehrlich 1997; McInnes 2008). Since the 1980s, disparate classification schemes have been developed often addressing specific habitat types (wetlands, coral reefs, rangelands, croplands, forests, etc.) and/or bioregions.

One of the many contributions of the UN's Millennium Ecosystem Assessment (MA) program (Millennium Ecosystem Assessment 2005) was the harmonization of these prior schemes into a consistent classification system suitable for comparison across major habitat types on a global basis. However, despite the attempts at harmonization, some ambiguities remain in the terminology (Lugo 2007) and also in the issues surrounding "double counting" of ecosystem services, especially when applying monetary valuation techniques (Boyd and Banzhaf 2007).

The primary division within the MA classification promotes a grouping of ecosystem services into the following four major categories:

Provisioning services
Regulating services
Cultural services
Supporting services

The Ramsar Convention's "wise use" concept is the longest established example of what have become known as ecosystem approaches for the conservation and sustainable development of natural resources (Finlayson et al. 2011). The Third Conference of the Parties (COP3), held in Regina, Canada, in 1987, defined sustainable utilization of wetlands as "human use of a wetland so that it may yield the greatest continuous benefit to present generations while maintaining its potential to the needs and aspirations of future generations." At the Ninth Conference of the Parties (COP9 2005), the definition of ecological character was redefined to include "ecosystem components, processes and benefits/services that characterise a wetland at a given time." This brought together two key concepts of the Convention (wise use and ecological character) and allowed for a formal alignment with the four categories recognized by the MA (Finlayson et al. 2011). Therefore, the category of regulating services has a formal international status and global resonance.

Regulating services are defined by the MA as ecosystem processes ". . .that affect climate, floods, disease, wastes, and water quality" (Millennium Ecosystem Assessment 2005). In simple terms, regulating services can be thought to represent the services that "maintain" desired environmental conditions, such as a stable coastline or steady water supply. Cumulatively, these regulating services are essential to moderate climate, environmental hazards, surface and groundwater hydrology, and pests and to purify air, land, and water.

Wetlands play a particularly significant role in moderating water-vectored services such as floods and storm surges or the cleaning of polluted water. However, many of these services are currently not valued by market economics, meaning that they, and the wetlands that provide them, are vulnerable to degradation or land use change in favor of other, more narrowly framed, services such as production of food and fiber (Gordon et al. 2010). The MA concluded that regulating services are among the least understood but potentially most valuable services offered by ecosystems (Millennium Ecosystem Assessment 2005).

Examples of Regulating Ecosystem Services

Regulating services are critical to maintaining and moderating ecosystem processes, such as hydrological discharges, carbon cycling, nutrient transformations, and fluctuations in local climate. The following provide brief examples of common regulating services associated with wetlands.

Climate Regulation

Peatland ecosystems cover 1.136 million km^2 of Canada's northern boreal region and have been estimated to store 147,000 Mt of carbon (Tarnocai 2006).When combined with the carbon locked up in Canada's northern forests, it is estimated that the total carbon is equivalent to approximately one third of the total carbon stored in the Earth's atmosphere making these wetlands globally significant carbon stores (Carlson et al. 2010). The defining characteristic of these peatlands is the gradual accumulation of carbon over millennia as net primary production exceeds decomposition. The net annual rate of carbon sequestration in the northern peatlands has been estimated to be approximately 20 gC/m^2. When this is scaled up to represent all of Canada's northern peatlands the estimate suggests that annual sequestration of carbon is about 23 MtC. In order to protect these globally important carbon stocks and maintain the sequestration processes it is essential that appropriate management actions are established. Failure to implement the appropriate management could release in significant carbon release and increase atmospheric CO_2. The following management actions were identified by forest, peatland, and climate experts in order to maintain the global climate regulation service of Canada's northern peatlands and associated forest systems (Carlson et al. 2010):

- Reduce deforestation and increase afforestation
- Avoid logging of natural forests
- Employ forest management practices that enhance carbon storage:
  1. Reduce soil disturbance and maintain coarse woody debris
  2. Silvicultural activities to increase productivity and accelerate regeneration
  3. Extend rotation periods
- Employ forest sector practices to enhance carbon storage and minimize greenhouse gas emissions:

1. Capture methane emissions from forest products at landfills
2. Increase recycling and switch production to longer lived forest products
3. Use energy in wood waste for power production

- Minimize the extraction of peat soils
- Minimize soil disturbance:
  1. Minimize ground disturbance in areas with saturated soils
  2. Avoid disturbance to permafrost
- Reduce the adverse climate impacts of fire and insect disturbances:
  1. Suppress fire and insect events where appropriate in the managed forest
  2. Restore the natural resilience of forest to disturbance
  3. Use salvage logging where appropriate to reduce harvest of undisturbed forest

The peatlands of northern Canada represent a vast wetland system. However, while not representing globally significant carbon stores, small wetlands can moderate air temperatures and regulate climate on a local scale (Pokorny et al. 2010). Studies in Stockholm, Sweden, have demonstrated that the city's microclimate is moderated by the large bodies of water in and around the city contributing to lowering of air temperatures and evening out temperature deviations both during summer and winter (Bolund and Hunhammar 1999).

Water Regulation

Wetlands play a significant role in the water cycle and moderate a variety of hydrological processes at the site and catchment scale (Bullock and Acreman 2003). The unique environmental properties of wetlands, including their geology, soils, and vegetation, can make them particularly effective at moderating water supplies to downstream users. For example, the drinking water for the 8 million citizens of Bogotá, Colombia, is supplied by a high-elevation wetland ecosystem called a *páramo* (a neotropical alpine wetland ecosystem found in the upper Andean region of Colombia, Venezuela, Ecuador, and Peru). The vegetation of the *páramo* absorbs, filters, and releases clean water at a reliable rate of 28 m^3/s with little seasonal variation and minimal need for subsequent treatment (Postel and Thompson 2005). This high reliability of water supply and quality translates into lower capital and treatment costs than would be the case without the *páramo*.

In addition to maintaining surface water supplies, wetlands can also play an important role in the recharge of groundwater and the subsequent support to agricultural systems and domestic water supplies. For instance, the Hadejia-Nguru wetlands in northern Nigeria are formed by the floodwaters of the region's two main rivers, the Hadejia and the Jama'are. The rivers have an ephemeral flow pattern, with a no-flow dry season between October and April. Conversely, approximately 80% of the total annual runoff occurs in following the rainy season in August and September (Thompson and Hollis 1995). During this period of runoff and inundation of the floodplain, waterlogged areas known as *fadamas* are formed which not only provide important fishing and agricultural resources but also significantly recharge the underlying aquifer (Hollis and Thompson 1993). Without this recharge function, both domestic water supplies and irrigation for agriculture would be severely compromised during the dry season.

Water Purification

It has long been established that wetlands possess unique properties that make them natural waste water cleaning systems. Across the world there are examples of wetlands being used to reduce concentrations of nutrients and other pollutants from flowing waters (Verhoeven et al. 2006). This regulating service can often be vital to maintaining clean water supplies downstream for human consumption, improving discharges to coastal waters, enhancing habitat quality for species of conservation concern, or protecting important natural resources such as lake fisheries from eutrophication.

Intensive livestock farming is common in many countries. In Ireland, both diffuse runoff of fertilizers applied to grazing pastures and point sources from the farm buildings and sheds have been implicated in the eutrophication of surface waters. In response to this issue, an initiative was developed for the Annestown stream catchment in south County Waterford to reanimate the landscape and to restore wetlands to regulate nutrient loads to both the river and the downstream coastal bathing waters (Scholz et al. 2007). The approach, which includes restored and created integrated constructed wetlands, explicitly seeks the integration of three objectives: water quantity and water quality management, including flood hazard management; landscape-fit towards improving site aesthetic values; and the enhancement of biodiversity to achieve sustainable management of waste water derived from livestock production (Harrington and McInnes 2009).

A series of wetlands were restored and created to intercept runoff from farmyards and cattle sheds, prior to discharging into a small stream. The reduction of phosphate for 12 different wetlands ranged from 81.36% to 99.71% with an average removal rate for the period between August 2001 and March 2009 of 95.60%. Similar rates of removal were also observed for ammonium-N concentrations with a range between 95.26% and 99.24% and an average rate of removal of 97.56% over the same time period (Harrington and McInnes 2009).

The functional area of the wetlands in the Annestown stream valley was identified as being the key factor in the removal efficiency for nutrients. This reinforces the view that wetlands need to have the appropriate functional capacities to deal with different types of effluent. Not all wetlands will be able to remove nutrients, for instance, at the same efficiency rates as illustrated by the integrated constructed wetlands in Ireland.

Eutrophication of surface waters from agricultural runoff is an issue faced by water bodies across the planet. Lake Victoria in Africa has seen a significant decline in its fishery since the mid-1990s. While overfishing has been implicated in this decline, eutrophication from agricultural runoff has also been identified as a significant factor. The increased use of fertilizers in the surrounding catchments and pressures from human population have reduced the buffering capacity of the surrounding wetlands and have had serious implications with regard to the water quality in the lake, with a shift from mesotrophic to eutrophic conditions observed (Simonit and Perrings 2011).

Estimates have been made of the buffering capacity of the role of the wetlands in the Yala estuary, which discharges into the Kenyan sector of Lake Victoria, to understand better of the value of a regulating service that is derived from the value of the provisioning services the wetlands protect. Through modeling, Simonit and Perrings (2011) found that the wetlands in the Yala estuary and around the margins of Lake Victoria played a role in reducing the nutrient loads to the lake. However, while this was recognized as an important regulating service, the modeling demonstrated that rather than rely on wetlands to moderate water quality and nutrient loads that source control, through better management of agricultural fertilizers, soil and land use practices was actually more important for the water quality of Lake Victoria.

Erosion Regulation

Coastal wetlands have long been recognized for their ability to stabilize shorelines and protect coastal communities. In many situations coastal wetland communities provide low-cost alternatives to engineered solutions for protecting the rural communities and reducing potential storm erosion damage. Given that approximately a third of the world's population inhabits coastal areas and small islands and more than 10% of all people on earth live within 10 m of sea level, this regulating ecosystem service is fundamental to future human well-being in light of rising sea levels.

The mechanisms for reducing coastal erosion are both direct and indirect (Gedan et al. 2011). Direct mechanisms include the physical ability of wetland plants to slow water velocity, reduce turbulence, and increase the deposition of sediments. Below ground, the dense network of plant roots directly reduces rates of erosion by stabilizing the soil substrate. Wetland plants indirectly affect coastal hydrodynamics in a variety of ways. Slowly decaying plant roots and the accumulation of organic matter produces organic rich soils which tend to be less susceptible to erosion than mineral soils. The ability of wetland plants to contribute to the accumulation of organic material and the building of peat soils also alters coastal bathymetry which is a fundamental control and wave energy. As vegetation growth and peat accumulation alters the elevation of intertidal areas, and given that wave height is proportional to the depth of water between the bed surface and sea level, wetland plants indirectly determine rates of dissipation of wave energy (Gedan et al. 2011). Once vegetation is established, wetland plants generate stagnation zones where sediment can be deposited and accumulated further elevating substrate levels. The long-term cumulative impact of these indirect mechanisms, by which wetland vegetation promotes sedimentation and mitigates erosion, results in lasting stability and accretion of shorelines.

Natural Hazard Regulation

The term wetland covers many different types of ecosystem. Each individual wetland type will perform different hydrological functions making it difficult to generalize the role of wetlands in reducing the flooding as a natural hazard (Acreman and Holden 2013). For instance, upland rain-fed, or ombrotrophic, wetland systems tend to be flood of generating areas, due to their high level of saturation and elevated water tables, whereas floodplain wetlands lower down a river system have a greater capacity to store water consequently attenuating and reducing the flood peaks.

Many floodplain wetlands have been disconnected from the main river channel through the construction of embankments in order to protect the relatively flat land for other uses such as built development and infrastructure. However, this strategy can often increase flood risk downstream due to a loss of physical storage. Hydrological and hydraulic numerical modeling was used to assess the impact of embanking or restoring floodplain wetlands on the River Cherwell in Oxfordshire, UK (Acreman et al. 2003). The restoration of the river channel through the floodplain would potentially reduce peak flow by between 10% and 15% and increase peak water levels stored within the floodplain by 0.5–1.6 m. However, the modeling predicted that building embankments along the river would potentially increase the river peak flows downstream by up to 150% (Acreman et al. 2003). This clearly demonstrates the importance of managing and restoring wetlands for flood risk reduction.

Pollination

Natural habitats, such as wetlands, can provide biological pest control and can also promote pollination which benefits agricultural production in adjacent land. In the United States, approximately 30% of all food supplied by volume depends on animal pollinators of which bee species (Apoidea) are the most important. While many farmers rely on colonies of the European honeybee *Apis mellifera*, native, unmanaged bee populations also provide important pollination services to various crops. A study by Kremen et al. (2004) demonstrated that the pollination service provided by native bees was significantly correlated with the proportion of natural habitats, including wetlands and uplands, on surrounding agricultural landholders. This relationship was demonstrated to be robust over space and time. Similarly, a study conducted on paired farms in Ireland found that insect-pollinated forb richness increased with proximity to wetlands (Power et al. 2012). The study suggested that the proximity of wetlands may increase overall plant richness in adjacent land by providing habitat for a range of species and forming a source from which colonization can take place, but more significantly increases in insect-pollinated plant richness may result from the fact that many pollinator species (including hoverfly and solitary bee species) utilize wetlands as larval and forage habitats.

Future Challenges

Wetlands clearly provide a great range of regulating services. Not only do these services regulate global and local climate, maintain water supplies, protect homes from hazards, and contribute to food production through controlling pest species and supporting pollinators, they also fundamentally maintain the ecosystem functions which support provisioning services. The future challenge is to ensure that the full value of regulating services is captured and incorporated into decision-making, otherwise society and decision makers will continue to undervalue the importance of wetland ecosystems.

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Regulating Services: The Basics

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Mark Everard

Contents

Abstract

The Millennium Ecosystem Assessment classification of ecosystem services comprised four major categories: provisioning services, regulatory services, cultural services, and supporting services. Regulatory services are defined by the Millennium Assessment as ecosystem processes ". . .that affect climate, floods, disease, wastes, and water quality". Cumulatively, these regulatory services are essential to moderate climate, hazards, hydrology, and pests and to purify air, land, and water resources. Wetlands are particularly significant in moderating water-vectored services such as floods and storm surges. However, many of these services are not valued by market economics, leaving them vulnerable to degradation in favor of more narrowly framed services. The Ramsar Convention's "wise use" concept recognizes the needs to balance the regulatory benefits provided by wetland systems with the production of provisioning, cultural, and supporting services.

Keywords

Regulation · Maintenance · Systemic management · Hydrology · Flood Regulation · Resilience

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Definition

The concept of ecosystem services, describing the diverse benefits that the natural world provides to people, has been emerging as a pedagogic and management tool since the late 1980s. Since that time, disparate classification schemes have been developed often addressing specific habitat types (wetlands, coral reefs, rangelands, croplands, forests, etc.) and/or bioregions.

One of the many contributions of the UN's Millennium Ecosystem Assessment (MA) program (Millennium Ecosystem Assessment 2005a) was the harmonization of these prior schemes into a consistent classification system suitable for comparison of major habitat types on a global basis. The primary division within the MA classification scheme was the grouping of ecosystem services into four major categories: provisioning services, regulatory services, cultural services, and supporting services.

Regulatory services are defined by the Millennium Ecosystem Assessment as ecosystem processes "*. . .that affect climate, floods, disease, wastes, and water quality*" (Millennium Ecosystem Assessment 2005a). Cumulatively, these regulatory services are essential to moderate climate, hazards, hydrology, and pests and to purify air, land, and water resources; wetlands are particularly significant in moderating water-vectored services such as floods and storm surges. However, many of these services are currently not valued by market economics, meaning that they and the ecosystems that provide them are vulnerable to degradation in favor of other, more narrowly framed, services such as production of food and fiber.

A Millennium Ecosystem Assessment synthesis specifically considering global wetlands and water (Millennium Ecosystem Assessment 2005b) (including lakes, rivers, marshes and coastal regions to a depth of 6 m at low tide but acknowledging that many wetland types were underrepresented) found that ". . .more than 50% of specific types of wetlands in parts of North America, Europe, Australia and New Zealand were destroyed during the twentieth century, and many others in many parts of the world degraded." Nevertheless, wetlands produce a diversity of all categories of ecosystem services, including significant regulatory services. The MA Wetlands and Water synthesis report also noted that "Maintenance of the key hydrological services performed by wetlands enables them to continue to deliver a wide range of critical and important regulatory and provisioning ecological services to humans" particularly including the attenuation of floods, storage of carbon, and maintenance of water flows in rivers and through the landscape upon which ecological integrity and the production of many other beneficial services depends. However, "Maintaining the hydrological regime of a wetland and its natural variability is necessary to maintain the ecological characteristics of the wetland, including its biodiversity" which, due to the observation in the MA that the degradation and loss of wetlands is more rapid than that of other ecosystems, threatens the capacity of the global and local wetland resources to contribute to human wellbeing.

The Ramsar Convention's "wise use" concept recognizes the needs to balance the regulatory benefits provided by wetland systems with the production of provisioning, cultural, and supporting services, cumulatively contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation.

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Balancing Water Uses at the Doñana National Park, Spain

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Laura Serrano

Contents

Abstract

The Doñana region, in south-western Spain, extends from the estuary of the Tinto river to that of the Guadalquivir river, and inland from the Atlantic coast to the Guadiamar river valley. It shelters one the most important wetland areas in Europe and gathers an impressive record of biodiversity which is partly inherited from its unique location on the path between two continents, and partly from a glorious past as a true wilderness during centuries. As much as biodiversity is supported by a mosaic of landscapes and habitats in Doñana, the key to their functioning lies on their ability to shift at several time-scales.

Keywords

Guadalquivir estuary · Seasonal marshland · Temporary ponds · Drought · Floods · Groundwater abstraction

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Introduction

The Doñana region, in south-western Spain, extends from the estuary of the Tinto river to that of the Guadalquivir river, and inland from the Atlantic coast to the Guadiamar river valley. It shelters one the most important wetland areas in Europe and holds a wide variety of aquatic systems (Fig. 1). This region displays an outstanding record of biodiversity coupled with a high primary production that supports over half a million wintering waterfowl on its migratory route to Africa every year.

The whole region covers nearly 3000 km^2 and approximately 40% is designated under specific protection plans, such as the Doñana National Park (542 km^2) and the surrounding buffer zone of the Natural Park (537 km^2) which both assemble into the Doñana Natural Area (Fig. 1). The Doñana National Park is a UNESCO World Heritage Site, a Biosphere Reserve, and a Ramsar Site though it has remained on the Montreux Record of Ramsar Sites under threat since 1990. The Aznalcóllar mining spill on the Guadiamar river in 1998 painfully showed the world that no park is an island, an ecological lesson concisely explained by Margalef as "the quality of the waters draining through a watershed reveals the health of the terrestrial ecosystems within that watershed." The Guadalquivir river watershed is about 100 times larger than the Doñana National Park, posing a huge challenge to both wetland managers and hydrological authorities.

The region is inhabited by a permanent population of over 200,000 people distributed in several villages and concentrated in four towns which each exceed 20,000 inhabitants (Almonte, Moguer, Lebrija, and Sanlúcar de Barrameda, Fig. 1). The population doubles in the tourist coastal resorts every summer and peaks to over half a million during the height of the pilgrimage to El Rocío village in late spring. The local economy is based on tourism and a combination of traditional uses (horse and cattle ranching, forestry, vineyards, olive groves, game hunting, fishing, shellfish, and crayfish harvesting) with a more recent production of rice, aquaculture, and an intensive groundwater-irrigated agriculture.

Biodiversity and Land Use

The Doñana National Park supports an impressive record of biodiversity which is partly inherited from its unique location on the path between two continents and partly from a past as a true wilderness during previous centuries. The list of vascular plants includes over 900 native species while more than half of Europe's bird species have been recorded in Doñana, many of them in large numbers. It is one of the few refuges for endangered species, such as the Iberian lynx or the Spanish imperial eagle though a humble microscopic invertebrate (the rotifer *Lecane donyanaensis* nova sp.) is the only animal species bearing the name of the region (García-Novo and Marín-Cabrera 2006). The Doñana wetlands are inhabited by half of all branchiopod species recorded in the Iberian Peninsula (Fahd et al. 2009) and by 83 aquatic plant species though 25% of this flora is under some threat according to the IUCN (García-Murillo et al. 2006).

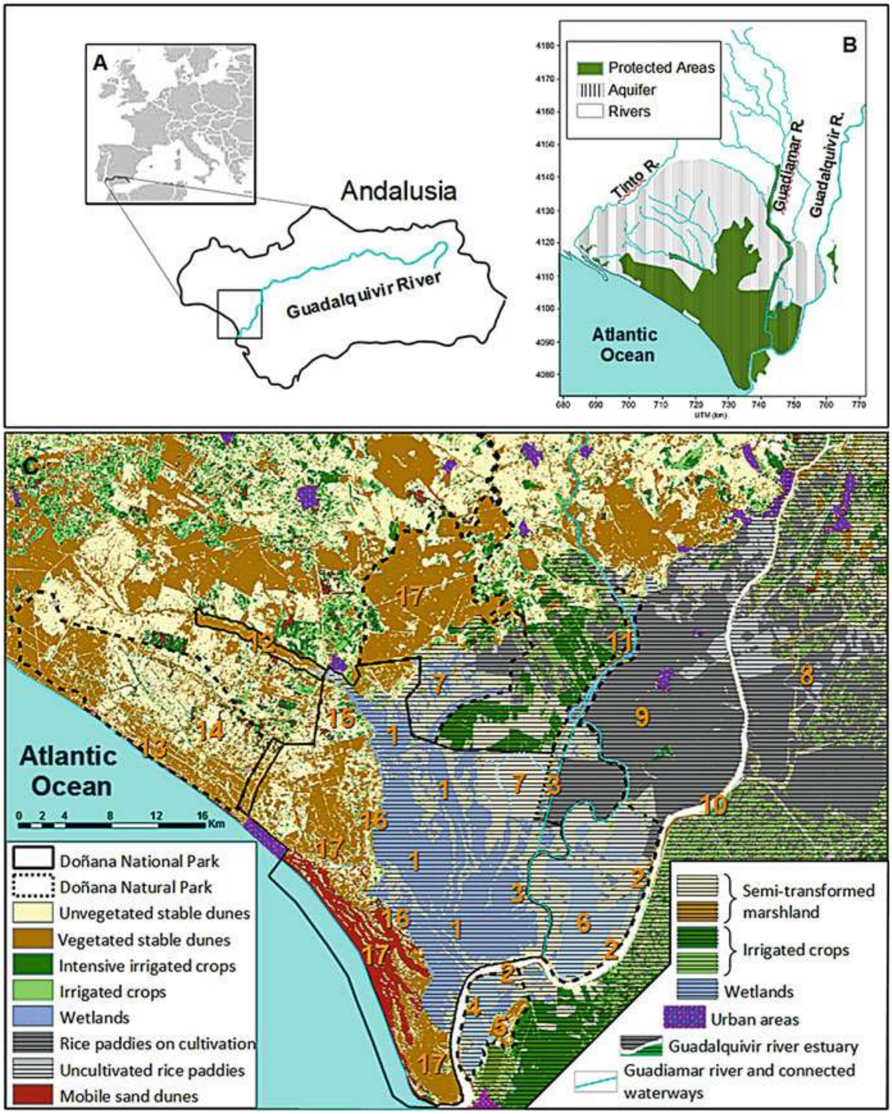

Fig. 1 (**a**) Location of the Doñana region in Europe and Andalusia. (**b**) Protected areas, main rivers, and the aquifer of the Doñana region. (**c**) Land use and natural habitats (Redrawn from Fernández et al. 2010). The horizontal striped area represents the silt-clay deposits of the marshland. Main aquatic systems: freshwater marsh (*1*), tidal marsh (*2*), freshwater marsh with tidal influence (*3*), salt pans (*4*), Tarelo lagoon (*5*), aquaculture farm (*6*), restored freshwater marsh (*7*), Brazo del Este old riverbed (*8*), rice fields (*9*), the Guadalquivir Estuary (*10*), Guadiamar river and Entremuros Canal (*11*), La Rocina stream (*12*), El Asperillo ravines (*13*), Rivatehilos peat bogs (*14*), dune creeks (*15*), dune-marsh ecotone (*16*), dune pond network (*17*) (Reprinted from Remote Sensing of Environment, 114, Néstor Fernández, José M. Paruelo, Miguel Delibes, Ecosystem functioning of protected and altered Mediterranean environments: A remote sensing classification in Doñana, Spain, 211–220, Copyright (2010), with permission from Elsevier)

This high species richness is supported by a wide variety of habitats. While some of them are relic habitats, others have been created by man-made changes in the landscape or are supported by traditional uses of natural resources. The map that relates biodiversity and land use in the whole Doñana region is yet to be made though the ecosystem functional types proposed by Fernández et al. (2010) provides an excellent start point (Fig. 1). Wetlands are located on both protected and nonprotected areas, as well as on the two main geomorphological units, the silty clay floodplain (marshland), and the aeolian sand systems (mobile and stable dunes), though wetland hydrology and main limnological features are distinct in each unit (Espinar and Serrano 2009). The central plain of the Doñana marshland is flooded seasonally by freshwater (Fig. 1, aquatic system type 1), direct rainfall, the water flow of several streams plus water diverted from the Guadiamar river on the northern side and, to a much lesser extent, groundwater discharge from the fringing dunes on its western edge. The tidal marsh (type 2) is presently restricted to a narrow fringe along the banks of the Guadalquivir river due to a long levee that isolates the freshwater marsh within the National Park from the river estuary (with the exception of a small areas close to the floodgates during heavy tides). Tidal flow also moves upward along a dead arm of the Guadalquivir river which is presently connected to the ending channel of the Guadiamar river (type 11) and eventually reaches the eastern edge of the freshwater marsh at several locations, thus creating several areas fed by a combination of fresh and tidal water (type 3). Both water sources do not necessarily mix inside these areas as freshwater dominates during the rainy season while the heaviest tidal surge occurs in mid-September. The freshwater outlet of the rice paddies, which also drains to the same dead river arm during October, increases the freshwater flow to the marsh and helps to reduce the salinity of the estuarine water. As a result, this type of marsh benefits from a moderate conductivity range (1.6–8.0 mS cm^{-1}) while it receives many colonizing organisms from several sources resulting in a total of 224 phytoplankton taxa in some localities (Reyes et al. 2008). Salt pans operated by private companies within the natural park provide hypersaline environments (type 4). An artificial gravel pit that has subsequently become undated and provides a permanent lagoon, the Tarelo lagoon, is fed by nutrient-rich groundwater seepage from the Guadalquivir river estuary (type 5). The most remarkable artificial wetland area is an aquaculture farm of 3200 ha (type 6) where tidal water from the Guadalquivir Estuary is first decanted to get rid of the excess in inorganic suspended solids, then treated and recirculated through 45 shallow earthen ponds of 70 ha each by means of a complex network of canals. Shrimp (*Palaemonetes varians*) and some native fish species are the main commercial products while a fifth of the annual production is estimated to be supporting up to 300,000 waterbirds that feed here when the natural marsh is dry (Klowskoski et al. 2009). This farm is located inside a private estate of 11,300 ha where over 200 bird species have been recorded. The Doñana 2005 Regeneration Project has, so far, two successful stories of freshwater marsh restoration, including newly created wetlands (type 7), while an old riverbed of the Guadalquivir river, which was protected in 1989 (type 8), has not been restored to the extent of preventing its classification as a heavily modified water body under the Water Framework Directive. The former vast extension of the Guadalquivir marshland has been drained and transformed during the past century in order to boost irrigation crops. About 35,000 ha was designated for growing rice though the actual surface area under cultivation varies from year to year. Nowadays, rice fields are irrigated by freshwater pumped from the Guadalquivir river in late spring, recirculated through multiple canals during the growing season, and finally discharged into the estuary before the rice is harvested in late September. Additionally, the introduced red-swamp crayfish (*Procambarus clarkii*) is also harvested from the canals as the rice paddies are gradually drained. Rice fields (type 9) are considered the most important crop for waterbirds worldwide, and Doñana is no exception. Toral et al. (2012) suggested the designation of these rice fields as a special protection area for birds under the EU Directive on the conservation of wild birds. The Guadalquivir Estuary (type 10), stretching from the ocean to the first dam 110 km upstream, is of huge importance for the Doñana water resources (six of the aforementioned wetland types depend on it) as well as for the fisheries in the region as a nursery ground (Martín-López et al. 2011).

On the aeolian sands, streams, creeks, springs, and wetlands are common features during heavy rainy periods due to both the rise of a shallow water-table and the regional discharge of the aquifer to the South and East, that is, towards the coast-line and the border between the sands and the marshland (dune-marsh ecotone) where dune creeks (type 15) and wet meadows (type 16) are formed. The catchment of La Rocina stream (type 12) is only partially protected but together with the springs and ravines of the Asperillo cliff (type 13) provide refuge to Atlantic hygrophytic vegetation. Wet heathlands of *Erica ciliaris* occupy the southern limit of their distribution in the peat bogs of Rivatehilos (type 14, Sousa et al. 2012). Amid the dunes, depressions of any size and depth are eventually flooded, sometimes after decades, when the water-table rises over the surface during heavy rainy periods, shifting their ecology from a more terrestrial to a more aquatic functioning, such as in the wet-slacks of the mobile dunes and the ephemeral ponds on the stable dunes. Unlike the coastal lagoons, the Doñana dune ponds (type 17) have no surface or groundwater connection to the sea though they have some oceanic influence from airborne sea salt deposition. Collectively, this network of groundwater-fed ponds can gather over 3,000 temporary water bodies of extraordinary importance for amphibians, macroinvertebrates, and aquatic macrophytes (Díaz-Paniagua et al. 2010).

Hydrological Regime and Timescale

As much as biodiversity is supported by a mosaic of landscapes and habitats in Doñana, the key to their functioning lies on their ability to shift at several timescales. The description of the Doñana marshland makes two contrasting landscapes that hardly relate to one another, and yet they are the same ecosystem at two different seasons: the scorching dry marshland versus the vast wet floodplain. The degree of persistence of the Doñana wetlands spans from days to decades as the system is driven by the uncertainty of the Mediterranean climate regarding the total annual rainfall. Even a partial desiccation exerts a significant influence on suspended and dissolved substances that the limnology of those wetlands too large or too deep to desiccate completely every summer cannot be fully understood without understanding their hydrological fluctuations. Therefore, simple hydrological features such as the duration of flooding and the variation of water depth can bring about wide changes of ionic, nutrient, and planktonic chlorophyll concentrations in the Doñana wetlands (Espinar and Serrano 2009). In the long term, the rhythm of drought and flood cycles give these temporary wetlands the chance to change widely and thus support a high number of species over time, from typically freshwater assemblages to euryhaline and even some hypersaline species in the same water body (Fahd et al. 2009). Complete desiccation promotes the mineralization by sunlight of organic matter deposited on the sediment during the aquatic phase and thus contributes to reduce the load of primary producers that otherwise could restart their growth during the mild temperature and humid conditions of the following autumn. Strong floods, on the other hand, produce a significant flushing effect that depletes primary production and reverts the system to an earlier successional stage dominated by filamentous algae during winter and by submerged macrophytes during spring in some dune ponds (Toja et al. 1991 in Serrano et al. 2006).

In order to preserve these Mediterranean temporary aquatic systems, it is essential to maintain a fluctuating disturbance regime between the two hydrological extremes of drought and flood cycles. Conservation management has traditionally extended the duration of flooding in the marshland to favor bird populations though the floodgates are eventually open and the marshland inside the national park is drained before summer to avert the risk of bird diseases associated with stagnant water. This management, however, has increased soil deposition and eutrophication in the Doñana marshland (Serrano et al. 2006), but it could be balanced by a flushing effect if the floodgates remained open during extremely heavy floods. On the sand dunes, the use of ground water for urban supply and irrigated crops, particularly strawberries and other berries, is supposedly balanced with the infiltration rate of the aquifer, but the recent tendency towards desiccation of dune ponds contradicts those calculations (Serrano and Zunzunegui 2008).

Future Challenges

If only all Spanish authorities, at a local, regional, and national levels, could commit themselves to the full enforcement of the law, the future of the Doñana region would be brighter. Mismanagement has resulted in widespread illegal groundwater abstraction, inoperative waste-water treatments plants to run for years, and a golf-course to open. The challenge is how to move on from past poor management practices amid a wider economic crisis and energy crisis. Mega-schemes for unsustainable development projects are constantly being proposed inside and around these protected areas (tourist megalopolis, highways, oil and gas pipelines, a water reservoir, and an underground gas reservoir). More effort, instead, should be towards understanding the value of the ecosystem services in the region in order to promote the sustainable use of natural resources in the long run.

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Groundwater Dependent Wetlands

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Ray Froend and Pierre Horwitz

Contents

Abstract

Groundwater-dependent wetlands (GDW) are no different to other wetlands in their need for management particularly under circumstances where hydrological changes threaten the conservation of wetland values. However, GDW have two important characteristics that make their management challenging. They derive a significant proportion of their annual inflow from hydrological pathways obscured by subterranean geology and geomorphology, and therefore understanding their response to altered groundwater regimes can be perceptually difficult. This same context creates a spatial and temporal "disconnect," where delays and thresholds need to be understood before cause and effect can be established. Accordingly, GDW are best approached from a starting point of complexity and uncertainty using management frameworks appropriate for the task.

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Keywords
Groundwater dependent wetlands · Conceptual models · Groundwater · Adaptive approach · Social learning processes · Monitoring

What Is Unique About GDW Management?

Groundwater-dependent wetlands (GDW) are no different to other wetlands in their need for management particularly under circumstances where hydrological changes threaten the conservation of wetland values. However, GDW have two important characteristics that make their management challenging. They derive a significant proportion of their annual inflow from hydrological pathways obscured by subterranean geology and geomorphology, and therefore understanding their response to altered groundwater regimes can be perceptually difficult. This same context creates a spatial and temporal "disconnect," where delays and thresholds need to be understood before cause and effect can be established. Accordingly GDW are best approached from a starting point of complexity and uncertainty using management frameworks appropriate for the task.

Questions arise about the management of GDW in the context of nearby/underlying (assumed) aquifers that may be used for water supply or are impacted by land use change. In cases where changes to groundwater inflow can be controlled (at least in principle), such as through abstraction from aquifers supporting GDWs or land use changes impacting aquifer recharge, adaptive management is the favored approach (Peterson 2005). However, in situations where there is neither control nor certainty, approaches such as scenario planning and resilience-building are more appropriate.

Integrated Management and Social Learning Processes

Choosing the most appropriate process to plan or implement an intervention in some instances will be at least as important as the desired outcome of the intervention itself (Horwitz et al. 2015). For example, plans aimed at reducing groundwater abstraction will require the appropriate participation of different users within local communities during the planning and implementation phases. Falkenmark and Folke (2002) emphasize the importance of social learning and therefore the roles of participation, empowerment, communication and education in water-related matters, and directing attention away from seeing these matters as merely technical issues. Ivey et al. (2004) propose five questions that will help elucidate a community's capacity to deal with these matters (specifically climate-induced water shortages):

- Are community stakeholders aware of the potential impacts of water shortages on human and ecological systems?
- Are local water management agencies perceived by community stakeholders as legitimate?

- Do local water management agencies and related organizations communicate, share information, and coordinate their activities?
- Is there an agency providing leadership to local water management organizations?
- Are members of the public involved in water management decision-making and implementation of activities?

Since surface and groundwater interactions are involved in the hydrological maintenance of the GDW, and both have "catchments" involving a range of different sectors with different perspectives, interests, and public mandates, an integrated catchment management approach is required. This should ensure that each sector can be engaged in the process of understanding, and deciding, where the trade-offs will need to be made. The most obvious case is where groundwater abstraction for water supply is considered a potential threat to the GDW, where water utilities, water resource management, environmental protection, and local and regional government, at least, will all need to be involved (MacFarlane et al. 2012).

An adaptive approach, aimed at building better understanding of the ecological importance of groundwater to a wetland, will continually seek to improve knowledge of the connectivity between wetland and groundwater, and the aquifers in question, through a social learning process (Holling 1978). This would include a participatory process to document and understand the different users of the groundwater and the trade-offs that are made (Horwitz et al. 2015) for each management scenario.

Conceptualizing the Relevance of Groundwater to the Wetland

Conceptual models of hydrological connectivity (surface and groundwater) can be applied as tools for identifying knowledge gaps and uncertainties. They can also be helpful in demonstrating the relative importance of groundwater as a hydrological input to the wetland (and also as a possible output or throughflow) (for instance, see Lloyd et al. 1993). Models can be used to formalize current understanding of the spatial and temporal connectivity between groundwater and the wetland. Furthermore, a model will assist with understanding the complexity (Gentile et al. 2001; Ogden et al. 2005; Richardson et al. 2011) by describing the wetland's hydrology and hydrogeology and biotic and abiotic components and processes. Hydrogeological processes including aquifer-to-aquifer interactions and surface water-to-groundwater interactions should be considered, including recharge, discharge, and storage processes and mixing and direction of groundwater flow. Although interactions between aquifers can be difficult to describe where data are limited, it is important to recognize the uncertainty. Similarly, understanding the spatial and temporal patterns of groundwater processes (seasonal and interannual) and how they relate to the ecology of the system are important aspects of hydrology-ecology interactions. These linkages should be described in a way that emphasizes the characteristics of the groundwater regime that supports the GDW. A critical groundwater service may include water provision for habitat or use, artesian (or other) pressure, thermal water supply, nutrient supply, or some other modifier of water quality critical to ecosystem function. In data-limited environments it is important that any assumptions made in the development of a conceptual model must be stated, along with an indication of the degree of uncertainty around these assumptions.

Measurement and Monitoring GDW Ecological Processes

Monitoring and evaluation, as an integral part of an adaptive management process, will build upon conceptual models for GDWs. Measurement programs can be used to provide a sound scientific basis for developing our understanding of the wetland system in question and informing proactive management objectives or responding to reactive changes in the wetland (Lyons et al. 2008). The ideal understanding of GDW response to altered groundwater flow would be based on quantified and validated relationships between the ecosystem and the groundwater source in question. It should be noted that, unless there is evidence to suggest otherwise, it is safer to assume the aquifer used for water supply, or affected by land use change within the catchment of the wetland, is the same as (or hydraulically connected to) the groundwater source important in maintaining the GDW. Through assessment and monitoring, these assumptions can be supported or disproved and contribute toward revising the conceptualization of interactions over time.

Each monitoring program will differ in their objectives and the components and processes of interest; however, a common requirement in GDW monitoring is the need to understand the connectivity and importance of groundwater (quality and quantity) to the wetland. Ideally, monitoring to achieve this will occur before planned impacts on the groundwater resource take place. However, in many cases this is not possible as groundwater inflow is already impacted by multiple stressors, currently and historically. Under these circumstances it is likely that there will be high uncertainty regarding the groundwater-wetland interaction. It is therefore important to have an adaptive management process in place that incorporates knowledge from biophysical monitoring into an improved conceptual model of the GDW. Furthermore, the process of learning should involve all sectors to ensure mutual understanding of the system and the trade-offs involved with changes to groundwater inflow.

For the purposes of assessing the level of dependency of a wetland on groundwater and how the ecosystem responds to changes in the groundwater system, long-term monitoring is very valuable (Parsons et al. 2011). Well-designed monitoring programs and hypothesis testing also add to the broader scientific knowledge base, which assists in the revision of conceptual models developed within the adaptive and consultative management process.

Adapting to Groundwater Change

A significant challenge in the management of GDWs is found in systems that are undergoing a hydrological transition (Kløve et al. 2011). In these scenarios, certainty in groundwater interaction with wetlands tends to be poor, and control over stressors impacting aquifers may be low (or at least varied), particularly where climate-related hydrological change is suspected (MacFarlane et al. 2012). Often, there is uncertainty regarding the relative importance of different (if recognized) stressors in altering groundwater inflow to wetlands.

In situations where groundwater inflow to a GDW is predicted to vary over the long term (e.g., rainfall reduction due to climate change or catchment land use change reducing recharge rates), questions are asked about the likely ecological responses to reduced or increased groundwater availability over time. Furthermore, it is unlikely there are sufficient data on the nature of future groundwater interactions to give accurate estimates of hydrological let alone ecological responses. The use of groundwater models to develop predictions on levels and potential inflow can be of benefit to the management process; however, it is important to maintain transparency of measurement error and integration of uncertainty when using modeled scenarios. Furthermore, all models should be subject to a process of continuous improvement based on outcomes from monitoring and research.

Adapting to changing groundwater conditions will require the regular revision of resource management objectives to determine if they are realistic under the altered groundwater regime. This should include assessment of previous hydrological, ecological, and social predictions and assumptions on which management objectives are based. In cases where changes to the groundwater resource have or are predicted to exceed management criteria for acceptable impacts or trade-offs, consideration should be given to the feasibility of supplementing (or replacing) groundwater use with an alternative source, subject to appropriate analysis of impacts.

Some GDWs are more susceptible to climatic variability and change than others and will require different management responses (Richardson et al. 2011). Aquifers with a low storage-to-recharge ratio (e.g., local groundwater flow system) are more hydrologically responsive to recharge reduction processes such as climatic variability. Although the biotic components of these hydrologically less robust systems will more likely have adaptive strategies to deal with natural variability (Shafroth et al. 2000), the novel environment created by the hydrological change may be outside the known response capacity of the wetland system. Questions should be raised about the suitability of such aquifers for development in light of the environmental trade-offs. Those wetlands associated with more robust aquifers (high storage-to-recharge ratio), and subject to low hydrological variability, are less likely to include biota with drought resilience traits. This can make these ecosystems more vulnerable to change over long periods even though the hydrogeological system supporting them is less likely to be affected by climatic variability.

Future Challenges

The cumulative pressures from increasing use of groundwater resources and reduced rainfall recharge due to climate change represent significant future challenges to managing GDW. Where declines in aquifer storage threaten the persistence of known GDW interactions, an integrated, risk-adverse approach to management is required. However, this is not always possible due to uncertainty in (or absence of) groundwater information/modeling on recharge and discharge processes or disagreement between stakeholders on management objectives and acceptable trade-offs. Developing trade-offs in favor of GDW conservation may be challenging where water for consumption becomes scarce as a consequence of increasing population and climate change. If threshold responses in GDW are to be avoided, integrated approaches will be required along with mitigation or avoidance of resource development practices that exacerbate climate or land-use-driven reduction in recharge.

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Managing Wetlands for Pollination

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Robert J. McInnes

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Abstract

Pollination is a fundamental process in plant biology whereby pollen is transferred from the anther (male part) to the stigma (female part) to facilitate fertilization and reproduction. Pollination is restricted to the flower bearing plants or angiosperms. Pollination can be mediated by abiotic and biotic factors. Approximately 87% of all flowering plants are pollinated by biotic vectors such as insects, birds, and mammals. The primary abiotic factor is pollination by the wind (known as anemophily). This form of pollination is common in many wetland grass species, numerous coniferous, and many deciduous trees. Some wetland and aquatic plants release and disperse their pollen directly into water and this becomes the vector for pollination, known as hydrophilous pollination.

Keywords

Pollination · Regulating service · Plant reproduction

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Introduction

Pollination is a fundamental process in plant biology whereby pollen is transferred from the anther (male part) to the stigma (female part) to facilitate fertilization and reproduction. Pollination is restricted to the flower bearing plants or angiosperms. Pollination can be mediated by abiotic and biotic factors. Approximately 87% of all flowering plants are pollinated by biotic vectors such as insects, birds, and mammals (Regan et al. 2015). The primary abiotic factor is pollination by the wind (known as anemophily). This form of pollination is common in many wetland grass species, numerous coniferous, and many deciduous trees. Some wetland and aquatic plants release and disperse their pollen directly into water and this becomes the vector for pollination, known as hydrophilous pollination (Cox 1988).

Hydrophilous pollination for angiosperms can be broadly categorized into two classes: (1) ephydrophily where pollination occurs at the water surface, common in Hydrocharitoideae, e.g., *Hydrocharis morsus-ranae* or common frogbit, and Stratiotoideae, e.g., water soldiers; and (2) hyphydrophily which occurs where flowers are completely submersed, e.g., *Zostera* or seagrasses. Hyphdrophily can be considered as a unique evolutionary departure from the normal pollination approach taken by the majority of terrestrial plants. This abiotic pollination system has evolved through structural and biochemical modifications of aerial pollination systems and the complete loss of aerial flowers in favor of a submersed approach to reproduction (Du and Wang 2014).

The Importance of Pollination

Pollination is recognized as an important ecosystem service. A variety of animal species play a key role in the pollination of commercial agricultural crops as well as wild plants. Almost three quarters of the main crops consumed by humans worldwide depend on insect pollinators, and the total global economic value of all wild and managed pollination services has been estimated to be US$215 billion at 2005 prices (Gallai et al. 2009). Therefore, pollinators play a crucial role in contributing to the availability of nutrition for humans and supporting overall well-being. However, despite the recognition in some circles of this important ecosystem service, a growing number of studies are showing that pollinators are declining across the globe. Evidence from Europe and North America suggest that stocks of pollinators are not sufficient to meet the demands of modern agricultural systems (Kremen and Ostfeld 2005).

Wetlands have a key role to play in pollination, not just through their importance in supporting hydrophilous species, but also through providing diverse habitats which support an abundance of pollinators. However, the loss, degradation, and mismanagement of wetlands have contributed to the overall reduction in pollinators across the world.

Threats to the Wetland Pollination Service

Globally wetlands are threatened resulting in loss of rare habitats and species, and consequently the pollination service they support. There are a variety of drivers behind this loss of service. It has been suggested that the decline in pollinating birds and mammals is primarily due to unsustainable agriculture, invasive species, and hunting. Similarly, declines in other important pollinators, such as bees and butterflies, have been attributed to habitat loss, fragmentation and degradation, pesticide use, and invasive species. For instance, wetland-specialist butterfly species have declined by approximately 15% across Europe in the 25-year period up until 2006 bringing with a concomitant loss of pollination function (van Swaay et al. 2006). Work undertaken on mammalian pollinators, principally bats, suggest that those on the IUCN Red List Index (www.iucnredlist.org) declined commensurate to a rate equating to an average of 1.4 species moving one Red List category closer to extinction each year. Many of these bat species, such as the Pemba flying fox *Pteropus voeltzkowi* which inhabits moist lowland forests and mangroves on the Island of Pemba off the Tanzania coast, rely on wetlands during part of their life cycle. The situation for pollinating bird species is similar with global declines observed for other taxa (Regan et al. 2015). Over 900 bird species pollinate plants (Whelan et al. 2008); many of these species are wetland-dependent for some of their life cycle. However, knowledge on their role as pollinators is limited, especially with regard to wetland-dependency.

A significant body of research has focussed on the role of bees as pollinators, especially given shortages in honeybee colonies in the United States during the early 2000s. Some bee species are wetland specialists, but many species including bumble bees (*Bombus*) and honeybees, utilize wetlands for both water and nectar. Often the diversity of habitats, especially within intensive agricultural landscapes, can strongly influence the stability and magnitude of pollination services and the associated diversity, abundance, and productivity of bees (Kremen and Ostfeld 2005). In Canada, it has been demonstrated that preserving uncultivated land, such as small or interconnected wetlands, increases agricultural yields per hectare by increasing pollinator habitat (and thus pollination) in canola fields.

The restoration of wetland habitats can also yield pollination benefits. The removal of invasive species, such as glossy buckthorn *Frangula alnus*, from degraded prairie fen in the Midwestern United States, has shown to increase pollinator abundance, diversity, and community structure on a rapid trajectory towards reference condition. The response of the pollinators is often more rapid than the plant community response and demonstrates the persistence and recovery potential of generalist pollinators within the landscape (Fielder et al. 2012). The successful management and restoration of coastal mangrove systems depends on pollination from bees and wasps. Studies conducted on *Ceriops decandra*, a near threatened mangrove species distributed along the east coast of India and Bangladesh, southwestern Thailand, and the western part of the Malay Peninsula, has highlighted the critical role played by *Nomia* bees and *Odynerus* wasps in pollination. Bees and wasps were observed frequently moving among the flowers of individual trees and between different trees enhancing cross-pollination (Raju et al. 2006).

Future Challenges

Pollination is known to be a critical ecosystem services which underpins the provision of agricultural systems and thereby supports human nutrition and health. Wetlands in the landscape are crucial to support both biotic and abiotic pollination processes. As the main repository for hydrophilous species, water within wetlands acts as the key vector for the pollination of a diversity of common and rare plants. Similarly, and potentially more importantly, wetlands add diversity to uniform industrial agricultural landscapes and support a variety of pollinators including bees, wasps, birds, and mammals.

Threats to both the physical structure and quality of wetlands persist. These threats bring commensurate impacts on the pollinators that wetlands support. Currently, the role of individual pollinating species or how wetland habitat structure and diversity contribute to supporting key pollinators is poorly understood and documented. Some studies have demonstrated the importance of restoring wetlands to assist in provision of pollinators but efforts are needed to understand better the role of pollinators in existing wetlands to ensure that the value of this important ecosystem services is understood.

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Managing Wetlands for Water Supply 161

Robert J. McInnes

Contents

Abstract

The Millennium Ecosystem Assessment emphasized that the provision of freshwater to humans is one of the most significant benefits derived from wetlands. Throughout the world, different types of wetlands play crucial roles in maintaining the supply of water to the majority of the world's population.

Keywords

Water supply · Surface water · Groundwater

Introduction

The Millennium Ecosystem Assessment emphasized that the provision of freshwater to humans is one of the most significant benefits derived from wetlands (Millennium Ecosystem Assessment 2005). Throughout the world, different types of wetlands play crucial roles in maintaining the supply of water to the majority of the world's

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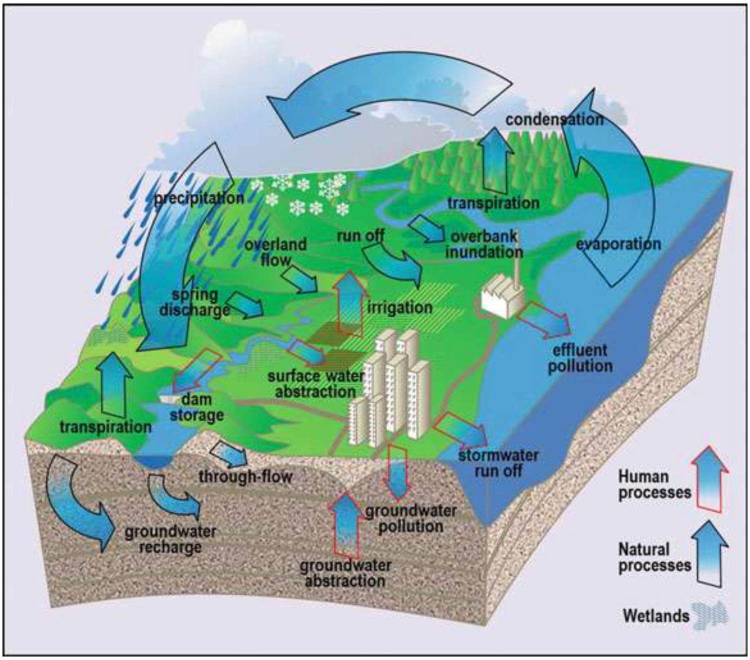

Fig. 1 Relationship between the water cycle and wetlands. © Robert J. McInnes shown as aquatic, subterranean, and coastal marine ecosystems (From Millennium Ecosystem Assessment 2005)

population. The role of wetlands in the water cycle and supply of freshwater for human purposes is shown in Fig. 1.

The water provided by these wetlands is used for a variety of purposes including domestic, agricultural, and industrial uses. Conversely, human modifications to the hydrological regimes of wetlands, through inter alia water abstraction and alteration to seasonal hydrodynamics, can have detrimental consequences for the integrity of wetland ecosystems. While the importance of water for producing food for people has been widely recognized, Falkenmark et al. (2007) have noted that in places the allocation of water for food production has gone too far with severe ecosystem degradation, including the loss of wetlands and the disruption of basic water supply for people. Much of the allocation of water for food and energy production has been facilitated through the construction of dams and reservoirs with the extent of water that has been intercepted and stored in reservoirs being shown in Fig. 2.

Insufficient water reaching wetlands, due to abstraction, storage, and diversion of water for public supply, agriculture, industry, and hydropower, is a significant cause of wetland loss and degradation. In addition, many wetlands have been polluted by both point and diffuse sources, including nutrients, trace metals, and a range of toxic substances (Millennium Ecosystem Assessment 2005; Falkenmark et al. 2007).

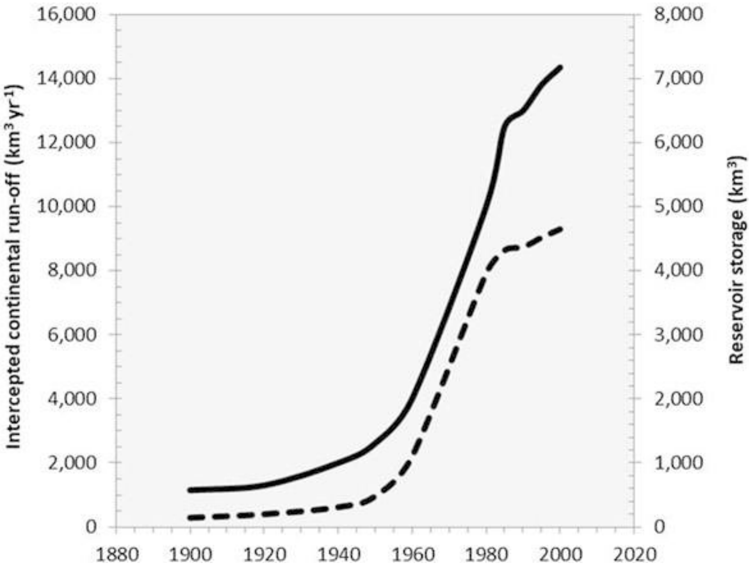

Fig. 2 Intercepted continental runoff (*solid line*) and reservoir storage (*dashed line*) (Data from various sources. © Robert J. McInnes)

Wetlands as Suppliers of Water

Wetlands provide two principle categories of benefits: (1) the supply of water for drinking, irrigation, or other abstractive uses and (2) the role of water in providing support for water-dependent species, such as fish and waterfowl, and the supply of non-extractive water-based benefits such as recreation, transportation, or flood control (adapted from Postel and Carpenter 1997). The focus of this article is on the role of wetlands in providing water supplies primarily for human domestic consumption.

There are two main reasons why wetlands are important as suppliers of water. Firstly, they are, as the name "wetland" implies, areas where the presence of water is essential either above or just below the ground surface. Consequently, in wetlands water is usually relatively accessible. In arid or semiarid areas dominated by distinct seasonality to rainfall patterns, the access to water in wetlands can be particularly important (McCartney and Acreman 2009). Secondly, there is a considerable body of evidence that demonstrates the effectiveness of wetlands in maintaining and improving the quality of water for human consumption and use (Verhoeven et al. 2006). This water cleaning function of wetlands has progressively increased in value as anthropogenic sources of pollution have degraded the quality of global water supplies.

Wetlands can be considered as sinks into which rainfall, surface water, and groundwater are delivered. By holding and retaining water, especially during dry periods, wetlands become important sources of water supply for human populations. However, the heterogeneity of wetland types, including inland, subterranean, and coastal/marine examples (see Fig. 1), which results from their unique topographical, hydrological, or geological properties, can make the understanding of the interactions with a wider hydrological cycle complex and difficult. This brings challenges for the management of wetlands for water supplies. Consequently, drawing generic conclusions on the hydrological functioning of wetlands may be far from simple especially where locally relevant data is absent (Bullock and Acreman 2003).

Surface Water Supplies

Many wetlands store surface water during wet periods, some through extensive networks of contributory streams, and slowly release it through seepage or distributary streams, during dry periods. Where there is an excess of water, the wetlands are essentially operating as reservoirs or as sponges that at least temporarily retain the water that has accumulated. This is well known for peat systems, such as those across the Tibetan Plateau that serve as a water tower for some of the largest rivers in Asia. Smaller wetlands also serve a similar purpose. For instance, coastal peatlands in Sarawak, East Malaysia, have been identified as playing an important role in contributing to the base flow of numerous streams and rivers. The large excess of rainfall over evapotranspiration maintains water levels at or close to the surface of the peatlands. Excess water collects in pools and depressions and coalesces into rivulets which feed the wider drainage network. It has been estimated that across Sarawak in excess of 3000 million liters of water is abstracted annually from the streams fed by these coastal wetlands (Mailvaganam 1994). In other instances water is accumulated in extensive inland lakes, swamps or marshes, and deltas.

Alterations to the natural hydrological functioning of wetland systems can impact both surface water supplies and wider ecological functioning. For instance, the Kafue Flats, Zambia, have historically been inundated from high flows in the Kafue River during April and May. The Flats would slowly release water back into the river through to October and November maintaining both the downstream domestic water and irrigated agriculture needs for some 1.3 million people (Mumba and Thompson 2005). However, the construction of hydroelectric generation dams on the Kafue River has altered significantly the hydrological dynamics of the Flats. In some areas downstream of the dams, the duration of inundation has been extended, changing the ecological conditions on the floodplain to ones that favor the spread of the alien invasive shrub *Mimosa pigra* which impacts the livelihoods of pastoralists and alters the habitat structure rendering it less valuable for threatened species such as the lechwe *Kobus leche kafuensis*. In other areas, flooding has been reduced with subsequent impacts on the availability of surface water (Mumba and

Thompson 2005). Therefore, changes to the surface water functioning of wetland systems can deliver impacts to both water supplies and the wider benefits provided to human society.

The extent and consequences of the alteration of natural flows of freshwater have been illustrated on many occasions with large volumes of water diverted for agriculture or stored behind dams (Fig. 1; Millennium Ecosystem Assessment 2005; Falkenmark et al. 2007). These assessments have also documented volume of water stored in reservoirs worldwide; from 1950 to 2000, the number of dams in the world increased from 5000 to more than 45,000, providing water for a mix of purposes, including domestic consumption, irrigated agriculture, and generation of electricity. Lehner et al. (2011) have developed a database of dams and associated reservoirs showing 6862 dams with a storage capacity of 6,197 km^3 and estimated that worldwide there were some 16.7 million reservoirs greater than 0.01 ha with a storage capacity of about 8070 km^3.

Groundwater Supplies

Groundwater represents the largest store of unfrozen freshwater on the planet, and it currently provides humans with the majority of their water supplies. For example, across Africa, 60% of the human population live in rural areas where they depend on small-scale groundwater supplies (Calow et al. 2010). The majority of groundwater is replenished by rainfall permeating slowly into the ground, and often through wetlands, over a period of many months. The regular and gradual replenishment of groundwater has the potential to provide water over several years and to moderate both seasonal and multi-year variations in rainfall and major droughts (Acreman 2012).

Groundwater recharge is a function often attributed to wetlands. However many wetlands are located either in topographic depressions or where impermeable substrates impede infiltration and the downward movement of water. Consequently, recharge rates in wetlands are often much slower than those in adjacent upland soils characterized by more permeable material (McCartney and Acreman 2009). However, in some circumstances wetlands play a crucial role in the recharge of groundwater. Hollis et al. (1993) demonstrated that by remaining inundated for a significant period, the floodplain wetlands of the Hadejia-Nguru system in northeast Nigeria played a significant role in recharging the groundwater aquifer and contributed to the water supply for approximately one million people living in the region. Smaller-scale recharge and groundwater have also demonstrated the other wetland systems with important ecological outcomes and support for local livelihoods. For instance, van der Kamp and Hayashi (1998) demonstrated that small wetlands in the semiarid northern prairie region of the United States were focal points for groundwater recharge. While the majority of the water entering these systems was observed to maintain high evapotranspiration rates and only a small proportion recharged the regional aquifer, this relatively small contribution was still significant in sustaining groundwater resources. Similarly, in arid regions, or areas with distinct seasonal rainfall patterns, the retention of soil water or water in shallow superficial deposits underlying wetlands can provide a valuable water resource essential to the well-being of many of people. Valley-bottom wetlands in Africa, which are locally known as dambos, are a good illustration of this phenomenon (Bullock 1992). The dambos retain water table at or close to the surface well into the dry season which contributes to an extended growing season, beneficial to cultivators and pastoralists alike, as well as providing local water supplies through shallow, hand-dug wells (Woodhouse 2003).

Future Challenges

The demand for water resources is growing inexorably as the human population increases and climate change threatens existing water supplies (Vorosmarty et al. 2000). In many parts of the world, the different demands on water resources, such as industrial uses, the need to grow food, and the domestic requirements of an ever-increasing human population, have led to rivers running dry for parts of the year and groundwater resources being depleted at rates in excess of the natural recharge (Hoekstra et al. 2012). As human demands on water resources expand, impacts upon wetlands are common. The alteration of components of the hydrological cycle, through the removal, capture, or alteration in flow regimes, can have significant negative impacts on wetlands (McCartney and Acreman 2009). Impacts affect both groundwater and surface water elements. Therefore wetlands need to be protected from unsustainable approaches to water resource development.

Similarly, wetlands, including streams and floodplains, have a role to play in maintaining both the quantity and quality of water resources. Consideration is being given to the designation of wetlands, river catchments, and areas of groundwater recharge as protected areas in order to support and sustain the future water supplies (Pittock et al. 2008). Hydrological modeling is increasingly being used in water resource management planning and in understanding their role in protecting water supplies. There is a need to ensure that wetlands are considered appropriately within such models in order both to protect them from degradation and hydrological impacts and to optimize the benefits which wetlands can deliver (Dickens et al. 2003). At the same time, the balance between the construction of dams and reservoirs and the maintenance and use of wetlands needs to be addressed and as raised by Falkenmark et al. (2007) avoid the costs of going too far while ensuring a freshwater supply for human well-being.

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Climate Regulation and Wetlands: Overview

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Robert J. McInnes

Contents

Abstract

One of the most important ecosystem services which links wetlands to human well-being is the regulation of climate. Through the storage and sequestration of carbon, wetlands play a significant role in global carbon cycles. However, wetlands can also act as a source and a sink for greenhouse gases and they can influence local and regional temperature, precipitation and other weather patterns. Increasingly, as a result of their important role in climate regulation, wetland management and restoration activities are being integrated into local, national and international programmes which aim to both mitigate and adapt to climate change.

Keywords

Adaptation · Climate regulation · Carbon cycles · Temperature control · Climate change mitigation

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Introduction

The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) highlighted the range of benefits that wetland ecosystems provide human society. One of the most important services, which links wetlands to human well-being, is the regulation of climate. However, the loss and degradation of wetlands, and the resultant impact on stored carbon can be a major contributor to greenhouse gas (GHG) emissions and atmospheric warming (Crooks et al. 2011).

The role of wetlands in storing and sequestering carbon is well established in the literature (Gorham 1991; Bridgham et al. 2006). However wetlands play a wider role in the regulation of climate beyond simply that of carbon dynamics. Along with their very important role in global carbon budgets, wetlands can act as a source and a sink for greenhouse gases and they can influence local and regional temperature, precipitation and other weather patterns. Increasingly, as a result of their important role in climate regulation, wetland management and restoration activities are being integrated into local, national and international programmes which aim to both mitigate and adapt to climate change (Erwin 2009).

Wetlands and Carbon

The presence of water at or near the soil surface of wetlands, resulting in extended periods of waterlogging, generates anoxic conditions which are optimal for the sequestering and storing of carbon from the atmosphere (Mitsch et al. 2013). Whilst much attention has been paid to peatlands, the unique soil properties associated with many wetland types means that globally many wetlands act as major carbon sinks. Despite the fact that estimates suggest that wetlands cover less than 8% of the earth's land surface they may contain 30% of the total global soil carbon. It has been estimated that a loss of just 1.6% of global peat would equate to the total global anthropogenic GHG emissions (Joosten 2009).

Vegetated coastal wetlands, including seagrass beds and mangroves are significant carbon stores from the equator to the poles. Seagrass beds occur in cold polar waters through to the tropics, whereas mangroves are confined to tropical and sub-tropical areas. Tidal marshes are found in all regions, but are most common along temperate coasts. It has been estimated that the combined area of these ecosystems covers approximately 49 million hectares (Pendleton et al. 2012). Mangroves are recognized as among the most carbon-rich forested wetland ecosystems with the prevailing physical and biogeochemical conditions being highly conducive to long-term carbon retention. Estimates suggest that mangroves contain on average in excess of 1 Mg carbon per hectare with 49–98% of the carbon stored in organic-rich soils (Donato et al. 2011).

Seagrass beds and tidal marshes often reside over organic-rich sediments that may be several meters deep and effectively 'lock up' carbon due to low-oxygen conditions and other factors that inhibit decomposition (Pendleton et al. 2012).

Due to both the trapping of allochthonous carbon and autochthonous production in seagrass beds it has been estimated that carbon burial rates may be between 45 and 190 gCm^{-2} $year^{-1}$ (data cited in Mcleod et al. 2011). Globe estimates suggest that seagrass beds may store as much as 19.9 Gt organic carbon making these systems significant carbon stores (Fourqurean et al. 2012). Similarly, the rates of gross primary production and carbon accumulation in tidal salt marshes can be considered exceptionally (Connor et al. 2001) with estimates suggesting that salt marshes accumulate carbon at rates in excess of 218 gCm^{-2} $year^{-1}$ (Duarte et al. 2005).

Significant interest has been paid to the role of tropical peat swamp forest in global climate budgets. It has been estimated that as much as 88.5 Gt of carbon (range 81.5–91.8 GtC), equal to 17–19% of the global peat carbon pool, is stored in tropical peatlands (Page et al. 2011). Forested peatlands of south east Asia are widely recognised as representing significant carbon stores, for instance in Central Kalimantan, South Sumatra and West Papua. The total carbon storage of Indonesian peatlands has been estimated to be in the region of 42 GtC and may be as much as 55 ± 10 GtC (Jaenicke et al. 2008).

Other wetland types can also be significant stores of carbon in both their soils and standing vegetation. For instance, the organic carbon pool of China's wetlands, such as *Larix gmelinii* and *Rhizophora stylosa* mangrove systems, andnon-forested communities dominated by*Phragmites australis*, *Glyceria acutiflora* and *Scirpus mariqueter*, has been estimated to be between 5.39 and 7.25 Gt, accounting for 1.3–3.5% of the global carbon pool (Zheng et al. 2013). In the UK, upland peatland systems dominated by *Sphagnum* species, have been estimated to contain over 3.2 Gt of carbon (Worrall et al. 2010), approximately twenty times that of UK forests. In North America it has been estimated that the prairie pothole region may have supported over 20 million ha of wetlands prior to European settlement and their consequential conversion to agriculture may have resulted in a loss of 10.1 Mg ha^{-1} of soil organic carbon (Euliss et al. 2006).

Wetlands and Greenhouse Gases

GHGs are naturally occurring components of the atmosphere, which are necessary for maintaining a habitable climate for life on Earth. However, due to their ability to both absorb and release infrared radiation, GHGs can trap the sun's warmth within the Earth's atmosphere resulting in changes in atmospheric gaseous concentrations. This process is widely held to be responsible for the gradual warming and changing of the Earth's climate.

In terms of the warming potential observed in Earth's atmosphere, the most significant GHGs are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), with other common contributors being water vapour and the anthropogenically generated hydrofluorocarbons and perfluorocarbons. Although much attention has been paid to CO_2 in the popular media, CH_4 has the potential for global warming per molecule of CO_2 of 25:1, while N_2O has a global warming potential of 298:1 that of CO_2 when considered over one hundred years (Forster et al. 2007).

All wetlands can sequester carbon and are a source of GHGs. The rates of generation and emission of GHGs will differ from wetland to wetland depending on the prevailing biogeochemical processes, which also regulate the overall carbon balance. However, these biogeochemical processes are strongly influenced by a variety of factors including land use, vegetation type and management, soil organisms, soil properties, hydrogeomorphology and climate.

At any wetland the major control on GHG emissions will be water table elevation and duration of waterlogging as well as nutrient availability and the prevailing climate. Under aerobic soil conditions organic matter decomposition releases CO_2, and atmospheric CH_4 can be oxidized in the surface soil layer (Trettin et al. 2006). In contrast, in the anaerobic soils which dominate wetlands, CH_4 can be generated in addition to CO_2. Consequently, wetlands can be an inherent source of CH_4.

N_2O emissions from wetlands are typically low. However, wetlands can be a source of N_2O emissions where there are significant inputs of nitrogen from adjoining uplands or from leaching into groundwater from agricultural operations. While there can be significant internal cycling of nitrogen within wetland soils, nitrogen can be returned to the atmosphere via denitrification, and also during nitrification as ammonium is transformed to nitrate. Denitrification can be defined as the microbial reduction of nitrate or nitrite to gaseous nitrogen either as molecular nitrogen or as an oxide of nitrogen. For denitrification to occur requires that nitrate (NO_3^-) is present, that denitrifying bacteria have a carbon source for energy and that no oxygen (O_2) is present to preclude microbes using O_2 as an electron acceptor.

As a result of the prevailing anaerobic conditions, wetlands inherently accumulate carbon in the soils and they are natural sources of CO_2 and CH_4 to the atmosphere. However, wetland management practices can influence both the pools of carbon and the fluxes of GHGs. There can also be a trade off in CH_4 and N_2O emissions through water level management practices. Wetlands that retain a high water level for the majority of the year, perpetuating anaerobic conditions, will tend to generate higher CH_4 emissions and lower N_2O emissions. This is due to the fact that the degree of soil waterlogging suppresses inherent nitrification and denitrification leads to the production of N_2 gas. However, a fluctuating water table, generating periods of anaerobic and aerobic soil conditions, will tend to enhance N_2O emissions and reducing CH_4 emissions, especially if there is a source of nitrogen available. Similarly, an intensification of such fluctuating water levels may also lead to more persistent aerobic conditions and the oxidisation of stored carbon and its subsequent return to the atmosphere.

The potential risk of increasing CH_4 emissions has been used as an argument against the restoration of wetlands (Mitsch et al. 2013). However, understanding the potential GHG fluxes is important to prevent undesirable outcomes. For instance, the reflooding of former drained agricultural peatlands may result in increased CH_4 emissions consequently management of water levels, the type of vegetation established and potentially the need to remove topsoil should all be considered to minimise the global warming potential of restoration measures (Zak et al. 2014). Similarly, the manipulation of water tables to restore wet grasslands and to create habitat for waterfowl may generate periods of soil aeration and saturation influencing emission rates of CH_4 and N_2O, as well as changing the carbon stock in the soil and other wetland vegetation.

Climate Warming and Wetland Carbon

There is a strong correlation between climate and the amount of carbon stored in wetland soils. It has been demonstrated that the organic carbon content of soils decreases with increasing temperatures as a result of decomposition rates doubling with every 10 °C increase in temperature. If global temperatures continue to increase, a comparable increase in the rates of decomposition of organic matter would prevail, and wetlands currently supporting anaerobic soils could subsequently become major emitters of CH_4. Whilst considerable attention has been paid to the impact of a warming climate on high latitude wetlands, and especially those in the tundra which may experience a thawing of soils and reduction in waterlogging, there is evidence that increased temperatures would also have a negative impact on GHG emissions from tropical peatlands (Kayranli et al. 2010).

Wetlands and Local Climate Regulation

Wetlands undoubtedly play a significant role in global carbon budgets and GHG emissions. However, wetlands also influence climate at a local level through fluxes, and the dissipation, of energy. While photosynthesis transforms and dissipates incident solar radiation, evapotranspiration, which is the sum of evaporation and transpiration from soil and vegetation surfaces, utilises far greater amounts of solar energy. Over millennia, plants have evolved mechanisms to control the rate of transpiration through their stomata and have developed adaptations to conserve water in times of drought-stress.

In wetlands, due to the common presence of water, energy input is transformed into latent heat of evaporation whereas on dry sites solar energy is converted into sensible heat, significantly increasing the local air temperatures. It has been estimated that evapotranspiration may consume in excess of 400 W/m^2. A comparison between a wetland and non-wetland area suggests that the difference in the distribution of solar energy between the latent heat of evaporation and sensible heat amounts to several hundreds of W/m^2 on a warm, sunny summer day (Pokorný et al. 2010). The localised cooling effect of wetlands in urban environments has been estimated to reduce temperatures by up to 8 °C in comparison to surrounding areas (Taha 1997) which has led to the recommendation that wetlands should be integrated into urban designs to reduce summer temperatures in built up areas (Sun et al. 2012).

Future Challenges

Climate change, and especially increasing temperatures, will impact on wetlands altering their role in global and local climate regulation. Effects of a changing climate will generate impacts to wetlands, but, conversely, the management and restoration of wetlands can assist in reducing the impacts of a changing climate. The future challenge is to understand the gaseous fluxes and the global warming potential of wetlands at a range of scales and to ensure that appropriate wetland management delivers on broader climate change mitigation and adaptation agendas.

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Weather, Climate, and Wetlands: Understanding the Terms and Definitions

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Jan Pokorný and Hanna Huryna

Contents

Abstract

The interactions between wetlands and the hydrological cycle are well known with increasing attention being focused on environmental flows and the links between surface and ground water. The relationships between the climate and the water regime in wetlands has also been increasingly investigated, including from a methodological side given the uncertainty and variability associated with many past measurements. As there is less clarity about the effect of weather and climate, these terms are explained below within the context of global climate change and the role of wetlands. The chapter provides terms and definitions to help clarify the terminology used to describe interplay between atmosphere and land surface. Main fluxes of solar energy in wetlands and on dry land surface are outlined.

Keywords

Climate · Radiative forcing · Albedo · Energy balance · Evapotranspiration

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Introduction

The interactions between wetlands and the hydrological cycle are well known with increasing attention being focused on environmental flows and the links between surface and ground water. The relationships between the climate and the water regime in wetlands has also been increasingly investigated, including from a methodological side given the uncertainty and variability associated with many past measurements. As there is less clarity about the effect of weather and climate, these terms are explained below within the context of global climate change and the role of wetlands.

Weather – concerns the conditions of the atmosphere prevailing during any particular time and place. It is often referred to by such terms as temperature, humidity, wind velocity, precipitation, barometric pressure, and cloudiness. It is the day-to-day state of the atmosphere, and its short-term variation is minutes to weeks. Weather on Earth occurs primarily in the troposphere, or lower atmosphere, and is driven by energy from the Sun and the rotation of the Earth (The American Heritage Dictionary of the English Language 2011).

Climate – in a narrow sense is usually defined as the average weather conditions of a certain region, including temperature, rainfall, and wind, or more rigorously, as the statistical description in terms of the mean and variability of relevant quantities over a period of time ranging from months to thousands or millions of years. Climate, therefore, represents the accumulation of daily and seasonal weather events over a long period of time. The classical period is 30 years, as defined by the World Meteorological Organization (WMO). Climate in a wider sense is the state, including a statistical description, of the climate system. On Earth, climate is most affected by latitude, the tilt of the Earth's axis, the movements of the Earth's wind belts, the difference in temperatures of land and sea, and topography. A simple way of remembering the difference is that "climate is what you expect (e.g., cold winters) and weather is what you get (e.g., a blizzard)" (Glossary of Climate Change Terms 2013).

Global warming – is a gradual increase in the overall temperature of the earth's atmosphere generally attributed to the greenhouse effect caused by increased levels of carbon dioxide, CFCs, and other pollutants (The American Heritage Dictionary of the English Language 2011).

Global climate change – is the periodic modification of Earth's climate brought about as a result of changes in the atmosphere as well as interactions between the atmosphere and various other geologic, chemical, biological, and geographic factors within the Earth system (Encyclopedia Britannica 2008).

Greenhouse effect – is the warming of an atmosphere by its absorbing and emitting infrared radiation while allowing shortwave radiation to pass on through (Ahrens 2011).

Radiation – the Sun with a surface temperature of about 6000 K radiates short wavelength energy (with a peak at 500 nm, corresponding to Planck's and Wien's

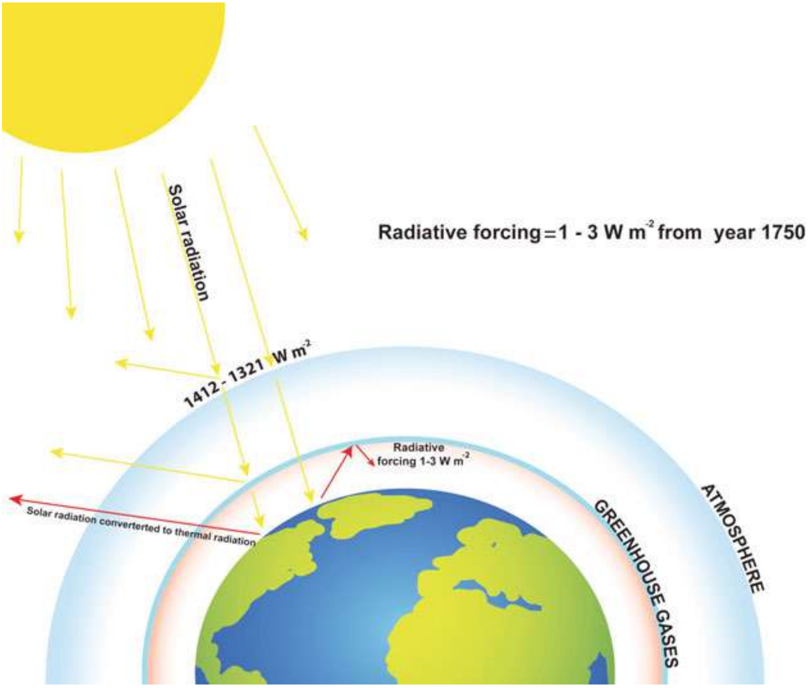

Fig. 1 1412–1321 W m^{-2} of solar energy comes on outer layer of Earth's atmosphere due to its elliptic trajectory. Radiative forcing caused by an increase in greenhouse gases in the atmosphere has risen by 1–3 W m^{-2} from 1750

laws). The atmosphere influences the spectrum of incident light both quantitatively and qualitatively. Shortwave radiation passes through clear atmosphere, and it is trapped by clouds. In the nineteenth century, Arrhenius pointed out that some atmospheric gases absorb longwave radiation, and an increase in their concentration would result in an increase of global temperature on the Earth. The gases mainly responsible for the earth's atmospheric greenhouse effect are water vapor, carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). They are called **greenhouse gases** (GHGs). The surface of the Earth, with its temperature ca. 300 K, emits longwave radiation (with a peak at 10000 nm).

Radiative forcing – is the change in the net radiative flux expressed in W m^{-2} (downward minus upward) at the tropopause or top of atmosphere. It occurs due to a change in an external driver of climate change, such as a change in the concentration of CO_2 or in the output of the Sun. The IPCC (2007) documents the radiative forcing caused by an increase in greenhouse gases in the atmosphere from 1750 as between 1–3 W m^{-2}. In the next 10 years, the radiative forcing is expected to increase by 0.2 W m^{-2}. Radiative forcing cannot be measured; it is calculated (Myhre et al. 2013) (Fig. 2).

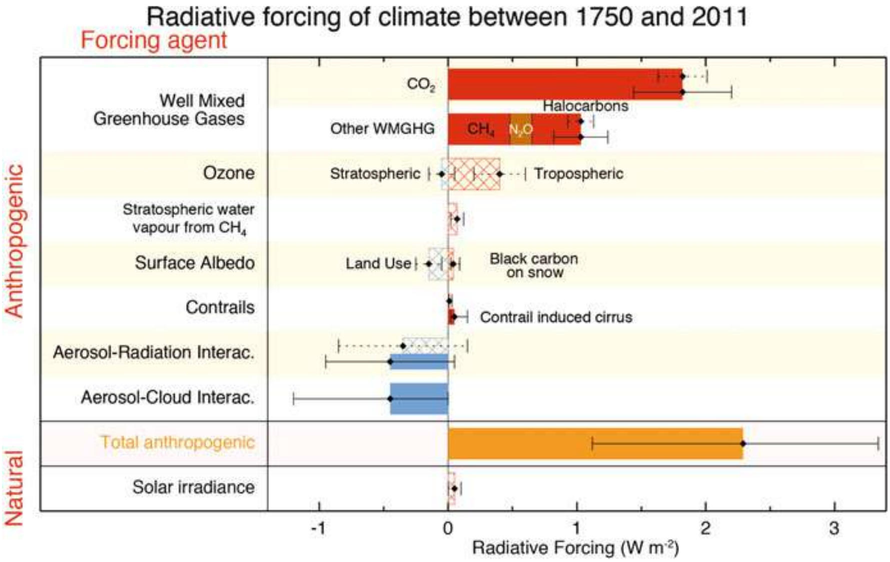

Fig. 2 Radiative forcing of climate caused by individual agents and total radiative forcing between 1750 and 2011 (Myhre et al. 2013). Total RF is less than 2.3 W m^{-2} with standard deviation 1.1 W m^{-2}

Solar Energy Flux Between Sun and Earth

For a mean distance between the Sun and the Earth, the intensity of solar radiation incident upon a surface perpendicular to the Sun's rays measured above the atmosphere is approximately 1367 W m^{-2}. This quantity is called the **solar constant**. The actual direct solar irradiance at the top of the Earth's atmosphere fluctuates during a year from 1412 W m^{-2} to 1321 W m^{-2} due to the Earth's varying distance from the Sun (Kopp et al. 2005). The maximum irradiance on Earth's surface commonly lies between 800 W m^{-2} and 1000 W m^{-2} in the tropics and subtropics and during the growing season in temperate zones. This indicates that approximately 25–40% of energy incident on the upper layer of the atmosphere is reflected, scattered, or absorbed in the atmosphere and does not reach the Earth's surface (Fig. 1).

The amount of incoming energy differs significantly with weather conditions. The difference between the amounts **of incoming radiation on a clear day** (e.g., 8.5 kWhm^{-2} and maximum flux 1000 W m^{-2}) can be an order of magnitude higher than the amount of incoming radiation **on an overcast day** (e.g., 0.78kWhm^{-2}, maximum flux 100 W m^{-2}). Part of the energy is reflected straight away after incidence. The ratio of reflected to incident radiation is called **albedo**. Dark surfaces such as water, wet soil, and wet vegetation absorb solar radiation whereas light surfaces like snow or sand are more reflective. The sum of incoming radiation minus all outgoing radiation across a unit area of the plane is called **net radiation**.

Main Fluxes of Solar Energy in Landscape

There is a big difference between the distributions of net radiation in functioning natural ecosystems of high plant biomass well supplied with water (such as wetlands) versus dry, nonliving physical surfaces. In ecosystems, net radiation (Rn) is divided in varying proportion into following four parts: latent heat flux (LE), sensible heat flux (H), ground heat flux (G), and storage of energy (S).

Latent heat flux represents the energy that is released or absorbed from the surface during phase transition process. Transition of liquid into a gas phase consumes energy and thus local cooling accompanies it. Latent heat flux is generally referred to as evapotranspiration, which describes the total evaporation from land surface and transpiration by plants. **Evapotranspiration** from wetlands use several hundred W m^{-2} on a sunny day.

Sensible heat flux represents the sum of all heat exchanges between the surface of landscape and its surroundings by conduction and convection. The proportion of sensible heat in the energy balance of an ecosystem increases when water is not present, since the capacity for evaporative cooling by latent heat is diminished. On dry surfaces, the sensible heat flux may reach values of several hundreds of W m^{-2} at a sunny day (Huryna et al. 2014).

Ground heat flux is positive when the ground is warming, normally being positive during the day and negative at night. During the plant-growing period in daylight hours, G can reach up to 100 W m^{-2}.

The **energy stored** in vegetation is the smallest part of R_n. There are two energy sinks within a plant stand: metabolic sink (**photosynthesis** with consequent biomass production) and a physical sink (**heating of the plant material** itself). Energy stored flux is a maximum of 30 W m^{-2} on a sunny day, i.e., several percent of $R_{n.}$

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Local Climate Regulation by Urban Wetlands

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Contents

Abstract

Urban areas are composed of built structures, such as buildings and roads, and naturalistic areas, such as parks and rivers, which create local-scale climatic regimes. Within the urban environment, these various elements can combine to modify significantly the fluxes of heat, moisture, and momentum and further alter atmospheric processes through anthropogenic inputs of heat, water, and pollutants. The creation and management of wetlands within urban areas can help to modify local climate and to decrease air temperatures.

Keywords

Climate regulation · Urban heat islands · Heat fluxes · Water sensitive urban design

Introduction

Urban areas are composed of built structures, such as buildings and roads, and naturalistic areas, such as parks and rivers, which combine to create local-scale climatic regimes (Grimmond and Oke 1999). Within the urban environment, these various elements can combine to modify significantly the fluxes of heat, moisture,

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and momentum and further alter atmospheric processes through anthropogenic inputs of heat, water, and pollutants. A common climatic result of the creation and expansion of cities is the production of "urban heat islands" (UHI) where the air temperatures are relatively higher than the corresponding latitudinal and altitudinal values found in rural areas. The form and configuration of urban areas can significantly influence local weather patterns, but the issue can be complicated because of the complexity of the urban terrain, habitats, and the associated turbulence and energy transfer processes.

Role of Evapotranspiration

The role of evapotranspiration (*ET*) has often been neglected in hydrological studies within urban environments. This is primarily because of the dominance of built structures and impervious surfaces and the relatively reduced rates when compared with other natural or rural ecosystems (Grimmond and Oke 1999). However, increasingly evapotranspiration, along with urban albedos which have been shown to decrease summer temperatures (Taha 1997), is being considered as a potential moderator of steadily increasing urban temperature extremes and particularly as a potential mitigation measure to combat the negative health impacts caused by urban heat islands (Haines et al. 2006).

Increasingly consideration within urban planning and management is being given to the "oases" effect of vegetated areas within towns and cities where, in addition to providing shading and a range of other benefits, under the appropriate conditions, evapotranspiration from urban green and blue space can reduce air temperatures by up to 8 °C in comparison to surrounding areas (Taha 1997; McPherson 1994). Under certain circumstance, the latent heat flux (λE) can be sufficient that the sensible heat flux (H) becomes negative, causing the air above vegetated surfaces and over drier built environments to supply sensible heat to the vegetated areas (Taha 1997). This phenomenon has been recorded from a range of urban habitats including suburban lawns, urban woodlands and parks, green roofs, and wetlands. In some of these situations, the Bowen ratio (ratio of H to λE) can tend towards the negative resulting in a significant reduction in air temperatures. The increased evapotranspiration rates associated with vegetated urban spaces have been shown to produce a maximum cooling of 1.6 °C from urban parks in Hong Kong (Tong et al. 2005) and 2 °C from urban grasslands in Tokyo (Ca et al. 1998).

Despite the predominance of impervious surfaces and built structures, evapotranspiration represents an important flux within urban environments, acting as an energy sink and reducing urban air temperatures. Not only do urban green and blue spaces provide a range of ecological and social benefits within a largely artificial environment, they can be important drivers of climatic functioning assisting to modify the local hydrological cycle. Studies from China have suggested that wetlands can produce "urban cooling islands" (UCIs) which can play an important role in mitigating extreme summer air temperatures. A study undertaking in Beijing concluded that urban wetlands played an important role in mitigating urban heat

Table 1 Urban ecosystems generating local and direct services relevant to Stockholm, Sweden (source infromation from Bolund and Hunhammer 1999)

	Street tree	Lawns/ parks	Urban forest	Cultivated land	Wetland	Stream	Lakes/ sea
Air filtering	X	X	X	X	X		
Microclimate regulation	X	X	X	X	X	X	X
Noise reduction	X	X	X	X	X		
Rainwater drainage		X	X	X	X		
Sewage treatment					X		
Recreation/ cultural values	X	X	X	X	X	X	X

island effects. Reservoirs, lakes, and rivers were investigated to understand the temperature difference and gradient between wetlands and their surrounding urban landscape. The results indicated that wetland shape and location strongly influence the degree of cooling in the urban environments. The spatial heterogeneity of both the wetlands and its surrounding landscape significantly impact upon the cooling effects of wetlands (Sun et al. 2012).

Water-Sensitive Urban Design

A study undertaken in Stockholm, Sweden, investigated six ecosystem services performed by different urban habitats (Bolund and Hunhammer 1999) (Table 1). In addition to wetlands, six different urban ecosystems were studied: street trees, lawns/parks, urban forests, cultivated land, lakes/sea, and streams. Of these all of them regulated the urban microclimate and provided recreational and cultural values. However, wetlands were the only ecosystem to deliver all six of the ecosystem services under investigation. This demonstrates the potential to combine climate regulation benefits within urban areas with a range of other ecosystem services.

Water Sensitive Urban Design (WSUD) is a well-established approach to dealing with the challenges of storm water management within towns and cities. WSUD adopts techniques and approaches which into retain water within the urban landscape through a range of measures including rainwater harvesting, infiltration into soils, and the retention within the specially designed urban drainage features such as wetlands. Of the main objectives of WSUD have traditionally been the reduction of urban flooding and the improvement of water quality combined with a visual amenity benefits. However, in Australia, due to increasing concerns regarding the elevated urban temperatures and the potential impacts on the thermal wellbeing of the human population, attempts are being made to integrate the cooling properties of wetlands within the WSUD approaches (Coutts et al. 2013). In the warm and dry climate associated with Australian cities, it is suggested that the integration of wetlands within the WSUD could produce local temperature reductions in excess of the 2–3 °C recorded for other cities in more temperate climates. Once more the objective within the Australian approach is not just to deliver climate cooling benefits but to combine these within the wider benefits provided by wetlands in urban environments.

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Climate Regulation: Salt Marshes and Blue Carbon

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Beverly J. Johnson, Catherine E. Lovelock, and Dorothée Herr

Contents

Abstract

Carbon sequestered and stored in, or released from, salt marshes, mangroves, and seagrass ecosystems is often referred to as coastal "blue carbon." The term was first used in 2009 as a means of highlighting the significance of carbon

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sequestration and storage in these highly productive coastal ecosystems, largely to the policy and carbon finance communities.

Keywords
Carbon sequestration · Carbon storage · Saltmarshes

Introduction

Carbon sequestered and stored in, or released from salt marshes, mangroves, and seagrass ecosystems, is often referred to as coastal "blue carbon." The term was first used in a 2009 United Nations Environment Programme (UNEP) report "Blue Carbon" (Nellemann et al. 2009), as a means of highlighting the significance of carbon sequestration and storage in these highly productive coastal ecosystems, largely to the policy and carbon finance communities.

The sustainable management, conservation, and enhancement of sinks and reservoirs of all greenhouse gases (GHG) in natural environments, "including biomass, forests and oceans as well as other terrestrial, coastal and marine ecosystems" (UNFCCC Art. 4.1(d)) has been enshrined into the United Nations Framework Convention on Climate Change (UNFCCC) since its adoption in 1992. Its accompanying Kyoto Protocol also clearly allows for specific accounting of reducing emissions by sources and removals by sinks in specific natural systems – mainly related to terrestrial land-use changes and forestry activities.

Terrestrial forested ecosystems have, therefore, received most attention for their role in the drawdown and sequestration of CO_2 and climate change mitigation and for years have featured prominently in UNFCCC processes. In contrast, coastal carbon ecosystems have been, until recently, largely ignored in international and national carbon accounting.

It was not until the development of, and growing international interest in, the Reducing Emissions from Deforestation and Forest Degradation Plus (REDD+) program as a financing scheme for forest restoration, conservation, and overall management that the marine community started looking into the "important missing sinks" (Pidgeon 2009) in the climate mitigation debate.

While scientific research has been conducted on carbon dynamics in coastal ecosystems for some time, it was not until 2009 with the publication of IUCN's report *The Management of Natural Coastal Carbon Sinks* (Laffoley and Grimsditch 2009) and the UNEP report *Blue Carbon* (Nellemann et al. 2009) that this topic received greater attention. Since then there has been an increase in scientific research as well as improved development of management and policy responses for coastal "blue carbon" ecosystems (Herr et al. 2012; Sutton-Grier and Moore 2016).

Coastal blue carbon ecosystems are among the most productive in the world (Mitsch and Gosselink 2000; McLeod et al. 2011). The average global carbon burial rates are an order of magnitude greater than those of upland forests (Fig. 1); thus CO_2 drawdown and carbon storage potential in blue carbon ecosystems is extraordinary.

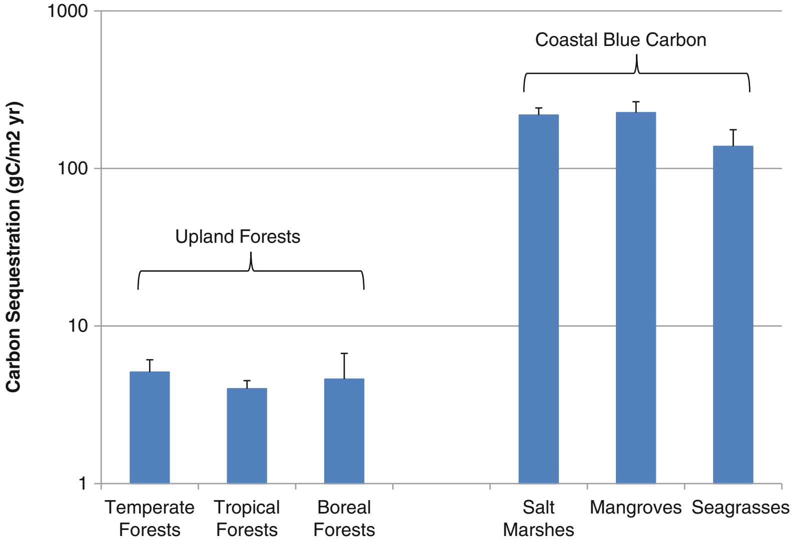

Fig. 1 Average global carbon burial rates (error bar equal to one standard deviation) for upland forests and coastal blue carbon ecosystems (Modified after McLeod et al. 2011)

In addition to carbon storage, salt marshes provide a variety of other benefits – ecosystem services – to humans. They provide nutrients and habitat for many estuarine and marine organisms, buffer against storm surges reduce coastal erosion, and filter out nutrients and pollutants. These ecosystem services are important for improving local livelihoods, tourism and culture, as well as for climate change mitigation and adaptation.

In this article, we present a broad overview of the general science behind carbon accounting in blue carbon ecosystems, with a focus on salt marshes, and refer the reader to the appropriate references for making these measurements. We also include an update on the current policies and programs in place designed to utilize blue carbon ecosystems for climate mitigation. It is important to keep in mind that the nature of these fields (blue carbon science and policy) is evolving rapidly at the writing of this article. The reader is encouraged to use the information presented here as a starting point for exploring blue carbon as an opportunity for coastal management efforts.

Salt Marshes and Climate Change Mitigation

Salt marshes and coastal blue carbon ecosystems mitigate climate change by reducing GHG concentrations in the atmosphere. To include coastal blue carbon ecosystems in a carbon accounting scheme, it is necessary to track the GHG fluxes in to

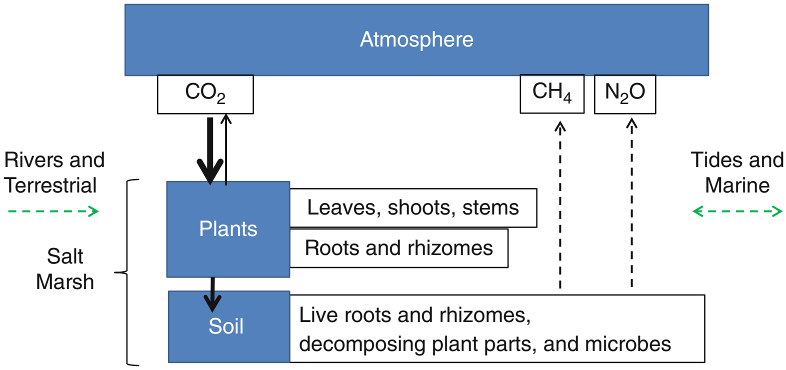

Fig. 2 Concept map of carbon reservoirs and GHG fluxes in a salt marsh. *Boxes* = reservoirs of organic matter (box size is not proportional to reservoir size); *arrows* = fluxes of organic matter and/or GHG. *Black solid arrows* = dominant fluxes of carbon (photosynthesis, respiration, and burial); *black dashed arrows* = variable, and usually minor, fluxes of GHG; *green dashed arrows* = variable, and often unknown, inputs of organic matter from terrestrial and marine biomes

(and out of) the area in question, as well as the amount of carbon that is stored within its reservoirs. A step-by-step manual for making carbon stock and GHG flux measurements in coastal blue carbon ecosystems has been developed by the Blue Carbon Initiative and can be accessed online (Howard et al. 2014).

Figure 2 provides a simple concept map illustrating the major carbon reservoirs and GHG fluxes in salt marshes. These are further described below.

Carbon Reservoirs

The major carbon reservoirs in a salt marsh are in soils and plants. Soils are composed of a combination of living below ground biomass, decomposing organic matter, and inorganic sediments and represent the largest reservoir of stored carbon (Murray et al. 2011). Salt marsh vegetation is primarily composed of a mix of herbaceous plants (grasses, sedges, rushes, forbs) and succulent and sometimes woody chenopods.

Very simply, CO_2 in the atmosphere is converted into organic matter via photosynthesis by plants. Some of the organic matter is used to synthesize leaves and shoots; most is transported to and stored within the roots and rhizomes (Elsey-Quirk et al. 2011; Pickoff 2013). Because salt marshes are inundated regularly by the tides, the underlying sediments are saturated and largely anoxic, and the organic matter stored within decomposes relatively slowly. Thus, the bulk of the organic matter in salt marsh (and all blue carbon) ecosystems is stored in the soils (Fig. 3).

The global average carbon storage in the top meter of salt marsh soils is approximately 255 Mg C ha^{-1} (with a range of 16–623 Mg C ha^{-1}) (IPCC 2013). But many salt marshes have carbon stored to depths of as much as 6 m below the surface, making the above estimates conservative. North American tidal marshes, for

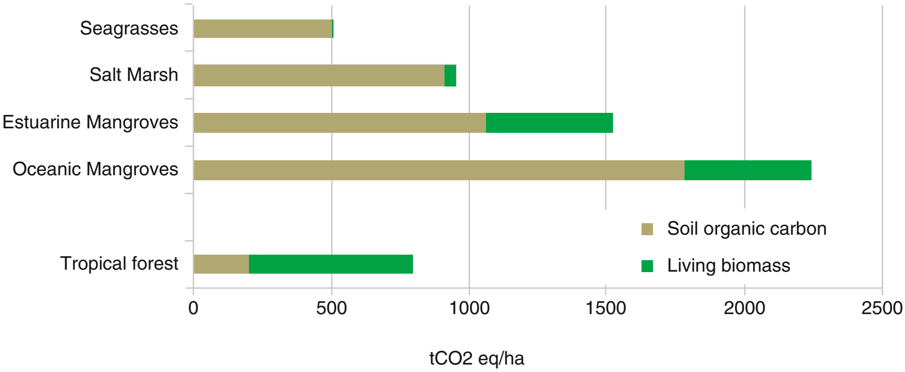

Fig. 3 Average tons of CO_2 equivalents per hectare stored in the above ground/living biomass and the upper meter of soil organic carbon for coastal blue carbon ecosystems and tropical rainforests (From Murray et al. 2011; https://nicholasinstitute.duke.edu/environment/publications/naturalresources/blue-carbon-report)

Fig. 4 North American tidal saltmarshes, such as this marsh in Phippsburg, Maine, store large amounts of carbon and many times more than forests. (Photo credit: Phyllis Graber Jensen © copyright remains with the author)

example, have been found to store up to 1700 Mg C ha−1, well above the global average (Fig. 4) in sediments up to 4000 years old.

Soil carbon in salt marshes is primarily autochthonous (i.e., generated in situ). Minor amounts of allochthonous organic matter (derived from terrestrial or marine sources) can also be incorporated into the soil carbon pool. It is possible to use stable isotopes and/or lipid biomarker analyses to identify the source of organic matter within the sediments (Chmura and Aharon 1995; Johnson et al. 2007). Distinguishing between autochthonous and allochthonous sedimentary carbon may be important, depending on the project.

Salt marshes have been subject to modification and conversion by humans for centuries to millennia (Adam 2002; Bromberg Gedan et al. 2009), with an estimated net loss of 35–50% (Murray et al. 2011). Recent global estimates of annual losses of salt marsh ecosystems ranges between 1 and 2% (Duarte et al. 2008). Marshes have been/are (1) used for grazing and haymaking; (2) converted to agricultural, aquacultural, and urban landscapes; and (3) altered for insect control *inter alia*. Salt marshes are currently vulnerable to sea level rise and coastal storms, sediment starvation, coastal squeeze, excess nutrients, invasive species, runaway consumer effects, and ill-informed management decisions (Bromberg Gedan et al. 2009; Kirwan and Megonigal 2013). The degree to which carbon dynamics are impacted by these and other types of alterations is a topic of study by many (e.g., Vincent et al. 2013; Macreadie et al. 2013; Gunn 2016; Martin and Moseman-Valtierra 2017; Kroeger et al. 2017).

Greenhouse Gases

The three major GHG emitted from salt marshes include CO_2, methane (CH_4) and nitrous oxide (N_2O) (Mitsch and Gooselink 2000; Moseman-Valtierra 2013). Because CH_4 and N_2O have global warming potentials that are 45 and 310 times greater than CO_2 over 100 years, respectively, it is necessary to consider the release of these gases for climate mitigation. Release of GHGs can be significant in marshes that experience reduced soil water salinities, changes in soil oxygen availability, and increases in anthropogenic nutrient loading (Moseman-Valtierra 2013; Adams et al. 2012; Emery and Fulweiler 2014).

Carbon Dioxide (CO_2)

CO_2 is drawn out of the atmosphere via photosynthesis and released via respiration. Though there is some uncertainty in flux measurements due to a lack of data and variability among ecosystems, approximately 20% of the carbon fixed during photosynthesis is thought to be respired as CO_2, 70% exported into the surrounding estuary as organic carbon or inorganic carbon, and 10% buried in the soils (Bauer et al. 2013). Recent estimates of global carbon accumulation rates in salt marsh sediments have continued to increase as more and more data are generated and synthesized. Estimates of 150, 218 and 244.7 g C m^{-2} $yr.^{-1}$ have been published in Duarte et al. (2005), McLeod et al. (2011), and Ouyang et al. (2014), respectively (Kroeger et al. 2017).

Drainage of wetlands, and lowering of the water table, results in a rapid emission of CO_2 through oxidation (Crooks et al. 2011; Lovelock et al. 2011; Pendleton et al. 2012). Restoring the hydrology to a drained wetland that is emitting CO_2, CH_4, or N_2O can reduce emissions effectively (Kroeger et al. 2017).

Methane (CH_4)

Highly variable in wetland systems, methane emissions are generally lower where water sulfate concentrations are high, because methanogens are outcompeted by sulfate-reducing bacteria (Bartlett et al. 1985). Using salinity as a proxy for sulfate concentrations (where both parameters are high in marine waters), Poffenbarger et al. (2011) have determined that methane emissions are minimal in tidal wetlands with salinity greater than 18 PSU (practical salinity unit).

Tidal restrictions and subsequent reduction of soil salinities below 18 PSU can result in CH_4 emissions (Gunn 2016). Restoring tidal flow to increase salinity may provide a reduction in methane emissions (Kroeger et al. 2017). Restoring a site below the salinity threshold could result in new methane emissions, and these should be quantified and weighed against the greenhouse gas benefits to accurately predict the overall climate mitigation benefits of such projects.

Nitrous Oxide (N_2O)

N_2O is a naturally occurring gas in wetland soils, with concentrations increased by anthropogenic nitrogen pollution. Creating or restoring a wetland where none exists may result in an increase in N_2O emissions (Adams et al. 2012). Additionally, anthropogenic loading of nitrate can promote N_2O emissions from salt marshes (Moseman-Valtierra 2013).

Climate Finance and Policies for Better Management

Policies and finance mechanisms are now being developed and implemented for climate change mitigation. They present the possibility to mobilize additional funds and revenue schemes to combine best practices in coastal management with climate change mitigation goals and needs (Herr et al. 2012; Herr et al. 2015).

Conserving and restoring salt marshes can become a management activity as part of the other land-use activities aimed at reducing carbon emissions and enhancing carbon sequestration in natural systems. Such activities are also known as nature-based solutions to climate change mitigation. Projects and activities incentivizing such management responses are most advanced for terrestrial tropical forests policies and programs, such as the REDD+ program developed under the UNFCCC (Pendleton et al. 2012).

Salt marshes can also be part of Nationally Appropriate Mitigation Actions (NAMAs). Some technical elements need to be fully integrated into the implementation of such mechanisms. For example, in order to put a value on their full coastal carbon potential soil carbon needs to be accounted for, not the least because salt marshes do not have much plant biomass (in comparison to tropical forest) for which to account.

"Wetland drainage and rewetting" has been accepted as a new management effort that may (but not must) be included in national accounting of GHG from the Land-Use, Land-Use Change and Forestry (LULUCF) sector by developed countries under the Kyoto Protocol, as part of the UNFCCC. For example, emissions arising from the drainage of tidal marshes would be a qualified activity, so would subsidence reversal by gradually raising water levels and building soil surfaces to intertidal elevation.

On an implementation level, saltmarshes can be included in national GHG accounting and national reporting from all countries to the UNFCCC, now that the IPCC 2013 Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories: Wetlands (Wetlands Supplement) has been issued (IPCC 2013).

Carbon markets are one of many options to incentivize and finance better management of coastal carbon ecosystems, alongside other types of climate finance and more "traditional" conservation approaches (Herr et al. 2015). While the regulated markets (including the Clean Development Mechanisms as part of the Kyoto Protocol) have limited scope or are not trading at all in land-use related credits, the voluntary carbon market has seen an increase in available carbon offset methodologies (Emmer and von Unger 2014; Herr et al. 2015). These are needed to develop, account for, and credit GHG removals or emissions. The Verified Carbon Standard (see: http://www.v-c-s.org) has one Methodology for Coastal Wetland Creation (VM0024) (limited to the Coastal United States) and one Methodology for Tidal Wetland and Seagrass Restoration (VM0033). It outlines procedures to quantify GHG emission reductions resulting from tidal wetland restoration projects. Such projects include creating or managing the conditions required for healthy, sustainable wetland ecosystems.

The American Carbon Registry issued a methodology on Restoration of Degraded Deltaic Wetlands of the Mississippi Delta (see: http://americancarbonregistry.org/carbon-accounting/standards-methodologies/restoration-of-degraded-deltaic-wetlands-of-the-mississippi-delta) that details requirements for GHG emission reduction accounting from wetland restoration activities implemented on degraded wetlands of the Delta. The methodology quantifies increased carbon sequestration in above ground biomass, below ground biomass, and soil organic carbon over and above the baseline scenario. A methodology on California Deltaic and Coastal Wetland Restoration is currently in development.

The new Paris Agreement to the UNFCCC, a result from its 2015 Conference of the Parties (COP 21), sets the framework of how to deal with GHG emissions mitigation, adaptation, and finance starting in the year 2020. It makes several references to all sinks and reservoirs/ emissions by sources and removals by sinks of GHG, also quoting Art 4.1(d) of the Convention itself, which specifically lists the conservation of coastal and marine ecosystems. This provides the right signals to increase post-2020 climate activities and continue the work being undertaken to implement various mitigation-related efforts in coastal areas – national GHG accounting, REDD+, NAMAs, and the voluntary carbon market.

The cost of conserving saltmarshes as a means to reduce GHG emissions relative to other emissions reduction approaches (such as energy efficiency or alternative energy generation) is less clear. As nations struggle to reduce GHG emissions, it has become apparent that targets and commitments cannot be met through increased regulation of any single source of GHGs. Avoided emissions through salt marsh and other coastal and marine ecosystem conservation is likely one of the many options that should be included in a portfolio of cost-effective mechanisms for GHG reductions. An expansion of the implementation of programs and projects, using the above mentioned mechanisms and means, all around the world is still needed to stop the ongoing loss of these systems and the resulting emissions.

Carbon Management Responses

In order to determine net climate mitigation benefits of a salt marsh restoration or conservation project, the changes in GHG reductions and emissions as a result of the project activities have to be calculated against the GHG reductions and emissions which would have occurred in the absence of the project (called the "baseline") (Emmett-Mattox and Crooks 2014).

Conservation of Intact Wetlands

Large emissions of CO_2 emissions can be prevented if intact salt marshes are not drained or changed to other land-uses such as agriculture or aquaculture. Although large-scale conversions are being regulated and limited in the USA, European Union, and Australia, wetland conversions with high impacts are still very common globally. Projects conserving carbon stocks within at-risk wetlands through regulation and/or land owner agreements are eligible for carbon credits and national carbon accounting.

Rewetting of Drained Organic Soils

CO_2 is continuously emitted from organic soils that have been drained until either the water table rises to near the surface of the soil or the stock of carbon is depleted. Management of water tables to reduce CO_2 from drained organic soils is an eligible climate change mitigation activity.

Restoration and Creation of Vegetated Wetlands

Restoration and creation activities such as lowering of water levels on impounded former wetlands, removing tidal barriers, raising soil surfaces with dredged material, or restoring salinity conditions restore or create new habitats. Activities that restore the combination of native plants, hydrology, and sediment will lead to a self-sustaining productive wetland, creating net GHG benefits. The success of mitigation projects also depends on site-specific examinations of habitat restoration and management potentials, as the response from salt marshes differ on a regional and watershed level.

Future Challenges: Filling Knowledge Gaps

The global extent of coastal marsh and rates of loss are currently associated with relatively large uncertainties, and further work is needed in this area. Additional mapping of converted and degraded salt marshes and the quantification of emissions from exposed organic soils is needed to enable inclusion in relevant databases (e.g., the IPCC Emission Factor Database).

Emission rates associated with specific human activities over time for a range of drivers of ecosystem degradation or loss (e.g., drainage, burning, harvesting, or clearing of vegetation at different intensity levels) are also limited at the moment.

A significant amount of the eroded coastal carbon is thought to be dissolved in the ocean water where it enters the ocean-atmosphere system, and the remaining eroded carbon is deposited in offshore sediments and sequestered. The fate of carbon eroded from salt marshes and carried offshore by ocean waves and currents is an ongoing topic of scientific research.

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Climate Regulation: Southeast Asian Peat Swamps

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Marcel Silvius and Arina Schrier

Contents

Abstract

Southeast Asia's coastal lowlands have been dominated by forested bog-type peat swamps, covering over 24 million hectares and storing approximately 68.5 Giga tons of carbon. Since the 1980s most peat swamps in Sumatra, Borneo and Peninsular Malaysia have been affected by logging and conversion to drainage-based land-uses. In Indonesia this started as part of large-scale transmigration schemes. Later, plantation development for palm oil and *Acacia* (pulp for paper) production became the main drivers throughout the region. Drained peatlands are affected by continuous oxidation and the desiccated landscapes have been frequently hit by widespread peat fires, affecting millions of hectares and causing major smog events. The resulting CO_2 emissions contribute substantially to climate change. Peatland drainage also results in soil subsidence, which in time will cause increased and prolonged flooding and subsequent loss of productivity. The paper suggest that this can be prevented only by rewetting the peat and introducing alternative - wet - production systems

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(paludicultures) preferably involving permanent crop species. This may be best done as part of mixed business approaches and may offer also opportunities for gaining carbon credits from peat conservation and carbon sequestration carbon under productive land-use, adding to sustainable livelihood options for local people.

Keywords
Peatlands · Peat swamp forests · Drainage · Deforestation · Carbon dioxide emissions · Rewetting · Sustainable livelihoods

Introduction

Extensive coastal lowland areas in Southeast Asia have been dominated by tropical peat swamp forests, covering an estimated 247,778 km^2 in total (Page et al. 2011), representing the largest tropical peat carbon store of the world. Until the 1980s, these forests remained relatively undisturbed as they were considered inhospitable and not suitable for agriculture. Local land-uses were confined to logging and harvesting of nontimber forest products (including fish), sometimes with enrichment planting of useful plant species (such as rattan). In the 1970s, most peat swamp forests in Indonesia were allocated to selective logging schemes. In the 1980s, exploitation of peatland areas started to gather momentum (Silvius and Diemont 2007), involving extensive transmigration schemes in east Sumatra and parts of Kalimantan which converted many of these areas to agricultural land. However, many of these developments failed, resulting in extensive deforested idle peatlands with extensive networks of drainage canals. Even after abandonment, the drainage canals were kept open as they provided a means for transport. As a result, the peat areas were progressively desiccated and extremely fire prone. In the regularly occurring dry years (linked to the El Niño Southern Oscillation), millions of hectares were burned. This generated a related significant trans-boundary haze phenomenon which was first noticed in 1981/82 and has in every dry year since impacted on the region's remaining (peat) forests, biodiversity, public health, and the local economy.

In Peninsular Malaysia, large-scale oil palm plantation development on peatlands also started in the 1980s, especially in Johore. Since 1990, oil palm plantation development has boomed and replaced many of the peat swamp forests in Peninsular Malaysia and Sarawak. Similarly, many of the remaining peat swamp forests in Sumatra and Kalimantan were turned into industrial plantation concessions, mainly for palm oil, but especially in Riau province also for pulp wood (*Acacia*) plantations.

By 2010, industrial plantations on peatlands in Malaysia and Indonesia covered around 3.1 million hectares (Mha), but this area may almost double by 2020 (Miettinen et al. 2012). Increasingly peat swamp forest areas have been claimed as community land under local customary "*adat*" rights, and many smallholder estates are located on peat. Many estates have been cleared with the use of fire.

In the 1980s and 1990s, climate change was not yet recognized as a major issue, and hence no attention was paid to CO_2 emissions from deforestation or drained peatlands. Awareness of peatland degradation-related emissions only started in the new millennium, when the extraordinary high CO_2 emissions were first reported in Nature by Page et al. in 2002, showing that peat fires in the dry year of 1997 contributed 15–40% of global emissions. However, in 2007, Indonesia was still shocked when it was found to be responsible for the third highest emissions of CO_2 worldwide. This was mainly a result of peatland degradation, deforestation, and fires, with continuous peat oxidation resulting from drainage contributing over half of the emissions (Hooijer et al. 2010).

Carbon Storage Function of Southeast Asian Peat Swamp Forests

Southeast Asian peatlands function as the earth's most spatially efficient carbon sink. They store approximately 68.5 G tons (1 Gt = 1 Gigaton or 10^9 metric tons) of carbon in the deep peats (compared to 550 Gt carbon in peatlands globally). The peatlands are dome shaped, lying largely above groundwater and therefore fed by rain water. An important consequence of this is that once deforested and drained, peat is rapidly degraded (Page et al. 2009). Following drainage of a tropical peatland, the hydrology is disrupted and affects the functioning of the vulnerable peat ecosystems, including aerobic decomposition. This results in loss of soil carbon (Wösten and Ritzema 2001) and in soil subsidence resulting from compaction, shrinkage, and oxidation (Wösten et al. 1997; Couwenberg and Hooijer 2013).

In 2011, the increased awareness on the atmospheric haze and carbon emission issues triggered the establishment of a 2-year moratorium on further development in primary forests and on peatlands in Indonesia. However, in Malaysia, especially in Sarawak, peatland development has continued unabated. The related soil subsidence issue and subsequent issues of flooding and salt water intrusion so far remain relatively unrecognized. With a subsidence rate of 3.7–5 cm per year (Couwenberg and Hooijer 2013), many of the drained and subsiding peatlands will reach the drainage base by the mid-2050s, resulting in flooding of peatland areas along thousands of kilometers of coast and rivers. In addition, because of changes in the drainage pattern the freshwater buffer function of these swamps in the coastal zone will decrease, enhancing salt water intrusion in the dry season and reducing prospects for agriculture and fisheries in the coastal zone (Silvius et al. 2000; Wösten et al. 2006). These impacts may result in irreversible loss of huge areas of arable land and reduce options for climate change adaptation.

Deforestation, Peat Drainage, Oil Palm, and Pulp and Paper

Almost all peat swamp forest in Southeast Asia have been impacted by either conversion, drainage, or logging. The current deforestation rate of tropical peat swamp forests is twice that of other forests in Asia, with highest rate of 8% in the peat swamp forests of Sarawak (SarVision 2011). In Indonesia, peat development is most extensive in Sumatra, followed by Kalimantan, while much of the peat in Papua remains undeveloped. Peatlands in Sulawesi have largely been converted. Between 1997 and 2007, industrial oil palm and pulpwood (*Acacia*) plantations expanded dramatically, from 0.3 to 2.3 Mha (comprising 2.1 Mha of oil and 0.2 Mha of *Acacia*), an increase from 2% to 15% of the total peatland area in Southeast Asia (Miettinen and Liew 2010). The reason for the pulp and paper business to focus its development on peat soils lies in the relative easy and cheap access to these areas, being located close to the coast (harbors, water transport). In addition, they have no or few traditional land right claims, are under direct jurisdiction of local authorities, and have still significant and large contiguous stretches of natural forests that can provide huge supply to the pulp mills as part of the process of plantation establishment (Barr et al. 2010)

Climate Impacts of Developments

When peatlands are drained, the peat will be subject to oxidation and a higher risk of forest and peat fires, resulting in increased GHG emissions and carbon losses (van der Werf et al. 2009). FAOSTAT 2012 indicates a global contribution of drained organic soils under cropland of 0.75 Gt CO_2/yr with the largest contributions from Asia's drained peatlands (0.4 Gt CO_2/yr) and Europe (0.15 Gt CO_2/yr). The total carbon dioxide emissions from degraded peatlands currently amount to one third of the worldwide emissions from Land Use, Land Use Change and Forestry (LULUCF) and to 6% of the global anthropogenic carbon dioxide emissions (Joosten 2009). In Malaysia and Indonesia, the largest emission sources are the drainage of peat, related fires, conversion of forest (Fig. 1), and degradation of peat swamp forest

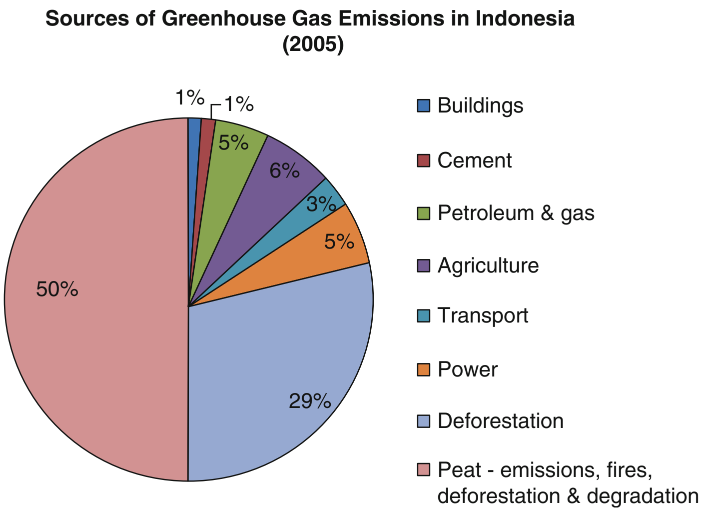

Fig. 1 Sources of the 2.055 million tons of GHG emissions in Indonesia in 2005 (Based on data from Indonesia's greenhouse gas abatement cost curve, © DNPI 2010)

(deforestation and forest degradation, fires, and peat oxidation), resulting in half of the national emissions of Indonesia. Independent research shows that drained peatland in SE Asia emits about 86 tons CO_2 per hectare per year (Jauhiainen and Silvennoinen 2012; Hooijer et al. 2012) under plantation management, or at least 900 g CO_2 m^2 a1 for each 10 cm of drainage depth (Couwenberg et al. 2010).

While conversion of forest for agricultural development is a one-point emission in time, emissions resulting from peat drainage constitute a continuous process that only stops if the drainage stops (through rewetting), when the peat is completely depleted, or if the plantation reaches the drainage base (in relation to river or sea level). Emissions from peat drainage are mainly from land-use rather than land use change.

Current Regulations for Conservation of Forest and Peat

Since 2011, the UNFCCC recognizes peatlands and their potential for emission mitigation action under the Kyoto protocol. Instruments for mitigation include, e.g., REDD+, Clean Development Mechanism (CDM), the Joint Implementation plan (JI), and Emissions Trading mechanisms. There is also a small but growing voluntary market, attracting private sector investors to invest in forest and peatland rehabilitation and conservation. The REDD+ mechanism requires measuring, reporting, and verification of all carbon pools, therefore implicitly also the peat soil carbon stock of peat swamp forests. Thus, the REDD+ mechanism may provide options for climate change mitigation through peatland rehabilitation and conservation.

A key REDD+ initiative at national scale in Southeast Asia is the bilateral Indonesia-Norway REDD+ Partnership which commenced in 2010. Under the agreement, Norway will provide Indonesia up to US$1 billion in performance-linked funds for reducing deforestation and forest and peatland degradation. This is tied to Indonesia's wider aim to deliver low carbon development and contribute to global action to reduce carbon emissions – by committing to a 26% emissions cut from business-as-usual levels and with international support by up to 41% by 2020 (www.un-8redd.org/UNREDDProgramme).

Another example is the policy process of the Round Table on Sustainable Palm Oil (RSPO) which has adopted a resolution in 2009 to explore and develop business models for optimizing sustainability of existing oil palm plantations on peat. The RSPO established a Greenhouse Gas Working Group and Peatland Working Group that explored impacts of oil palm plantations on peat, possibilities for emission reduction, and alternative uses and methods for rehabilitation of peat. In April 2013, the RSPO in a Special General Assembly adopted new principles and criteria that reflect these issues by requiring monitoring and minimizing of GHG emissions, avoidance of peatland areas in new plantation developments, and review of drainability and subsequent rehabilitation needs in existing plantations on peat (Fig. 2).

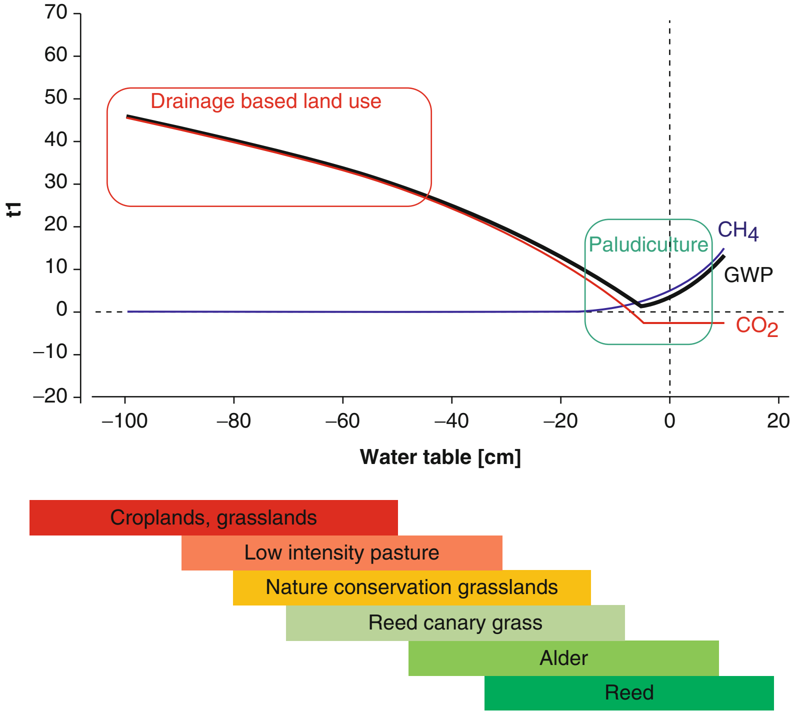

Fig. 2 Greenhouse gas emissions (CO2 and CH4) as a function of water table and land use (After: Couwenberg and Hooijer 2013; Couwenberg et al. 2011; Wichtmann et al. 2010)

Reducing Emissions from Current Agriculture on Peat

The most obvious measure to prevent further negative (climate) impacts from peatland degradation is by stopping deforestation and development of drainage-dependent land-uses in the remaining forested peatland areas. Degraded peatlands should be targeted for rehabilitation through rewetting and revegetation. This may involve productive land-uses that do not require drainage (paludiculture).

In cases where peatlands are already cultivated involving drainage systems, one of the first measures to reduce negative environmental impacts is to reduce drainage depth and to optimize the hydrological system in favor of highest possible water levels. An elevation of the water table from 100 cm to 40 cm below the field level may decrease the emissions by over 50%. Therefore, choosing crops that are adapted to high soil moisture could reduce emissions considerably (Fig. 3). Other measures are cultivation of permanent crops, avoiding regular ploughing, maintaining a cover vegetation, and limiting nitrogen inputs.

Paludicultures are land management techniques that cultivate biomass from wet and rewetted peatlands under conditions that maintain the peat body, facilitate peat accumulation, and sustain the ecosystem services associated with natural peatlands

Fig. 3 The development frontier: harvest of oil palm fruit bunches at Sungai Gelam, Jambi, Province Sumatra, Indonesia, and in the back some remaining peat swamp forest (Photo: © M.J. Silvius)

(Joosten et al. 2012). This presents an alternative for drainage-based land-uses on peat and will generally involve utilization of indigenous (peat-adapted) species. It may involve mixed business approaches, combining permanent tree crops such as the latex producing jelutong (*Dyera* species) or oil producing tengkawang (*Shorea* species), or intercropping such plantations with nondrainage reliant annual crops (e.g., indigenous vegetable or medicinal plant species). This may be combined with other land uses such as fisheries or aquaculture or, e.g., chicken husbandry. Paludiculture will considerably reduce CO_2 emissions (Fig. 3) and thus offers innovative opportunities for gaining carbon credits from peat conservation and sequestration of carbon under productive land-use, adding to sustainable livelihood options for local people.

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Hydrological Services of Wetlands and Global Climate Change

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Charlie Stratford

Contents

Abstract

The presence of wetland habitats in the landscape can have a significant influence on the movement and storage of water at a range of scales. As seasonal or perennial wet areas, wetlands typically slow the passage of water from one place to another and this slowing can provide various benefits, for example, by temporarily storing flood waters or dissipating the energy of coastal storms. Worldwide, wetland hydrological services (including disturbance regulation, water regulation and water supply) are estimated to have an annual value of $2{,}757 \times 10^9$ US$. It is anticipated that global climate change will increase pressure on wetlands and their capacity to provide hydrological services may be affected. Prolonged drought may affect soil structure so that wetlands do not soak up water as readily, and increasingly intense storm events

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may simply overwhelm a wetlands ability to reduce flooding. Understanding the role of wetlands in the hydrological cycle enables us to work towards optimum delivery of hydrological services and how they should best be managed and valued.

Keywords
Natural flood management · Extreme events · Climate regulation · Groundwater recharge

Introduction

The presence of wetland habitats in the landscape can have a significant influence on the movement and storage of water at a range of scales. As seasonal or perennial wet areas, wetlands typically influence the movement of water from one place to another and this can provide various benefits, for example, by temporarily storing flood waters or dissipating the energy of coastal storms. In Costanza et al.'s (1997) valuation of the world's ecosystem services and natural capital, wetland hydrological services (including disturbance regulation, water regulation, and water supply) were estimated to have an annual value of $2{,}757 \times 10^9$ US$. However, it is not the case that all wetlands always mitigate water-related problems, and understanding the role of wetlands within the hydrological cycle is crucial to understanding how they should best be managed and valued.

Global climate change is widely accepted to result in changes in the magnitude, timing, and duration of, among other things, temperature, precipitation, and sea level. As some of the key wetland hydrological drivers, these changes have the potential to affect most, if not all, aspects of wetland habitats. This is likely to be compounded by other pressures such as land management and abstraction, with the result that the character of many wetlands will change markedly. This in turn will affect their ability to provide hydrological services. This section explores these issues.

Wetland Hydrological Services

Wetlands occur throughout the world and in many different locations within the landscape, from headwater seeps situated at the upstream end of a catchment to tidal marshes at the coast. In all settings, wetlands interact with the movement of water and have varying levels of influence on the hydrology of the surrounding areas. Three categories of hydrological service have been defined in the Millennium Ecosystem Assessment (2005) and a broad indication of the possible magnitude of service provided by different types of inland and coastal wetland proposed (Table 1).

The hydrological services result from different properties of the wetland system, and in order to understand the processes involved in delivery of the services, a description of each is given below:

Table 1 Hydrological services of inland and coastal wetlands. A likely magnitude of hydrological servic is proposed for each wetland type. (Information from multiple sources including Millennium Ecosystem Assessment 2005)

		Hydrological services		
		Climate Regulation (Regulation of temperature, precipitation, and other climatic processes)	**Hydrological Regimes** (Groundwater recharge and discharge; storage of water for agriculture or industry)	**Natural Hazards** (Flood control; storm protection)
Inland Wetlands	**Permanent and temporary rivers and streams**	Low	High	Medium
	Permanent lakes, reservoirs	High	High	High
	Seasonal lakes, marshes and swamps, including floodplains	Low	Medium	High
	Forested wetlands, marshes, and swamps including floodplains	High	Medium	Medium
	Alpine and tundra wetlands	Low	Low	Medium
	Springs and oases	Low	Low	Low
	Geothermal wetlands	Low	n/a	n/a
	Underground wetlands including caves and groundwater systems	Low	Low	Low
Coastal Wetlands	**Estuaries and marshes**	Medium	Low	High
	Mangroves	Medium	n/a	High
	Lagoons, including salt ponds	Medium	Low	Low
	Intertidal flats, beaches and dunes	Low	n/a	Low

(*continued*)

Table 1 (continued)

		Hydrological services		
	Kelp	n/a	n/a	n/a
	Rock and shell reefs	Low	n/a	Medium
	Seagrass beds	Low	n/a	Medium
	Coral reefs	Medium	n/a	High

Climate Regulation

It is increasingly recognized that wetlands have an important role in regulating climate through promoting evapotranspiration which has a considerable capacity to equalize temperature differences in time and space (Cizkova et al. 2013). Evapotranspiration can promote cloud formation and subsequent rainfall. Analysis of data from the Inner Niger Delta has identified that during periods of inundation, there is greater cloud cover in the area surrounding the wetland and that this is associated with a 54% increase in the initiation of new storms compared to periods without inundation (Taylor 2010). This phenomenon, known as the "wetland breeze" effect, has the potential to influence the rainfall patterns over a considerable area around the wetland, and this in turn provides water for drinking and irrigation (Dadson 2013).

Hydrological Regimes

One of the most commonly cited attributes of wetlands is their ability to "act as sponges", soaking up water when it is in excess and releasing water when it is lacking. The very essence of a wetland as a wet place tells us that it has some ability to hold water. Wetlands can store water both as open water at the surface and as soil pore water in the subsurface, and as hydrological gradients between the wetland and its surroundings change, these stores either fill up or empty. Water moving through a wetland is often slowed by vegetation and this can further promote infiltration and storage. As the stores empty, the water released recharges surface and groundwater features in the surrounding landscape.

Natural Hazards

Floodplain wetlands and open water features have the potential to store water during times of flood and therefore reduce the amount of water arriving downstream at a point in time. The importance of these landscape features in contributing to an overall reduction in flood risk is increasingly recognised (e.g. natural flood management). Coastal wetlands, in particular mangroves and salt marshes, can similarly provide an important flood mitigation function as areas of vegetation can dissipate wave and tidal energy therefore reducing erosion of the coast line.

However it should be borne in mind that wetlands can only soak up water if they have available storage capacity and an already saturated wetland may provide little or no hydrological storage.

Quantifying Wetland Hydrological Services

Accurately quantifying the extent to which hydrological services occur is difficult in practice. Field measurements which capture, with enough spatial and temporal detail, the hydrological processes in and around a wetland can be very expensive and time consuming to collect. In an attempt to provide some scientific underpinning, Bullock and Acreman (2003) reviewed 169 studies of wetland hydrology and and analysed the 439 published statements relating to hydrological services contained within them. They summarized that

- the vast majority (81%) of statements conclude that wetlands either increase or decrease a particular component of the water cycle.
- 23 of 28 studies on floodplains show that floods are reduced or delayed.
- 30 of 66 studies on headwater wetlands show that floods are reduced or delayed.
- 27 of 66 studies on headwater wetlands show that wetlands increase flood peaks.

These findings highlight the importance of landscape position in determining the influence of a wetland on the water cycle and it is important that this is considered when quantifying the wetland hydrological services. Factors that should be considered include landscape position, hydrological conditions, characteristics of the wetland and management regime (Acreman and Holden 2013).

Global Climate Change

Although disputes still exist on the drivers of observed climatic trends, the scientific community generally accepts that the change over the past 100 years has been at a different pace to that experienced previously. Many climate models have been developed, each with a view to understanding processes and/or predicting the future climate under a range of different scenarios. The Intergovernmental Panel on Climate Change (IPCC) has sought to bring together the full range of model outputs in order to provide a robust set of predictions about the future. Junk et al. (2012) summarized predictions of the IPCC (Pachuari, 2007) regarding major changes in global climate until 2100 as follows: (1) increase in temperature affecting high latitudes more than tropical and subtropical regions, (2) changes in total precipitation and precipitation pattern, (3) a rise in sea-level of 20–60 cm, or even more, and (4) an increase in extreme climate events.

Temperature (and its influence on evapotranspiration), rainfall, and sea level are important drivers in the hydrological functioning of wetland systems, and changes in one or more will likely have a significant impact on the movement of water in and out of the wetland. Globally, there is spatial variability in the magnitude of predicted change, and in some areas pronounced temporal variability in the timing of some drivers is anticipated (e.g., in the UK, although the total annual rainfall is not predicted to changed markedly, more rain is expected to fall in winter and less in summer).

How a Changed Climate Might Affect Wetland Hydrological Services

Having investigated the aspects of wetlands that contribute towards delivery of hydrological services and the changes that might be expected in a future climate, the two elements are now brought together. This subject has been the subject of considerable research, for instance see Junk et al. (2012).

Climate Regulation

Evaporation from a wetland is closely related to the surface area of wet or moist conditions. If the area increases or decreases, due perhaps to changes in the extent or timing of rainfall events, there will likely be an impact on the evaporative fluxes from the wetland and subsequent promotion of cloud formation and rainfall. Taylor (2010) suggests that decreased evaporation resulting from a reduction in the flooded area of the Inner Niger Delta (in this case due to construction of a hydro-electric power scheme) could result in a reduction in rainfall at the regional scale.

Hydrological Regimes

In areas where the amount or timing of rainfall changes such that drought conditions become more prevalent, the structure of wetland soils may be negatively affected. Oxidation of soil organic matter can result in a reduction in water holding capacity which can further perpetuate soil drying and oxidation. As the wetland soils become less able to retain water, and drought conditions become potentially more severe there will be a reduction in groundwater recharge (Junk 2013). Changes in the duration and timing of very wet or dry conditions have the potential to bring about a shift in wetland plant species composition towards one more suited to the prevailing conditions, and this may in turn affect the hydrological regime as some species transpire at higher rates than others.

Natural Hazards

An increase in extreme climate events is likely to expose wetland habitats to conditions outside of their normal range. Prolonged and/or sudden inundation, and changes in the timing of inundation through the year have the potential dramatically change wetland species composition. The relative contribution of the wetland to reducing flood risk may be reduced as the wetland is overwhelmed by a large quantity of water arriving in a very short time. More frequent and severe coastal storms have the potential to damage the vegetation that provides a defense, leaving the coast more exposed and there will be a reduction in the ability of coastal wetlands to dissipate energy and provide useful defense against coastal storms.

Future Challenges

Wetlands have the potential to provide various hydrological services and these could play an vital role in hydrological regime as pressures from climate change increase. However degraded wetlands are unlikely to fulfill that role as effectively and it is important to understand activities that we can undertake in order to maximise the benefits provided by wetland habitats. For example, improved soil management might partially offset some of the adverse effects of climate change on groundwater resources and base flow to rivers (Holman et al. 2011).

It is also important that we continue to gather evidence of wetland hydrological services and use this both to raise awareness and manage expectations. Wetlands have a crucial role to play in mitigating flood risk, however on their own they are unlikely to provide the complete solution. If extreme events become more frequent, it may become increasingly difficult to convince the public of the benefits of a natural approach to dealing with hydrological hazards, and this may in turn lead to growing support for ecosystem unfriendly flood defense measures (Cizkova et al. 2013).

Scientifically robust monitoring of case studies should be encouraged and the information captured should be analysed and presented in a defensible manner.

Increasing efforts should be made to collect defensible evidence of the hydrological services of wetlands. Scientifically robust monitoring of case study sites should be encouraged, possibly using a range of incentive measures, and the information collected should be analyzed and presented in a defensible manner. Care should be taken not to exaggerate the role of wetlands, and unhelpful sweeping generalities should be replaced by a bespoke case-by-case approach (Bullock and Acreman 2003).

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Climate Regulation by Capturing Carbon in Mangroves

168

Daniel M. Alongi

Contents

Abstract

Mangroves play an important role in regulating climate by sequestering carbon within soils and to a lesser extent in forest biomass, as well as exchanging carbon dioxide with and emitting methane to the atmosphere. The rate of soil accretion averages 5.8 mm yr^{-1} with most measurements between 0 to 2 mm yr^{-1}. The median is 3 mm yr^{-1} with one standard error of 1.0 mm yr^{-1}. The average carbon sequestration rate is 171 g C_{org} m^{-2} yr^{-1} with a median of 103 g C_{org} m^{-2} yr^{-1}. Assuming a global area of 137,760 km^2, and using the median value, carbon sequestration in mangroves equates to 24 Tg C_{org} yr^{-1}. Assuming a destruction rate of 1-2% yr^{-1}, we can estimate a loss of carbon equivalent to 5 to 11% to recent estimates of global deforestation. These losses also offset 23-49% of the carbon sink in the global ocean continental margins. The range of these losses underscores the global consequences of continuing mangrove losses to the global carbon cycle.

Keywords

Mangroves · Climate regulation · Carbon

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Introduction

Mangrove forests and their associated waterways play an important role in climate regulation by sequestering carbon within soils and to a lesser extent in forest biomass, as well as by exchanging carbon dioxide with and emitting methane to the atmosphere. Mangrove ecosystems are the only woody halophytes that live along the world's subtropical and tropical coastlines, occupying a global area of 137,760 km^2. Mangrove forests are structurally simple, harboring few tree species and lacking an understory, but their standing biomass can be immense in equatorial regions, equivalent in weight to many tropical rainforests. These ecosystems are true ecotones, having both terrestrial and aquatic components, but they have also evolved a number of unique structural and functional adaptations, such as physiological mechanisms to tolerate salt and aerial roots that enable the trees to respire in anoxic, waterlogged saline soils.

Mangrove forests develop along coastlines and in estuaries where near-horizontal topography coincides with sea-level, so a relatively stable period of sea level is a prerequisite for forest development. The response of mangroves to environmental change is therefore often indicative of past coastal changes, especially sea level. The reality is that mangroves and their ability to accumulate carbon ebb and flow at considerable speed, as these tidal forests are dynamic, living in nonlinear, nonequilibrium conditions at the land-sea boundary. Mangroves therefore persist at timescales over which shoreline evolution and changes in sea-level occur (Woodroffe 2002). Under static conditions, the soil surface of mangroves accretes asymptotically until the forest floor becomes raised above tidal range. Under a constantly rising sea level, the rate of soil accretion matches the pace of sea-level rise, whereas when sea-level fluctuates, the forest floor accretes with time above the mean tidal range. Under conditions where subsidence occurs, mangrove development is complex; the forest floor accretes back to tidal range if sea-level is stable but if subsidence is episodic. Under rising sea-level, if there is no subsidence or change in sediment volume, the forest floor is set back or abandoned; with episodic subsidence with rising sea-level, the overall pattern is of net accretion.

The ability of mangrove forests to develop, and thus capture and store carbon, is dependent on rate of sea-level change relative to changes in accretion and subsidence. Mangroves are currently keeping up with the pace of sea-level rise (Alongi 2008), but not all mangroves accrete, with geomorphological setting and water circulation being important drivers of net soil accumulation and net carbon sequestration (Woodroffe 2002). A mangrove forest located mid-estuary may be rapidly accumulating soil carbon, but this material may be coming from mangroves or other ecosystems that are eroding upstream, such that at the whole-estuary scale, there is no net accumulation. At the within-forest scale, rates of soil carbon accumulation decrease with tidal height up to the transition point between mangrove and lowland forest where there is no longer any significant accretion, except for possible litter accumulation and burial. These processes vary over time as well as space, with primary temporal drivers being seasonal changes in precipitation and subsequent river runoff and seasonal changes in marine inputs.

Mechanisms of Carbon Capture and Storage

Forested ecosystems capture carbon in two ways: (1) via carbon fixation and growth of tree biomass (wood) and (2) via accumulation of carbon in soil accumulating with time. Mangroves are usually highly productive forests and, like other forests, vary in size and age and thus vary in rates of primary production and carbon balance. Mangrove forest dynamics are similar to other forests in that there is an initial period of early rapid growth during colonization and early establishment followed by a slow decline in growth rate into maturity and senescence (Fromard et al. 1998). The mature old-growth stage can be prolonged in some forests such that an alternate succession state is reached as the climax stage is reset by successive disturbances. Mangroves may therefore constitute a carbon sink for up to a century if left relatively undisturbed. However, despite the huge above-ground carbon stocks of some equatorial forests, on average, mangrove inventories (Donato et al. 2011; Kauffman et al. 2011; Alongi 2012) show that most carbon is stored below-ground with 75–95% of tree carbon stored below-ground in dead roots (Alongi et al. 2003, 2004). Most above-ground biomass is eventually lost due to clear-cutting and human use, decomposition, and export to the adjacent coastal zone.

Over the long-term, carbon is captured and stored below-ground and, under the right conditions, as peat. Accumulation of carbon in soil depends on a number of factors including location of the forest in relation to the open coast, distance to adjacent aquatic habitats, tidal amplitude, forest position in the tidal zone, and primary productivity. Mangroves actively capture silt, clay, and organic particles; they are not just passive importers of fine particulates (Mazda et al. 2009). Until slack water, turbulent wakes created by tree trunks, prop roots, and pneumatophores maintain particles in suspension, but most small flocs and free particles settle just before slack high tide. Most flocs and particles are retained within the forest, despite the pull of the ebb tide, as turbulence and water motion necessary for their resuspension is inhibited by the high tree density. Mangrove waters have high suspended loads due to movement of the turbidity maximum zone where incoming bottom flow meets outward river flow within an estuary. Tidal mixing and pumping within this zone facilitates flocculation and resuspension of particles. As these flocs move into the forest while flooding, turbulence generated by tidal flow around the trees helps to maintain flocs in suspension. Rapid settling of particles is facilitated by the sticking of microbial mucus on the soil surface and by pelletization by invertebrate excreta.

Rates of Carbon Capture and Storage

The rate of soil accretion in mangrove forests averages 5.8 mm year^{-1} with most measurements ranging from 0.1 to 2.0 mm year^{-1} (Fig. 1). The median is 3.0 mm year^{-1} with one standard error of 1.0 mm year^{-1}, based on a sample size of $n = 229$.

A few measurements show net erosion (negative value) or massive accretion in highly impacted estuaries. Soil accretion rate is a function of frequency of tidal inundation. More frequent inundation of particle-laden water facilitates more particle

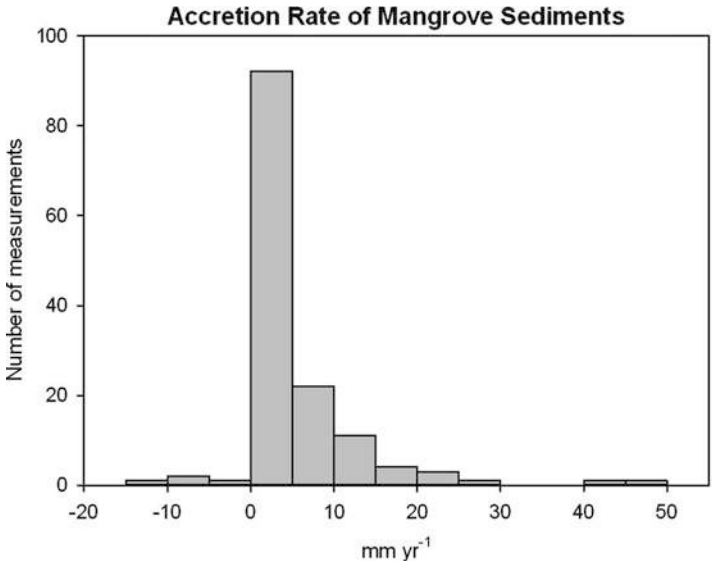

Fig. 1 Soil accretion rates measured in various mangrove forests worldwide (Updated from Alongi (2011, 2012))

settlement; mangroves in high intertidal areas experience less soil accretion than forests located closer to the sea, so there is an overall pattern of declining sedimentation landwards. Often overlooked is the role of below-ground roots in vertical accretion of the forest floor. Surface growth of microbial mats and turf algae as well as litter and felled wood also contribute to net accretion. In some forests, these biological forces can contribute more to vertical accretion than accumulation of particles via tides (McKee 2011). Natural subsidence plays a key role in overall rates of soil accretion, being important in estimating the susceptibility of mangroves to changes in sea level. Over long timescales, rates of soil accretion vary in relation to climatic variability (Krauss et al. 2010). Although most mangroves are accreting soil and carbon, on some islands in the Pacific and in the Caribbean, sedimentation rates are slower than rates of sea-level rise, although accretion rates at some of these sites are higher than eustatic sea-level rise (Sanders et al. 2010).

The average global burial rate for soil carbon in mangroves is 24 Tg C year^{-1} which equates to 171 g C m^{-2} year^{-1}, with values ranging from 10 to 920 g C m^{-2} year^{-1} (Fig. 2) and with a median burial rate of 14.3 Tg C year^{-1} (= 103 g C m^{-2} year^{-1}).

The standard deviation of these data exceeds the mean, reflecting the high level of variability in carbon burial rates in mangroves globally. Burial rates exceeding the mean and median values are in mangroves inhabiting estuaries bordering catchments in southern China and in Southeast Asia that are heavily used and impacted by human activities (Fujimoto 2004; Alongi 2008). A more recent study (Breithaupt et al. 2012) estimated a similar mean burial rate of 26 TgC year^{-1} (=163 g C m^{-2} year^{-1}) for mangroves globally. Both sets of recent estimates do not differ greatly from the first attempt to estimate carbon burial rates in mangroves (Twilley et al. 1992).

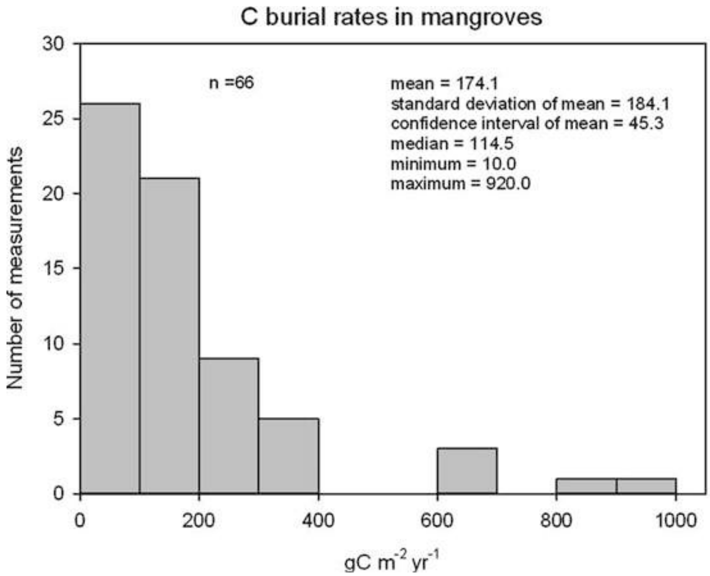

Fig. 2 Annual rates of carbon burial in various mangrove forests worldwide. Units for statistics are g C m^{-2} $year^{-1}$ (Updated from Alongi (2011, 2012))

Global Significance

A simple scaling up of the mean carbon burial rates to total mangrove area equates to a global sequestration rate of 13.5 GtC $year^{-1}$. The same exercise for boreal, temperate, and other tropical forests extrapolates to 451.1, 327.6, and 422.4 GtC $year^{-1}$, respectively (IPCC 2003). Mangroves therefore account for about 3% of carbon sequestered by the world's tropical forests although accounting for <1% of total tropical forest area. However, if mangroves are disturbed, their high area-specific carbon stocks suggest the potential for significant greenhouse gas emissions. For example, deforesting mangroves on peat soils (Lovelock et al. 2011) results in CO_2 emissions (2900 tC km^{-2} $year^{-1}$) comparable to rates estimated from collapse of terrestrial peats (150–3200 tC km^{-2} $year^{-1}$). A plausible range of CO_2 emissions of 112 to 392 tC ha^{-1} from cleared mangroves and soils gives a global emissions range of 0.02 to 0.12 PgC yr^{-1}which is equivalent to about 2–10% of global deforestation emissions and up to 50% of emissions from the world's tropical peatlands (Donato et al. 2011).

The contribution of mangrove to global forest carbon sequestration is small, but from a marine perspective, the mangrove contribution is impressive. Compared to other marine ecosystems mangroves, although accounting for only 0.05% of total coastal ocean area, contribute roughly 14% to carbon sequestration in the global ocean (Table 1).

Table 1 Global contribution of mangroves to carbon sequestration in the global coastal ocean (Alongi 2012)

Habitat	Area (10^{12} m^2)	Sequestration rate (g C m^{-2} year^{-1})	Global sequestration rate (Tg C year^{-1})
Mangroves	0.14 (0.5%)	171	24 (14%)
Salt marshes	0.22 (0.8%)	150	33 (20%)
Seagrass beds	0.3 (1.1%)	54	16 (10%)
Estuaries	1.1 (4.0%)	45	50 (30%)
Shelves	26 (93.6%)	17	44 (26%)
Total			167

Source: Reprinted with permission from Future Science Publishers

The average carbon burial rate in mangroves is much greater than that from all other coastal habitats, except for salt marshes. Mangroves thus contribute disproportionately as a carbon sink, having the highest per area rates of carbon capture and storage compared with all terrestrial and other marine ecosystems.

Future Challenges

There are large uncertainties in these sequestration estimates and future challenges remain to better constrain estimates of mangrove carbon capture and storage. First, within forests, a number of methods are used to measure soil accretion; some are either highly inaccurate and others reflect only real-time, not time integrated rates of accumulation. Analysis of excess^{210}Pb and^{137}Cs derived from atmospheric fallout from atomic bomb testing and accumulating in soils provides long-term estimates of accumulation and a chronology of sedimentation up to a century, but this method also has pitfalls. Second, not all mangroves accumulate carbon and rates of soil accretion, when measurable, vary greatly among sites for a wide variety of reasons, some of which remain unclear. Third, carbon balance including carbon burial is linked to forest age and location, so uncertainty is endemic; there is natural variability inherent in all carbon processes. Finally, a large proportion of mangrove carbon is unaccounted for in global budgets (Bouillon et al. 2008), thus further constraining our level of certainty regarding carbon sequestration rates. These issues have management implications. For instance, in either a PES or REDD+ scheme, ecosystem constraints and uncertainties of sequestration potential must be considered and factored into the design, timeframe, and execution of climate regulation projects.

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Greenhouse Gas Regulation by Wetlands 169

Jan Pokorný, Hanna Huryna, and David Harper

Contents

Abstract

Wetlands are unique and productive ecosystems that perform essential ecological functions. They cover only 6% of the earth's surface, yet they play a crucial role in maintenance and improvement of water quality; controlling soil erosion and floods, regulating the hydrological cycle and retention of nutrients and carbon. Wetlands also contribute to local climate regulation through distribution of incoming solar energy, by transferring solar energy from latent heat flux (cooling) into sensible heat flux (warming of air). The amount of water vapour, as a greenhouse gas, found in plant stands and in the atmosphere is many times higher than the amount of CO_2 and it changes dramatically across time and space. Water exists on the Earth in three phases and its transition between these phases is linked with uptake or release of high amounts of energy. The cooling effect of evapotranspiration is introduced in terms of solar energy and water vapour fluxes. The effect of wetlands on the daily dynamic of surface temperature is shown by thermographic and visible pictures of the mosaic of a cultural landscape with wetlands. We thus demonstrate that wetlands cool

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landscape and moderate daily extremes of temperature; in this way we seek to quantify the global role of wetlands in regulation of greenhouse gases and influence on local climate.

Keywords
Climate regulation · Evapotranspiration · Greenhouse gases · Surface temperature · Transpiration efficiency · Wetlands

The Balance of Greenhouse Gases in Wetlands

There has been increased interest in understanding the role that wetlands play in the regulation of greenhouse gases. The dynamics of greenhouse gas exchange are largely determined by site-specific conditions including hydrology, soil type, vegetation, and meteorological and climatic conditions. Wetlands, just as other ecosystems, may act as carbon dioxide (CO_2) sinks in some periods, and as sources in others, depending on the meteorological conditions (Čížková et al. 2013). The emission of methane (CH_4) and nitrous dioxide (N_2O) from wetlands is similarly variable in time.

Čížková et al. (2013) provided an overview of case studies that focused on the balance of greenhouse gases in different wetland types. They found that peatlands were by far the most important of all wetland ecosystems with regard to affecting the global balance of greenhouse gases and globally represent a highly important store of carbon, sink for CO_2, and a significant source of atmospheric CH_4 (from the point of view of its importance for the greenhouse effect). In general, N_2O emissions are small in natural peatlands (Joosten and Clarke 2002). In addition to actively growing peatlands (mires), littoral wetlands with abundant plant cover such as reed (*Phragmites australis*) in Central and North Europe can be important sinks for carbon. Floodplains can also accumulate organic matter and carbon if floods are maintained and the river-floodplain connectivity allows the plant communities (especially riparian woodlands) to develop in response to the ecohydrological cycle.

Two types of impact considerably affect the greenhouse gas balance of wetlands: changed hydrology and nutrient enrichment. More frequent summer droughts increase the frequency of situations under which wetlands, especially peatlands, act as sources of CO_2 to the atmosphere due to mineralization. At the same time, CH_4 emissions decrease. There is also evidence that peatlands reclaimed for agricultural use are releasing significant amounts of N_2O because they have become enriched with mineral nutrients including nitrogen (Couwenberg 2011). Long-term nutrient enrichment of wetlands with organic soils can also promote CO_2 efflux. Eutrophication of permanent wetlands associated with standing waters can promote anaerobic decomposition processes including CH_4 production (Fig. 1).

Wetlands have been both taking up and releasing greenhouse gases continuously since their formation, and thus their influence on the atmosphere must be modeled

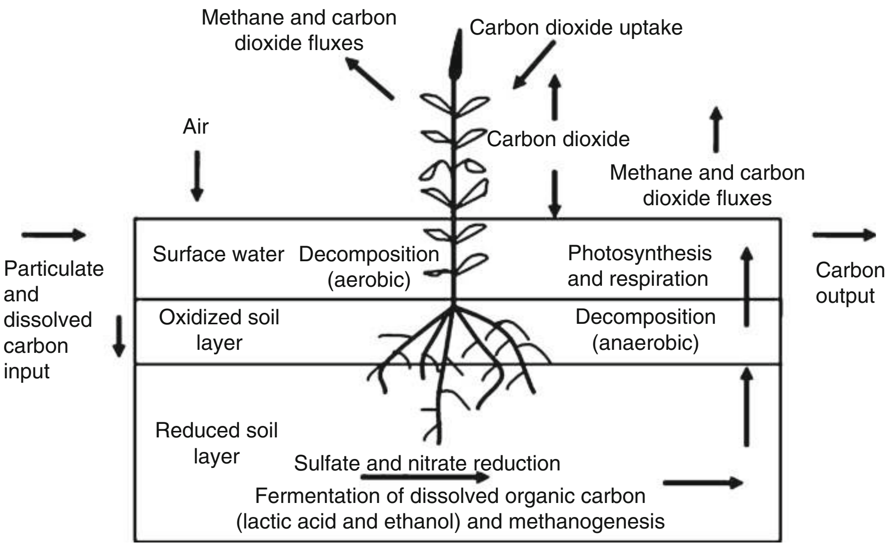

Fig. 1 Schematic diagram showing the major components of the carbon cycle and conditions in the root zone (With permission from Kayranli et al. 2010)

over time. When this is considered, the sequestration of CO_2 in peat outweighs the CH_4 emissions. In terms of greenhouse gas management, the maintenance of large carbon stores in undisturbed peatlands should be a priority.

Wetlands may initially accumulate organic matter at higher rate than it is decomposed, but as this material accumulates, the continued decomposition of steadily increasing amount of peat or sediment means that carbon loss is also progressively increasing (Clymo 1984). Eventually mean carbon input roughly balances the rate at which carbon is released. Marshes and swamps reach this point relatively rapidly: perhaps a few hundred to a maximum of a few thousand years. Peatlands, on the other hand, may take thousands of years to reach a "steady state" of carbon losses balanced by carbon inputs. Carbon accumulation in wetlands can be highly sensitive to environmental conditions such as temperature, precipitation, fire, or flood. Studies in Finland and Canada found that bogs and nutrient-poor fens in general accumulate more carbon (ca. 20–25 gC.m^{-2}.a^{-1}) than more mineral-rich fens (ca. 15–20 gC.m^{-2}.a^{-1}) (Tolonen et al. 1996; Robinson and Moore 1999). Nowadays, the world's wetlands may be net carbon sinks of about 830 Tg CO_2 year$^{-1,}$ with an average of 118 g-C m^{-2} year^{-1} net carbon retention (Mitsch and Hernandez 2013).

Carbon release can be elevated by ten or more times for months to years immediately after drainage (of the order of 250–1,000 gC m^{-2}.a^{-1}, Maltby and Immirzi 1993) and decreases over time as more labile carbon compounds are decomposed and more refractory material remains. Long after drainage, for more than 60 years, for example, however, stored peat can continue to decompose.

Exchange of Water and CO_2 in Plant Stands

Most plant tissues contain large amounts of water. The biomass of non-woody tissues typically is made up of 80–95% water. Most water taken up by roots is transported through plants in the soil-plant-atmosphere continuum (SPAC) and transpired into the air.

The cooling process of transpiration is often considered a side effect rather than a mechanism to control leaf temperature (Lambers et al. 2008). Transpiration is also perceived as a rather negative process. Plant physiologists and hydrologists may use negative terms such as "transpiration loss" and "evapotranspiration losses." Transpiration efficiency (TE) is defined as the amount of water used in transpiration per unit of dry matter produced. TE normally reaches a value of several hundred kilograms of water consumed per kilogram of dry biomass produced. The amount of water molecules exchanged by plants is at least two orders of magnitude higher than the amount of carbon dioxide fixed in biomass.

The amount of water vapor, as a greenhouse gas, found in plant stands and in the atmosphere is many times higher than the amount of CO_2 as a greenhouse gas. Moreover, it changes dramatically across time and space. For example, air saturated with water at 21 °C contains 18 g m^{-3} of water vapor, i.e., 22,400 ppm. Air saturated with water at 40 °C contains 50 g m^{-3} of water vapor, i.e., 62,200 ppm. The amount of water vapor in air is often two orders of magnitude higher than that of CO_2. The content of water vapor in the atmosphere is highly variable, and furthermore, water exists in three phases (solid, liquid, and gaseous). The transitions between these phases are linked with the uptake or release of high amounts of energy and with immense change of volume (18 ml of water liquid forms 22,400 ml of water vapor). The energy absorption spectrum of water is broader than that of CO_2 (Sondergard 2009).

Cooling Effect of Evapotranspiration

Climate change and global warming are widely believed to be caused only by an increase in CO_2 concentration from 250 to 390 ppm. Novel recent research, however, highlights the dynamic role of water vapor in climate change, with its concentration two orders of magnitude higher than that of other greenhouse gases. The implication of this research is that human landscape management affects the behavior of water vapor and its role in the dissipation of solar energy, in a much more important way than formerly appreciated (Pokorný et al. 2010).

This research has focused on wet meadows in the Czech Republic, which evapotranspired about 7 mmol m^{-2} s^{-1} (i.e., 126 mg m^{-2} s^{-1}) during a sunny afternoon, converting about 315 W of energy per square meter of its surface into latent heat flux (Rejšková et al. 2010). The wetland, which covered an area of about 4 km^2, evapotranspired about 500 kg of water per second, which is equivalent to the flow rate of a small river. This invisible stream represents the latent heat flux of approximately 1,260 MW. Thus, this ecosystem regulates the temperature through energy and water fluxes with a power equivalent to that of a moderately large power station. If a wetland is situated in the middle of a dry landscape, it is predestined to function as a water funnel, and all evaporated water which runs through it is locally lost via rapid convective movement. In drained or dry landscapes, wetland ecosystems thus act as “wet islands,” important both for their conservation value and for their important hydrological function (in addition to their hydrologically dependent nutrient processing).

The drainage of large areas of natural vegetation and the loss of their latent heat function causes surprisingly large amounts of sensible heat to be released into the atmosphere. A drop in evapotranspiration by 1 L m^{-2} (equivalent to about 700 Wh) is capable of increasing the daily flux of sensible heat about 40 times more effectively (by 70 W) than the quoted effect of greenhouse gases [radiative forcing, Intergovernmental Panel on Climate Change (IPCC)]. For example, a drop in evapotranspiration of 1 mm over the territory of the Czech Republic (79,000 km^2) within a single day, releases an amount of sensible heat comparable to the annual production of electric energy from all Czech power plants (about 60,000 GWh).

The Czech study also measured the daily dynamics of radiation surface temperature and air temperature of different land cover types in a temperate, “cultural” landscape and their consequences for the local climate (Hesslerová et al. 2013). Seven localities with different land cover types were chosen in Trebon Biosphere Reserve, Czech Republic, Central Europe. A combined method of airship thermal scanning of Ts (radiation surface temperature) and ground measurement of thermodynamic Ta (air temperature measured in a meteorological screen at 2 m height) was used (Fig. 2). The localities differed markedly in both the values and the dynamics of Ts and Ts–Ta. In the early afternoon, the difference in Ts between the different land covers reached almost 20 °C. Ecosystems with nonfunctional or no vegetation largely resembled the asphalt surface, whereas ecosystems covered with dense, bushy, or tree vegetation showed relatively well-balanced daily temperature dynamics with low temperature extremes and a slow temperature morning increase or afternoon decrease. Ts−Ta at the peak solar irradiance ranged between −1 °C at the forest and 14–17 °C at the dry harvested meadow and the asphalt surface, respectively (Fig. 3). Therefore surface radiation temperature (Ts) can be considered as a measurable indicator of ecosystem and landscape functioning, and the importance of functional vegetation for local climate should also be considered.

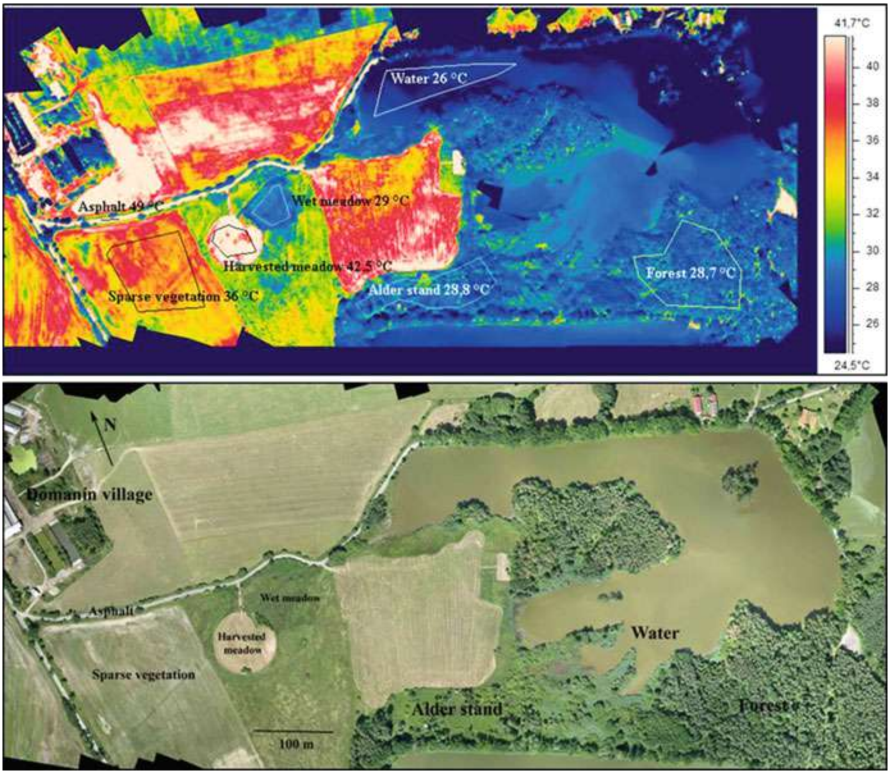

Fig. 2 Surface temperature of a "cultural" landscape on summer sunny day in Třeboň Biosphere Reserve (Czech Republic) at 2 PM, taken by thermographic and visible cameras carried by an airship

Future Challenges

The feedback between vegetation, surface temperature, water, and climate are crucial in landscape management with important implications for climate regulation and climate change. The importance of wetlands for the regulation of greenhouse gases and local climate regimes has long been assumed; the exact extent and effect have not, however, largely due to the absence of specific measurements, such as those referred to above. The measurements that have been undertaken illustrate the important role that wetlands have in regulating greenhouse gases and climate. We strongly suggest that wetland restoration, as mitigation for the predicted impacts of climate change, be placed on the agenda of climate scientists as well as conservation scientists.

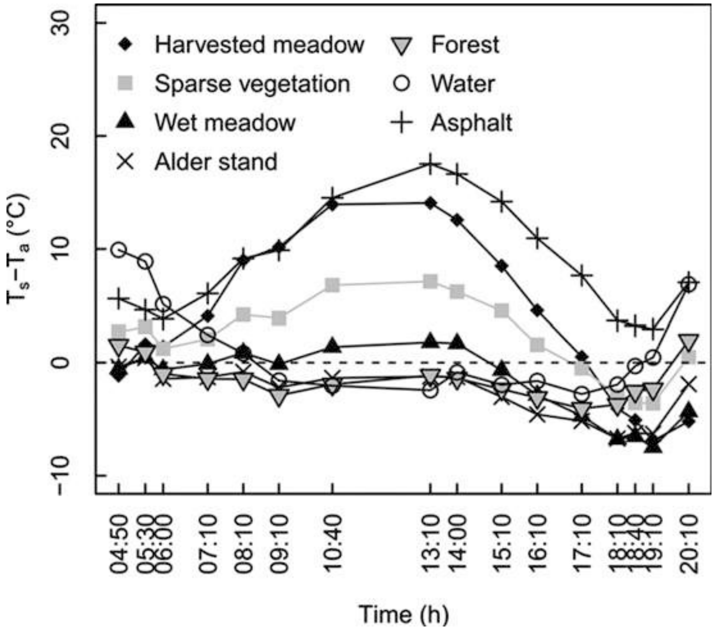

Fig. 3 Temperature differences $T_s - T_a$ between surface T_s and air temperature T_a (at 2 m above ground under white screen) at all the studied localities (With permission from Hesslerová et al. 2013)

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Natural Hazard Regulation: Overview

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Robert J. McInnes

Contents

Abstract

Natural hazards can come in many forms and can be both rapid and catastrophic and slow-moving and chronic. While major hazards such as earthquakes, drought, and famine often draw significant media interest, a much greater proportion of the world's population are at risk from chronic issues such as a violent conflict, illness, and hunger. Evidence indicates that over 85% of all hazard-related deaths are from of slow onset events rather than rapid, cataclysmic disasters. Therefore in order to understand the role of 25 wetlands in natural hazard regulation it is also necessary to be cognizant of the socio-political landscapes within which wetlands find themselves.

Keywords

Drought · Earthquake · Hazards · Regulating services

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Introduction

Natural hazards can come in many forms and can be both rapid and catastrophic and slow-moving and chronic. While major hazards such as the earthquake, drought, and famine often draw significant media interest, a much greater proportion of the world's population are at risk from chronic issues such as a violent conflict, illness, and hunger. Table 1 demonstrates that over 85% of all hazard-related deaths are from of slow onset events rather than rapid, cataclysmic disasters.

One of the crucial points about understanding why disasters happen is to acknowledge that natural disasters act in tandem with social, political, and economic environments and that the impact, in terms of human deaths and disruption, are a product of a combination of these factors (Blaikie et al. 2004). Too often the emphasis is on the "natural" element of the disaster and not on the social or political context within which it has occurred. Therefore, in order to understand the role of wetlands in natural hazard regulation it is also necessary to be cognizant of the sociopolitical landscapes within which wetlands find themselves. Therefore disasters should be considered to be a complex mix of natural hazards and errant human actions.

Even within the most complex sociopolitical environments, wetlands still have the potential to regulate the frequency and magnitude of natural hazards. There is a strong body of literature to demonstrate that, with the possible exception of volcanic eruptions, wetlands can play a significant role in regulating the impact of all the natural hazards shown in Table 1 and also in increasing resilience and recovery postdisaster (Millennium Ecosystem Assessment 2005; Wetlands International 2011). Wetlands can also contribute significantly to a variety of mitigation options related to natural hazards including integration within management plans, land-use planning, and climate adaptation; modification of the magnitude and frequency of hazards; providing important infrastructure; mitigation measures (*stricto sensu*); and financial incentives (adapted from Bouwer et al. 2014).

Table 1 Hazard type and their contribution to deaths, 1990–1999 (source data from Blaikie et al. 2004)

Hazard type in rank order	Deaths (%)
Slow onset	
Famines – drought	86.9
Rapid onset	
Floods	9.2
Earthquakes and tsunami	2.2
Storms	1.5
Volcanic eruptions	0.1
Landslides	<0.1
Avalanches	Negligible
Wildfires	Negligible

Wetlands and Hazard Regulation

Disasters and the environment are inextricably linked in two principle ways. Disasters resulting from extreme natural events can generate adverse environmental consequences which affect not only people in urban and peri-urban areas but also the wider ecosystems in which they live (Mainka and McNeely 2011). Similarly, degraded environments, and especially wetlands, can cause or augment the negative impacts of disasters. “Healthy” and sustainably managed ecosystems both reduce vulnerability to hazards by supporting livelihoods, while acting as physical buffers to reduce the impact of hazard events, and also enhancing postdisaster recovery (Sudmeier-Rieux and Ash 2009).

Wetlands have a key role to play in mitigating different potential disasters. Often the role of wetlands in hazard regulation can also provide an opportunity to deliver other benefits to local communities, such as improved water quality, recreation, or livelihoods. Examples of the role of wetlands and the benefits that they can provide are provided for a range of hazards.

Flood Regulation

In urban areas, efforts are being made to restore wetlands in order to provide sustainable solutions for flood risk management. Numerous cities around the world are integrating wetlands as “blue” or “green” infrastructure in order to deliver multiple benefits for local communities (Wade and McLean 2014). Mayesbrook Park lies in a densely urban area of East London in the Borough of Barking and Dagenham. Five of its 17 wards are ranked within 10% of the most deprived wards in England. The area is characterized by relatively high unemployment; low income per household; and high rates of teenage pregnancy, cancer, and heart disease. The Mayes Brook which runs through the Park is severely degraded, suffering from poor water quality and impoverished biodiversity and places several residential and commercial properties at risk from flooding. A restoration scheme for the Park and the Brook considered how the physical restoration of the river and associated wetlands could not only improve water quality and ecological condition but could provide socioeconomic uplift for the local community while reducing flood risk and improving resilience to climate change. An assessment of the economic benefits associated with the restoration scheme demonstrated that an investment of £3.84 million yielded a lifetime benefit-to-cost ratio of approximately 7:1. Gross annual benefits delivered through the range of ecosystem services provided has been estimated at approximately £880,000. Of this the regulating services, including reduction of flood risk and climate regulation, yielded a gross annual return of approximately £28,000. However, the cultural services, including recreation, social relations, and education, return a gross annual value of approximately £820,000 demonstrating how wetland restoration which provides hazard regulation can also enhance substantially human well-being in the urban environment (Everard et al. 2011).

Global sea levels have risen throughout the twentieth century and they continue to rise today (Nicholls and Cazenave 2010). With rising sea levels the vulnerability of coastal communities to flooding and storm surges increases. In order to mitigate current and future threats from coastal flooding due to storm surges and to increase resilience increasingly, the restoration of coastal wetlands is being advocated as a sustainable hazard regulation strategy (Nicholls et al. 1999).

The Muthurajawela Marsh covers some 3,086 ha in the coastal area approximately 10–30 km from Colombo, Sri Lanka. More than 300,000 people live in close proximity to the marsh with at least 5,000 people residing within the coastal wetland complex. The location of Muthurajawela Marsh in a rapidly developing urban area makes it an extremely vulnerable ecosystem. Large parts of the marsh system have been altered, through drainage, pollution, and hydrological modification. Traditionally, land-use planning processes have paid little heed to the need to maintain green spaces for Sri Lanka's city-dwelling populations, and have almost always resulted in development decisions which have taken place at the cost of the few remaining urban and peri-urban conservation zones (Emerton and Kekulandala 2002).

As a result of growing concern regarding the degradation of the coastal Marsh, in 1989 the Sri Lankan government decided to freeze all public and private sector development proposals until an environmentally sound Masterplan was developed for the Muthurajawela Marsh. The Masterplan was published in 1991 resulting in a land-use strategy being proposed and implemented for the future, based on dividing the Muthurajawela-Negombo area into various development and conservation zones. Essential to establish a clear rationale for land-use zonation was a credible economic assessment of the various benefits delivered by the Marsh. This analysis demonstrated that the gross annual value of the Marsh was in excess of US$8 million (Table 2).

Table 2 Economic value of Muthurajawela Marsh (source data from Emerton and Kekulandala 2002)

	Value (US$/year)
Flood attenuation	5,394,556
Industrial wastewater treatment	1,803,444
Agricultural production	336,556
Support to downstream fisheries	222,222
Firewood	88,444
Fishing	69,556
Leisure and recreation	58,667
Domestic sewage treatment	48,000
Freshwater supplies	42,000
Total Economic Value	**8,072,111**

The Masterplanning process for the Muthurajawela Marsh has demonstrated that:

- There is a clear economic incentive to maintain the integrity of the ecosystem.
- The main economic value of the Marsh was the reduction in flood risk.
- That land-use zonation should be informed by economic assessment of wetland ecosystem services.
- Future degradation of the Marsh will compromise the welfare of the urban residents and increase their exposure to flood hazards.

One of the highest profile cases of the role of wetlands in hazard regulation has been the Mississippi River Delta following the impact of Hurricane Katrina on the Mississippi–Louisiana coastline of the USA in August 2005. The hurricane sustained winds in excess of 125 mph and generated a storm surge in excess of 10 m in height. The associated flooding was most severe in communities where the levees and floodwalls failed and previous wetland buffers had already been degraded and lost. Hurricane Katrina left a trial of havoc across the Mississippi River Delta, affecting an area greater than 90,000 square miles and impacting over two million people (Batker et al. 2010).

Since the 1930s, the lower Mississippi River has been engineered and reengineered, constricting the flow through the construction of levees and effectively cutting the channel off from its floodplain. Immense volumes of sediment and trillions of gallons of freshwater have been channeled into the Gulf of Mexico and off the edge of the continental shelf instead of being allowed to accrete new land within the delta area. It has been estimated that through a combination of subsidence, reduced sediment supply, and rising sea levels, the Mississippi Delta has lost some 1.2 million acres of land since 1930. Despite human interventions, estimates suggest that the Mississippi River Delta ecosystems provide economically valuable services including hurricane storm protection, water supply, climate stability, food, furs, habitat, waste treatment, and other benefits worth at least \$12–47 billion/year to the American economy. These annual benefits provide a vast amount of value to people across time. Estimates of the present value of the benefits from 11 Mississippi Delta ecosystem goods and services are between \$330 billion and \$1.3 trillion (3.5% discount rate). A recent report (Batker et al. 2010) has described three scenarios:

1. A “do-nothing” approach which will cost at least \$41 billion in damages
2. A “hold the line” scenario avoids the \$41 billion, but without generating any additional benefits
3. A “sustainable restoration” option, involving large-scale controlled diversions of water and sediment which will reduce the rate of land loss, improve fisheries, and protect urban settlements and essential infrastructure by increasing the land available to buffer hurricanes, will avoid \$41 billion in losses and secure \$21 billion in benefits, providing \$62 billion in present value.

Earthquakes and Tsunami

Occupying a harsh environment subjected to daily tidal changes means that coastal mangroves forests are fairly robust and highly adaptable systems. Anecdotally there have been suggestions that intact mangrove forests can provide a degree of protection from catastrophic events, such as tsunamis triggered by offshore earthquakes (Ewel et al. 1998). The level of protection will depend on a variety of factors including the magnitude of the event, the environmental setting, the width of the forest, the slope of the forest floor, tree density, tree diameter, proportion of aboveground biomass vested in root, tree height, soil texture, forest location (open coast vs. lagoon), type of adjacent lowland vegetation and cover, presence of foreshore habitats (seagrass meadows, coral reefs, dunes), size and speed of tsunami, distance from tectonic event, and angle of tsunami incursion relative to the coastline (Alongi 2008).

The Indian Ocean tsunami of December 2004 is considered to be the most devastating in history, killing more than 283,000 people and generating wave heights in excess of 30 m (Alongi 2008). Several studies have attempted to assess the role of the mangrove forests in mitigating the impacts of this catastrophic event. In southeastern India, the Andaman Islands and Sri Lanka initial postimpact surveys indicated that mangroves provided significant protection against the full impact of the tsunami (Danielsen et al. 2005; Chang et al. 2006). Further studies along the Tamil Nadu coastline also suggested that impacts on human infrastructure located directly behind mangroves was subject to a lower level of destruction. While caution needs to be taken when making sweeping generalizations about the role of mangrove forests in protecting coastal communities and infrastructure from tsunamis, there is some evidence that these important coastal wetland systems offer a degree of natural hazard regulation.

Landslide Regulation

The Nilgiris District in the Western Ghats has experienced an increase in the frequency of landslides in recent years. In a major event in October and November 1978 heavy rains triggered more than 100 landslides resulting in the deaths of 90 people. Significant landslide events have occurred in 1993, 1995, 2002, 2007, and again in November 2009 when over 80 people died and wide-ranging damage to property resulted (Kumar and Bhagavanulu 2008). While rainfall is a significant factor, there is increasing evidence that unplanned tea plantations and associated deforestation and drainage of wetlands in the Nilgiris District may have degraded soils and ultimately increased the likelihood of landslides. One study estimated that over a quarter of all forests have been cleared in the Western Ghats between 1973 and 1995, increasing rates of soil erosion and run-off (Ganapathy et al. 2010).

In order to reduce the risk to human life and damage to infrastructure, increasing efforts are being made to improve both landslide risk planning, through the engagement with local communities, scientists, and planners, and ecosystem management

Table 3 Relative importance of different wetland types for natural hazard regulation (information adapted from Millenium Ecosystem Assessment 2005)

	Permanent and temporary rivers and streams	Permanent lakes, reservoirs	Seasonal lakes, marshes, swamps, including floodplains	Forested wetlands, marshes, swamps, including floodplains	Alpine and tundra wetlands	Springs and oases	Underground wetlands, including caves and groundwater systems	
Inland wetlands	**M**	**H**	**H**	**M**	**M**	**L**	**L**	
	Estuaries and marshes	Mangroves	Lagoons, including salt ponds	Intertidal flats, beaches and dunces	Kelp beds	Rock and shell reefs	Seagrass beds	Coral reefs
Coastal wetlands	**H**	**H**	**L**	**L**	**L**	**M**	**M**	**H**

H high, *M* medium, *L* low degree of hazard regulation to ensure that the implications of land-use change and habitat management, such as that resulting from deforestation and wetland degradation, are understood in terms of reducing the potential of landslides and minimizing the consequential losses.

Future Challenges

The restoration of naturally functioning wetland ecosystems can be a cost-effective solution which improves resilience to disasters and secures wider socioeconomic benefits. Different wetlands, whether they are inland or coastal, will regulate different hazards (Table 3). However, the degree to which wetlands regulate hazards will invariably depend on a complex mix of social and environmental factors.

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Tsunamis and Wetland Management

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Robert J. McInnes

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Abstract

A tsunami (literally Japanese for "harbour wave") is a series of ocean waves generated by sudden displacements in the sea floor through landslides or volcanic activity. In the deep ocean, the tsunami wave may only be a few centimetres high but as the tsunami wave approaches the shore it may increase in height to become a fast moving wall of turbulent water in excess of 10 m high. Although a tsunami cannot be prevented, the impact of a tsunami can be mitigated through community preparedness, timely warnings, effective responses and, in some situations, the management of coastal wetlands.

Keywords

Regulating ecosystem service · Coastal wetlands · Disaster management · Tsunami

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Introduction

A tsunami (literally Japanese for "harbour wave") is a series of ocean waves generated by sudden displacements in the sea floor through landslides or volcanic activity. In the deep ocean, the tsunami wave may only be a few centimetres high but as the tsunami wave approaches the shore it may increase in height to become a fast moving wall of turbulent water in excess of 10 m high (Baird 2006) characterized by single or multiple waves or it might dissipate as a low wave (http://www.tsunami.noaa.gov/).

Normal wind waves usually have a wavelength (from crest to crest) of approximately 100 m and a height of roughly 2 m, a tsunami generated in the deep ocean can have a considerably larger wavelength of up to 200 km. Such waves can travel at well over 800 km per hour, but owing to the enormous wavelength the wave oscillation at any given point takes 20 or 30 min to complete a cycle and has an amplitude of only about 1 m. This makes tsunamis difficult to detect over deep water, where ships are unable to feel their passage.

The reason for the Japanese name "harbour wave" is that fishermen could sail out of their harbour, and encounter no unusual waves while out at sea fishing, and come back to land to find their village devastated by a huge wave.

Tsunamis can result in massive destruction of coastal property, land and infrastructure and a devastating loss of life, as seen during the 2004 Indian Ocean tsunami when more than 230,000 people lost their lives along the Asian shore of the Indian Ocean Spalding et al. (2014a), made approximately two million people homeless and resulted in a direct economic loss of US $6 billion in 13 countries (Kathiresan and Rajendran 2005). Although a tsunami cannot be prevented, the impact of a tsunami can be mitigated through community preparedness, timely warnings, effective responses and, in some situations, the management of coastal wetlands.

Coastal Wetlands and the Management of Tsunami Damage

The role of mangroves in reducing the impact of sea-waves and mitigating storm surges has been scientifically proven (Kathiresan and Rajendran 2005) but the level of protection from tsunamis provided by mangroves remains subject to debate (Alongi 2008). Similarly, other coastal wetlands, such as saltmarshes (Möller et al. 2014) and seagrass beds (Newell and Koch 2004), can offer a degree of protection to some extreme storm events and smaller tsunamis.

Mangrove forests provide two key mechanisms for protecting coastal communities and infrastructure from tsunami damage through the direct and indirect attenuation of waves. Directly, the structure of the mangrove forest, comprising trunks, branches, leaves, and understory vegetation, provides additional drag against the wave and dissipates energy. Indirectly, mangroves can promote the stabilization of shorelines and assist in the accumulation and establishment of coastal mineral and organic sediments. Mangroves can capture riverine or coastal sediments that pass through as well as add their own organic matter in the form of roots, leaves, and woody debris. Due to the prevailing anaerobic conditions, the decomposition of organic matter is retarded allowing organic matter to accumulate over time, producing the deep peaty soils. This accumulation can help build the elevation of the soil surface, and in some case keep pace with sea level rise (Marois and Mitsch 2015).

The spatial dimension of the mangroves can also be important in mitigating tsunami damage. Mangrove belts up to several hundred meters wide have been shown to be effective in reducing the wave height of tsunamis by between 5% and 30% with wider mangrove belts being more effective at reducing wave heights and velocities (Spalding et al. 2014b). The density of the mangrove forest vegetation also influences the ability to reduce the tsunami wave height and consequently the area flooded. However, significant tsunamis with a wave height in excess of more than 4 m, for instance waves up to 12 m high were recorded during the Indian Ocean tsunami in 2004 (Baird 2006), have been seen to damage stands of mangroves and even destroying them thus rendering them less effective at protecting from subsequent tsunami flows (Spalding et al. 2014b). Some tsunami waves may be of a greater height than the mangrove tree canopy, so that the incoming wave passes over them. Whilst this might reduce the effectiveness of coastal mangroves in providing protection from tsunamis the same is true for engineered structures, which are rarely built to the height of the mangrove trees.

Natural ecosystems, such as coastal mangroves, will not always provide adequate protection from hazards such as tsunamis and spatial variability will influence the degree of protection provided (Spalding et al. 2014a). However, unlike built or heavily engineered infrastructure, many natural systems have the adaptive ability to regenerate over time and self-repair following the damage wrought by the tsunami (Paling et al. 2008).

Since the 2004 Indian Ocean tsunami discussions have been published in the scientific literature regarding the degree to which mangroves can provide protection from tsunamis (see Baird 2006; Alongi 2008). The overriding view is that for major catastrophic events the degree of protection will be limited. However, despite some acknowledged limitations, the evidence is strong that mangroves and coastal forests can still offer a degree of protection in certain circumstances and localities (Wolanski 2006). Even if the mangroves assist in providing a modest reduction in the total area inundated as a result of a tsunami, this can translate into significant protection of human lives and a reduction in economic damage. Equally importantly, mangroves can trap floating debris and provide a safety net for humans caught up in the floodwaters or the returning waters.

Mangrove Restoration

Over the last few decades, the conversion of mangroves in the pursuit of alternative economic activities, such as shrimp farms, tourist resorts, agricultural or urban land, as well as destruction of off-shore habitats such as coral reefs and seagrass beds, has contributed significantly to the increased risk to human lives and the potential for a reduction in the resilience of coastal communities. While it could be effective to restore and protect mangrove forests and other natural defences in parallel (Dahdouh-Guebas et al. 2005), the need to invest in establishing early warning systems and increased public awareness and understanding of should not be underestimated (Baird 2006). When considered as part of wider coastal zone management approach, the restoration of mangrove ecosystems not only provides opportunities to mitigate to a degree the potential impact of tsunamis but also delivers multiple societal benefits (Costanza and Farley 2007).

Future Challenges

Further information is needed to guide future action on the management of risks associated with tsunamis. In particular, there is a need for assessments of coastal wetlands to understand both the distribution of wetland types and the benefits, including storm and tsunami protection, they provide society. Such an assessment will enable more efficient targeting of resources and inform integrated policy interventions, which may include early warning systems, hard engineered structures, and natural infrastructure such mangrove restoration (Spalding et al. 2014b).

Sea level rise is an ongoing challenge for many countries. Through their ability to trap and accumulate sediments, mangrove soils are actively growing in many places and therefore off-setting sea level rise. Whilst the incremental levels of sea level rise may be inconsequential when compared to a major tsunami with a 4 m wave height, for smaller events ability of the elevation of the soil surface to keep pace with increasing sea levels may be important and represents a more resilient option than the removal of the mangroves.

Potentially, mangroves and other wetland ecosystems have a significant role to play in coastal zone management and protection in certain areas of the world. However, their role with regard to mitigating risks associated with tsunamis is not simple. There are many different types of mangrove forest in a wide variety of geomorphic settings, and in some places mangroves are simply absent from the coastal environment (Dahdouh-Guebas et al. 2005). Regardless of the potential extent of the coastal protection provided by mangroves, integrated coastal zone management should combine comprehensive coastal protection programs which should not rely on just mangroves for protection but also include investment in early warning systems, evacuation protocols, and post-disaster recovery programmes to ensure better protection of human life. This combination of strategies would allow for livelihoods dependent on mangroves to be maintained while also potentially providing improved protection from coastal natural disasters (Marois and Mitsch 2015).

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Soft Engineering for Coastal Protection: Natural Hazard Regulation

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Contents

Abstract

Coastal wetlands, beaches and dunes have always played a role in the protection of coastal lowlands. Using their capability to attenuate waves and prevent erosion has gained more momentum in recent years. Critical to their design is an understanding how different parts of the coastal system function as one coastal protection system, with sediment sources and pathways as their basis. A dune depends on its beach, and salt marshes and mangroves depend on a muddy foreshore. Natural systems perform well under natural conditions, but less in the case of extreme storms. Combinations of soft and hard structures may provide the optimized design needed. It is important to design and value soft defenses as multifunctional assets in coastal development and to realign their design to the needs of local communities, so they are fully integrated and their management and long-term presence can be safeguarded. A major challenge is to convince engineers, that there are 'hard' arguments for using soft defenses so they are regarded as an engineering challenge.

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Introduction

Dunes, sand spits, lagoons, salt marshes, and mangrove forests have always played a role in the flood protection of coastal lowlands. Most of these systems have since been replaced by dikes, often combined with beach armoring or simply been lost by land reclamation. The function and possible use of wetlands and dunes in coastal protection has however gained momentum in later years. Working with Nature (www.pianc.org/workingwithnature), Engineering with Nature (www.engineeringwithnature.org) and Building with Nature (www.ecoshape.nl) are but some of the ongoing initiatives and research programs that focus on nature-based solutions for coastal protection. The devastating tsunami in South-East Asia in 2004 has led to a wide array of studies into the function of especially mangroves in attenuating floods by tsunamis. In the USA, Hurricane Mitch and Catharina triggered research into in the possible role of mangroves in hurricane protection. Also Hurricane Sandy led to similar evaluations. In Europe, there is a reevaluation of the role of salt marshes in flood protection schemes, and in Germany they are even formally protected as such.

All these studies have enlarged our understanding of the role of dunes and coastal wetlands in wave attenuation and flood protection. At present, models are available that can predict the effectiveness of wetlands and dunes in coastal protection with reasonable accuracy (see also ▶ Chap. 176, "Mississippi Watershed and the Role of Wetlands in Flood Management").

Overall the following picture emerges:

- Coastal wetlands are effective in trapping sediment, thus forming shallow foreshores that are crucial in the attenuation of waves. They also have the ability to grow with sea-level rise if sufficient sediment supply is available.
- Coral reefs, sea grass beds, and adjacent mangrove forests function as wave energy attenuating systems. Also sandy shoals, mudflats, and salt marshes form a functional combination, as do sand berms, beaches, and dunes Only the combination will perform well as a protective system.
- The development and maintenance of coastal wetlands is dependent upon local and regional sediment budgets of sand and finer particles and local hydraulic conditions. These also determine their capacity for resilience and for growing with sea-level rise.
- Because all natural wetlands are formed by coastal processes, their height cannot exceed natural sedimentation levels. Shoals, salt marshes, and mangroves do not grow far above springtide levels. The height of coral reefs is restricted to lowest ebb tide levels. The exceptions are dunes that may reach heights over 30 m in favorable conditions.
- Because their height reflects the natural tidal range, coastal wetlands are most effective in the case of smaller and more frequent storms with storm surge levels

that are not significantly above springtide levels. These wetlands are less effective in the case of hurricanes and larger tsunamis.

- Mangroves can provide flood protection also in the case of larger storms, but only in the case of very wide forest belts. Wide mangrove belts can slow down a tsunami but are not effective in lowering the wave height.
- Most wetlands develop in low energy environments and are vulnerable in the case of larger flood events. Hurricanes will uproot and destroy mangrove forests and coral reefs and their natural regrowth can be slow. Larger hurricanes will also erode, breach, and transform spit bars and salt marshes, completely altering coastal morphology and initiating alternative routes for coastal development.
- Coastal wetlands that are effective in the case of larger storms need to have substantial width and therefore require much space. This space may not be readily available especially in densely populated coastal lowlands. The combination with other land uses may be essential for their development and maintenance.
- Trees that are uprooted can do major damage to structures and houses. Especially in the case of smaller mangroves belts, uprooted trees may add to the problem instead of solving it.
- Coastal wetlands can play a vital role in coastal protection schemes, but their integration in protection schemes should be carefully planned. A design that is only based on wave attenuation may fail to meet the ecological and morphological requirements needed to develop a self-sustaining salt marsh, dune or mangrove belt. Emphasis should be on developing and maintaining complete functional combinations.It is also important to combine natural systems with artificial components to generate protective systems that function well also in the case of larger storm events and that do provide desired ecosystem services under average conditions.

General Principles

Especially in developed countries there is a bias towards hard engineering solutions. However, dikes tend to be subject to many possible hidden failure mechanisms that require intense monitoring and testing. Cost-effective dikes are often subject to costly upgrades because of higher safety standards, upgraded and more severe hydraulic conditions, or a better understanding of failure mechanisms. In the long-run, they are often less cost-effective than anticipated.

A soft defense has the advantages that it is simpler in design, it is adaptable at low costs and can be more cost-effective especially where conditions are favorable and building material is available at low costs. There are no hidden failure mechanisms, since height and volume and the presence of vegetation determine its effectiveness. What you see is what you have.

Soft defenses can be more attractive, offer more opportunities for multifunctional use, and provide more ecosystem services than dikes. If a soft defense adds an attractive beach to an urban environment, the beach itself may create more added value to the local economy than the costs of flood protection. Dikes will remain the preferred option in situations that are unfavorable for the use of coastal wetlands and dunes. Often combinations of hard and soft elements generate better alternatives.

A dike is mainly built on the principle to withstand waves with robust revetments and a height that prevents overtopping of waves. Soft defenses are built on the principle of wave attenuation by absorbing wave energy. This is achieved by breaking waves on shallow foreshores, by higher surface friction and energy dissipation due to the presence of vegetation. Waves lose energy and height as soon as the water depth becomes too shallow. This mechanism depends on wave characteristics (e.g., wave height and length), water depth and surface roughness. Waves can break on salt marshes with wave/water depth ratios between 0.45 and 1.60 depending on wave characteristics and surface roughness. In order to be effective in wave attenuation, the width of the shallow foreshore or coastal wetland should be at least several times the wave length.

Dissipation is a complex process that depends on wave and vegetation characteristics. Dissipation increases with the density of roots and thickness of stems and most wave attenuation is achieved in the first 20–30 m of a mangrove forest or salt marsh. However in the case of larger storms and higher waves, wave attenuation may be as small as 0.5 m for every kilometer of mangrove forest. There are several models that can calculate wave dissipation like SWAN (Booij et al. 1996), but differences with field measurements can be large and calibration with storm events are difficult to achieve. It should be noted that dissipation is effective in the case of wind-driven, steeper waves, but less effective in the case of oceanic swells, that as a consequence penetrate much deeper into mangrove forests.

In the case of dunes, sand is reallocated during a storm so an equilibrium profile forms that takes out all the energy of the wave. The steepness of this equilibrium profile depends on wave energy and sand fraction and usually is between 1 in 30 and 1 in 80. Under normal conditions, sand lost in a storm to deeper parts of the profile will be re-allocated to higher parts of the profile by prevailing waves and winds. Resilience is limited where sand is transported away by tidal currents to deeper water out of reach of waves. Beaches and dunes work best as a protective system in the case of a shallow foreshore and a straight coastline.

Soft defenses that consist of dunes are designed on the basis of sufficient volume, needed to create the equilibrium profile that forms during storm conditions, and sufficient height to prevent wave overtopping. Models like Duros and Xbeach are used to model the erosion and deformation of sandy profiles in the case of dunes and are also used to test and design soft defenses that consist of beaches and dunes.

Robustness Under Design Conditions

It should be noted that the aforementioned mechanisms assume that under design storm conditions the vegetation still stands and that the shallow foreshore has not been eroded. However, trees may get uprooted and salt marshes may be heavily eroded during storms.

Not much is known about the robustness of mangrove trees and what wind and wave conditions they can withstand. Studies and measurements indicate that deeply and densely rooted trees of 3–6 m may withstand 3–6 m high tsunamis. However, observations show that larger storm events usually lead to the erosion and loss of mangrove forests. Hurricane Mitch completely destroyed large tracts of mangrove forest up to 200 km from its epicenter. Defoliation, erosion, and burial of vegetation under sand are major processes observed. Also regrowth can be very slow, so the capacity of a wider mangrove belt to withstand major storms will vary in time.

Salt marshes are often built up by a combination of sand and clay. Thinner clay layers on top of sand may lead to major erosion during storms, especially when slope gradients are steep. In the case of a shallow foreshore the waves that hit a salt marsh are already limited in height. Recent experiments have shown that a salt marsh vegetation can withstand waves up to 0,9 meter (Möller et al. 2014) which coincides with storm surge conditions. When the top layer consists of thicker consolidated clay layers, salt marshes can be robust enough to withstand erosion during major storms also in the case of steeper gradients.

Oyster reefs may help stabilize salt marshes but natural oyster reefs formed under prevailing wave climates will also erode during major storms. So most natural systems work well in the case of moderate storm events, but their integrity and performance during major storm events is often limited.

Effectiveness Under Design Conditions

Natural wetlands have a composition and height that directly depends on the processes that form them. Salt marshes can only grow to a level slightly above spring tide. Occasionally storms may add additional sediment, but the upper limit is usually not more than several decimeters above spring tide levels. In estuaries, the combination of river floods and tides may add additional height over the years especially where sedimentation is enhanced by the mixing of sediment laden river water with the sea. An overview of the effectiveness of coastal wetlands can be found in Gedan et al. (2010).

The effectiveness of salt marshes under design conditions depends on storm surge levels, surface roughness and wave characteristics. If storm surges are much higher than springtide levels, the water depth above the salt marsh becomes too high for substantial wave attenuation (see also ▶ Chap. 30, “Hydrologic and Treatment Performance of Constructed Wetlands: The Everglades Stormwater Treatment Areas”). Surface roughness is enhanced by salt marsh vegetation and does contribute significantly do wave height reduction.

Mangrove forests are a wave attenuating system that consists of a shallow muddy foreshore and mudflats in front and different types of mangroves trees that occupy zones that differ in wave energy, sedimentation and salinity. Similar as with salt marshes, the maximum height of the forest floor usually reflects spring tide conditions. The wave attenuation of mangrove forests also depends on a combination of bottom friction and dissipation. Dissipation becomes less with higher storm surge levels. The reason for this is that the density of roots and stems is larger near the surface and become less with height. Mangrove trees have different root systems. Rhizophora has the highest density in terms of roots and is consequently more effective.

Overall the effectiveness of mangroves and salt marshes is highest in the case of more frequent storms with storm surge levels a little above spring tide. Hurricanes are usually accompanied with much higher storm surge levels and may render salt marshes and mangroves less effective. In the case of category 4 hurricanes or higher, a reduction in storm surge levels of 0.5 m per 1 km of mangrove forest has been observed.

Tsunamis are not stopped by salt marshes, nor effectively slowed down. Mangroves appear to have an effect on the speed of movement, but only in the case of very wide mangrove zones. The effect on the tsunami height is very limited.

Dunes can grow up to heights in excess of 30 m and with sufficient height and volume can act as a formidable sea defense in the case of major storms. In the Netherlands, dunes safeguard the coast at safety levels reflecting 1 in 10,000 year storm events. In the case of dunes and beaches, nourishment is often needed to maintain sufficient volume or to stimulate active dune formation.

Upgraded Combinations

As indicate above, natural coastal wetlands have their limitations in the case of major storm events. It is however possible to combine natural wetlands with artificial structures in order to have a more effective and robust combination.

Salt marshes are often combined with a dike, such as along the Dutch and German Wadden Sea. It is also possible to increase the height of the salt marsh near the dike in order to increase wave attenuation, so the required strengthening of the dike is reduced or can be postponed. However, most salt marshes are protected as nature reserve and levels above spring tide can diminish their biodiversity. A salt marsh can be combined with a dune. An example of this is the conceptual design for WaddenWerken (http://www.hosper.nl/waddenwerken). This is not a natural combination, since dunes seldom form where salt marsh is present. Combinations of wide beaches with a sea wall are common as well as the combination of a mangrove with a sea dike. Not all these combinations work well. Especially in the case of mangroves a minimum width of natural forest is needed to have a vital mangrove forest. Sometimes additional artificial structures are needed to create optimal low energy zones for the development of salt marshes, mangroves forest or beaches.

Maintenance, Restoration, New Construction, and Upgrading

Maintaining existing wetlands as part of the coastal defense system is a sensible strategy. Existing wetlands are located in places that are favorable for their development. An upgrade to improve their robustness and effectivity should be considered if their role in coastal protection is important.

Restoring lost wetlands for coastal protection can make sense as well, depending on the mechanisms that have led to their demise and disappearance. Sometimes only limited interventions are needed to restore them, but often favorable conditions have been irreversibly altered. Especially if regional sediment balances have been disturbed by hydropower projects, illegal sand mining, navigation channels, restoration can be difficult.

Salt marshes and mangrove forests are characteristic of low energy zones. It is not possible to establish them in areas that are heavily exposed to wind and waves or lack the necessary sediment input without using also artificial structures and forms of nourishment. Breakwaters may help to obtain the required shelter but a good design is complex and seeks to reduce wind driven waves but enables sediment fluxes towards the coast.

Depending on their position, any wetland utilized as a natural defense will require management. Management will be limited on low energy and accreting coasts, but can be substantial on less favorable locations. Management should not only focus on the wetland itself, but especially on maintaining those conditions that ensure its vitality as a living ecosystem, such as longshore processes, sand and mud budgets. So dunes should be seen in combination with a beach and a mangrove in combination with its wave attenuating foreshore.

Future Challenges

In spite of the increasing interest in the role of coastal wetlands and dunes in soft defenses, different actions are needed to bring these soft defenses more into mainstream decision making. The challenges are:

- Design to give more concrete guidelines for their robust and cost-effective and location specific design, in terms of the capability to meet specific safety standards, with special attention to hybrid designs and the possible role of vegetation. There is still a need to convince “engineers” that it is a realistic alternative and also an engineering challenge.
- Costs and benefits to estimate and calculate overall costs and benefits, in term of added value investments and maintenance costs and in terms added value to other functions but also the strategic value that is offered by its inherent flexibility and adaptive qualities with respect to climate change but also to physical planning. Traditional defenses mainly have costs, but soft defenses also have added benefits that may convince decision makers in the form of tailor-made packages for rural (e.g., focusing on ecosystem services for fisheries, agriculture, and forestry) and urban areas (e.g., with emphasis on water front development, combinations with urban utilities and infrastructure and recreation; see also ▶ Chap. 164, “Local Climate Regulation by Urban Wetlands”).
- Business cases to provide clear integrated examples that focus not just on the technical aspects but give business case approaches. Soft defenses can act as a driving force in coastal (economic) development and as the backbone to

innovative and structurally sound coastal strategies. This may help planners to use soft defenses as a versatile element in coastal planning.

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Wetland Pest and Disease Regulation 173

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Contents

Abstract

Wetlands are a source of health through a range of provisions yet can also be a source of ill-health in the form of exposures to intoxicants, pathogens and parasites, and invertebrate disease vectors. Wetlands have a number of characteristic which put them at specific risk of disease emergence and a broad range of anthropogenic drivers are creating conditions that exacerbate the disease risks in and around wetlands. Many wetland types, including constructed treatment wetlands, can regulate pollutants, pests and disease agents; biodiversity in itself can help to buffer against disease emergence. For specific threats such as vector-borne diseases, there are a range of targeted management actions that can be undertaken. More broadly, taking an ecosystem approach to health in wetlands reduces a wide range of risks and promotes health across the board.

Keywords

Pest regulation · Disease · Pathogens · Parasites · Management · Vectors · Health

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Wetlands as a Source of Health

It is now widely recognized, for example, by the work of the Ramsar Convention on Healthy Wetlands, Healthy People (Horwitz et al. 2012) that wetlands can determine human health and well-being through the provision of ecosystem services including provisioning of clean water, food, medicinal products, places from where livelihoods are derived, as well as psychological relaxation and recreation. Health of domestic animals and wildlife is similarly dependent on well-functioning wetlands, and the health of all these sectors is inextricably linked due to complex interactions and interdependencies of one sector on the other. Moreover, as seen in Fig. 1, the majority of both infectious and toxic diseases are literally shared between these sectors, either from environmental exposure or transmission between hosts. This appreciation of health across the sectors and their complex interrelationships has led to the emergence of the concept of One Health and the understanding of the need for ecosystem approaches to health in wetlands rather than addressing health in organizational and structural silos without appreciating broader impacts of management actions. As an example, the draining of wetlands and application of insecticides as part of human malaria control has led to a range of negative health outcomes for humans, domestic animals, and wildlife derived from ecosystem disruption and loss of productive habitats and livestock support systems (Horwitz et al. 2012).

Wetlands as a Source of Pathogens, Parasites, Disease Vectors, and Intoxicants

Pathogens, parasites, and invertebrate intermediate hosts are a natural part of the biodiversity and functioning of wetlands, often serving to regulate populations. In terms of infectious agents, water is often exploited as a route of transmission to ensure viability (both temporal and spacial) of the pathogen/parasite between hosts and a relatively easy route of entry to the host via drinking, feeding, aerosol inhalation, or penetrating the skin of the host. Wetlands are also a source of natural intoxicants both abiotic (e.g., naturally occurring heavy metals within sediments) and biotic (e.g., the toxin produced by *Clostridium botulinum* which is responsible for avian botulism). In terms of wildlife health, at any time diseases may be endemic and/or cyclical (possibly in relation to seasonal changes) and affecting host births, deaths, fitness, sexual selection, and/or behavior. Despite this, in stable, well-functioning wetland systems, diseases will most likely not cause population change over time.

Anthropogenic Drivers Disrupting Wetland Systems

There are a number of specific attributes which result in wetlands being at particular risk of emerging and re-emerging diseases. These include:

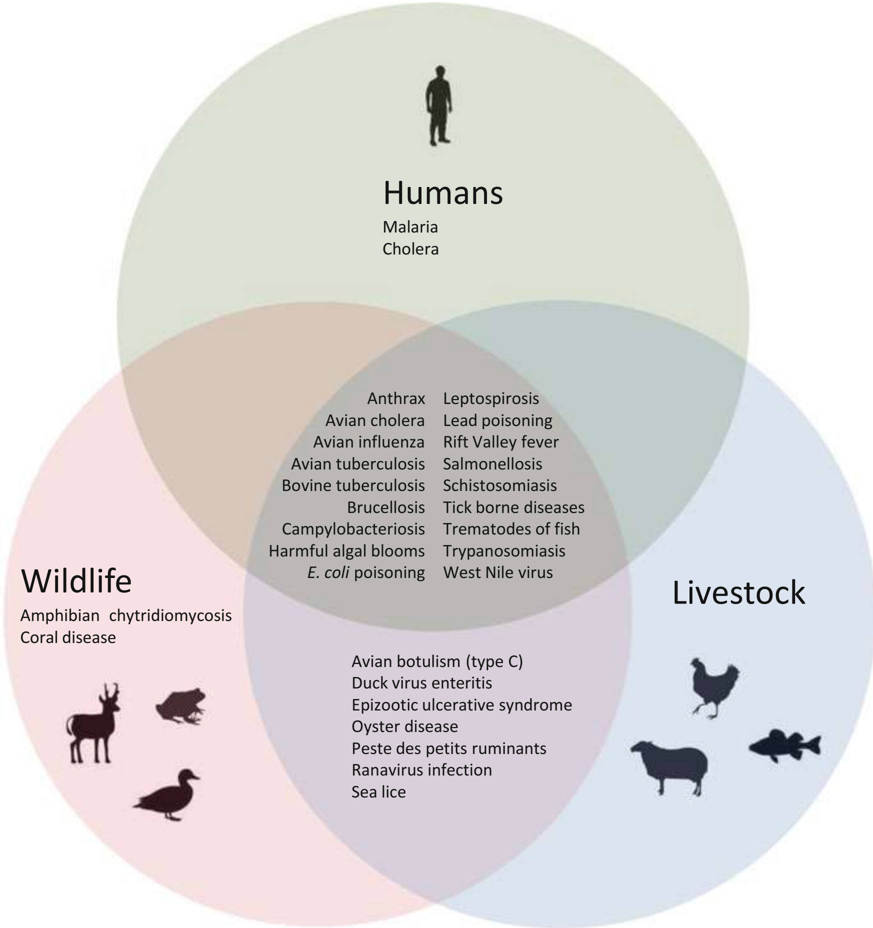

Fig. 1 A number of important wetland diseases (as defined by Ramsar Wetland Disease Manual, Cromie et al. 2012) mapped according to the hosts they affect: the majority of both infectious and noninfectious diseases are common to all three sectors

- Their association with high population densities of people, agriculture including aquaculture, and industry
- Pollution from the above
- Sites providing interfaces between livestock, wildlife, and people thus facilitating disease transmission
- The high diversity of host taxa
- Having been subject to substantial habitat modification, degradation, and poor management
- Sites rarely being isolated, instead usually being connected within catchments
- Routes for trade allowing anthropogenic spread of pathogens/parasite and vectors

Table 1 Selected factors driving disease emergence in wetlands (Cromie et al. 2012, adapted from Morse 2004)

Factor	Examples of specific factor	Examples of diseases in wetlands
Agriculture	• Production systems • Dams • Water management changes • Habitat loss/ degradation • Pollution	• Highly pathogenic avian influenza, e.g., H5N1 • Salmon and trout sea lice • Schistosomiasis • Avian botulism • Harmful algal blooms • Lesser flamingo *Phoeniconaias minor* toxicoses
Globalization	• Food production changes • International trade • Alien species	• Highly pathogenic avian influenza • Amphibian chytridiomycosis • Crayfish plague
Human demographics and/or behavior	• Poor sanitation • Wildlife interface • Encroaching wildlife areas • Civil conflict • Non-sustainable harvesting • Hunting	• Cholera and other intestinal parasites (micro and macro) • Ross River fever • Acanthocephalan outbreaks ineider ducks *Somateria mollissima* • Lead poisoning
Technology and industrial changes	• Food production changes • Breakdown in medical services	• Antibiotic-resistant pathogens • Cholera and typhoid
Climate change	• Changes in rainfall and temperature	• Avian botulism • Bluetongue disease • Yellow fever

- The high proportions of novel and invasive alien species with their associated parasites
- The specific impacts of climate change on wetlands, their hosts, vectors, and pathogens.

In effect, wetlands are "meeting places" where humans, domestic animals, and wildlife are increasingly coming into contact, creating interfaces, which together with other anthropogenic factors are resulting in disease emergence or re-emergence affecting public health, livestock productivity, ecosystem health, biodiversity, and economies at multiple scales. Table 1 provides some of the diseases that result from the specific attributes of wetlands and specific forms of rapid social and ecological change.

For humans: diseases such as malaria (caused by protozoa of the genus *Plasmodium* spread by certain species of mosquitoes), schistosomiasis including bilharzia (caused by trematodes with intermediate stages of their life cycle completed within some species of freshwater snails), and human cholera (caused by the water borne-bacterium *Vibrio cholerae*) are responsible for colossal human mortality and morbidity, constituting a great burden on the economic and social development of developing nations (Horwitz et al. 2012).

For domestic animals: in numerous areas of the world, infectious diseases that were previously endemically stable (vector, host, and environment co-existing with the virtual absence of clinical disease) are now unstable due to these anthropogenic changes, e.g., as seen with the diseases theileriosis and heartwater (Deem et al. 2001).

For wildlife: disease has become of conservation concern due to the impacts of the diseases themselves and also due to management actions occasionally targeted at wildlife where reservoirs of infection may exist in free-living populations. The case of highly pathogenic avian influenza H5N1 is a good example of a pathogen which emerged within human food production systems, spread to wild birds (in multiple situations), caused significant mortality of wild birds, and also resulted in significant human action against wild waterbirds and wetlands in misguided attempts to control a perceived wildlife reservoir (despite a persistent reservoir not actually being identified (Newman et al. 2010)).

Capacity and Ability of Wetlands to Regulate Pests and Diseases

Many wetland types, including specifically designed constructed wetlands, can perform the function of removal of pathogens and parasites (such as viruses, bacteria, and helminth eggs) and pollutants including heavy metals; and remove and store nutrients, thus decontaminating and sanitizing water. This is achieved through a combination of processes. Physical processes may include mechanical filtration by vegetation, adsorption to organic matter, and sedimentation. Chemical processes of oxidation and exposure to biocides excreted by some hydrophytes act to reduce bacterial loads. Biological processes include predation by nematodes and protozoa, attack by lytic bacteria and viruses, and natural die-off in the wetland vegetation (Kadlec and Wallace 2009).

It is also worth appreciating the role that biodiversity plays in regulation of pathogen and parasite diversity, in effect “buffering against” the emergence of disease (Keesing et al. 2010).

Management of Wetlands to Improve Pest and Disease Regulation

Although there are limits to the extent to which wetlands can either filter and remove microorganisms, pollutants, and nutrients and/or control invertebrate vectors, both natural and constructed wetlands can be managed to maximize this regulation of pests and thus minimize disease risks.

Disease prevention and control usually require integrated approaches targeting the host, its environment, and in the case of infectious disease, the agent, and possible vectors. This following section deals with wetland management only. For a broader overview of animal disease prevention and control in wetlands, the reader is directed to the Ramsar Wetland Disease Manual (Cromie et al. 2012).

Management of Pathogens, Parasites, and Pollutants

The efficacy of a wetland to reduce and regulate pathogens, parasites, and pollutants will depend on a number of factors such as:

- The plant and substrate types which need to be appropriate for the target organism or pollutants
- Types of flow within a wetland (surface or sub-surface, the latter generally providing better pollutant removal)
- Hydrological regime including sufficient retention time of contaminated water within a wetland to allow cleansing processes to occur
- Area and depth of wetland
- Pathogen/pollutant/nutrient "load," both natural and constructed wetlands will have a limit to their capacity to decontaminate and if overloaded and/or poorly managed may become themselves a source of contamination (for many pathogens, parasites, and pollutants, prevention of contamination of the wetlands at source is appropriate)
- Climate, higher temperature, and UV radiation will, in general, improve efficiency of decontaminating processes, although even cold temperature wetland treatment systems can be effective
- Management regime of the wetland (Wetlands International 2010)

Management of Invertebrate Disease Vectors

There are a range of arthropods such as ticks and the biting flies (in particular mosquitoes responsible for transmitting malaria, West Nile disease, and Rift Valley fever), and molluscs such as freshwater snails (in particular the various species responsible for transmission of schistosomes) that act as important vectors of infectious disease, transmitting viruses, bacteria, protozoa, and helminths. Consequently, these invertebrates have been the targets of numerous vector control strategies which generally aim to reduce disease transmission by reducing or eliminating the vectors or by reducing contact between them and potential hosts, be they humans or livestock (Malan et al. 2009).

A good understanding of the biology and ecology of the "pest" vector species in question is required to determine the most appropriate management strategy. Any active environmental management measures should be conducted within the context of an ecosystem approach yet may involve altering (temporarily or permanently) hydrology, topography, or vegetation to either reduce the capacity of the wetland to maintain populations of disease vectors and/or to provide suitable habitat for predators of the vectors such as fish.

Case study: Mosquitoes. Oviposition sites requirements of female mosquitoes can be quite specific, and knowledge of these is required prior to taking control measures (SWS 2009). Measures such as increasing water flow, e.g., through irrigation schemes, removing certain vegetation types, or altering water levels at key breeding times can be effective. Mosquito larvae prefer nutrient-rich water and reducing nutrient loading by, e.g., good water treatment and/or fencing livestock out of wetland areas can be beneficial. A reduction in suitable breeding sites can be helped by: a general reduction in isolated stagnant, shallow areas; good management of storm-water retention facilities; and covering of artificial containers which collect water. In artificial and/or managed ponds, installing a pump to keep water moving and removing surface free-floating vegetation can also reduce suitability for breeding mosquitoes.

Native larvivorous fish can be encouraged by connecting shallow mosquito breeding habitat to deeper habitats, e.g., >0.6m with steep sided channels and ensuring suitable fish habitat throughout the year (i.e., persistence of at least some permanent deep water). Predators of adult mosquitoes, such as birds and bats, can be specifically managed for, including erection of artificial nest/roost sites as appropriate (indeed, concern has been raised in North America about the impact on mosquito numbers of the sudden loss of many millions of bats due to white nose syndrome (USGS 2012)).

Case study: Freshwater snails. Reducing snail feeding and breeding habitat can take a number of forms. Populations can be reduced by altering flow rate, e.g., for species within the genera *Biomphalaria* and *Bulinus* flows greater than 0.3 m/s is sufficient, although most snails can withstand flows up to 0.5 m/s. “V” shaped banks in irrigation channels are useful as is removing vegetation/silt in channels to avoid a drop in velocity which may lead to further vegetation growth and good habitat for snails. However, thought should be given to downstream conditions and the potential for the liberated snails to recolonize new habitat. Exposing snail habitat by removing littoral vegetation from the sides of canals feeding irrigation projects can reduce snail numbers, as can heavy rain which can also cause snail displacement. Where possible littoral zones can be dried out to strand snail populations, however, the specific ecology and the resilience of the target species needs to be taken into account.

Future Challenges

Diseases are emerging and re-emerging at an increasing rate (Jones et al. 2008) and dynamics of pests and diseases are shifting within wetland ecosystems. The most important driver of this is unequivocally the dramatic growth and requirements of the human population and the rapid ecological change driven, directly or indirectly, by human activity. Livestock production systems in particular and their impacts on wetlands and subsequent disease emergence is a particular cause for concern. Solutions will come from implementation of One Health approaches which are being advocated at the highest organizational levels such as tripartite collaborations between WHO, FAO, and OIE. The value and lessons from such undertakings and the wider implementation of ecosystem approaches to health in wetlands (as adopted by the contracting parties to the Ramsar Convention at the 2012 Conference of Parties) will hopefully help to promote health in wetlands across the board.

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Flood Management and the Role of Wetlands

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Robert J. McInnes

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Abstract
Historical fluvial flood management practices have largely focused on reducing flooding and reducing the susceptibility to flood damage. Traditional flood management has employed structural and nonstructural interventions as well as physical and institutional interventions. However, increasingly the potential to apply land management options, and in particular the management of wetlands, is being investigated as part of flood risk management practices.

Keywords
Flooding · Natural hazard · Regulating ecosystem service · Disaster risk reduction

Introduction

Historical fluvial flood management practices have largely focused on reducing flooding and reducing the susceptibility to flood damage. Traditional flood management has employed structural and nonstructural interventions as well as physical and

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institutional interventions. Typical interventions include (World Meteorological Organization 2009):

- Source control to reduce runoff (permeable pavements, afforestation, artificial recharge)
- Storage of runoff (wetlands, detention basins, reservoirs)
- Capacity enhancement of rivers (bypass channels, channel deepening, or widening)
- Separation of rivers and populations (land-use control, dikes, flood proofing, zoning, house raising)
- Emergency management during floods (flood warnings, emergency works to raise or strengthen dikes, flood proofing, evacuation)
- Flood recovery (counselling, compensation, or insurance)

However, increasingly the potential to apply land management options, and in particular the management of wetlands, is being investigated as part of flood risk management practices (McAllister et al. 2000). Wetlands are complex and dynamic; therefore, the role that they can play in reducing flood risk and regulating flooding will vary from watershed to watershed and from wetland to wetland. A review conducted by Bullock and Acreman (2003) demonstrated that about 80% of all studies on floodplain wetlands indicated that they reduced or delayed flooding, but the evidence was less conclusive for headwater wetlands where only 45% of studies demonstrated a reduction or retardation of downstream flooding. In fact over 40% of the studies reviewed by Bullock and Acreman (2003) suggested that headwater wetlands increased flood peaks due to rapid responses to rainfall generating higher volumes of flood flows. Therefore, general and simplified statements about their role in particular hydrological functions should be avoided as these fail to acknowledge diversity or complexity of the functioning of wetland systems.

Managing Fluvial Flooding

The role that wetlands can play in regulating fluvial flooding can be simplified into two categories: the storage and attenuation of flooding by river and floodplain systems; and the wider catchment or watershed-scale management of wetlands.

River and Floodplain Management

Despite being “dry” for much of time, floodplains are the riverbed during times of spate. However, many lowland rivers have been straightened, embanked, and disconnected from their floodplains in attempts to convert the fertile flat lands into agricultural or urban environments. In recent years, across the world, attempts have been made to reconnect floodplains to their rivers, to restore more natural riverine hydro-geomorphology, to enhance water quality and in-stream ecology, and to balance the conflicting needs of society whilst reducing flood risk (Wharton and Gilvear 2007). Two examples from Europe demonstrate how sustainable wetland restoration and management can regulate fluvial flooding and also provide wider benefits.

In the north east England, the European Union funded a river restoration project on the River Skerne, near Darlington, to specifically demonstrate and assess the use of more natural processes in flood risk management and at its heart was the desire to involve the local community in the design, construction, and future monitoring.

Over the previous 200 years, the River Skerne had been straightened and deepened. Much of the floodplain had been elevated above the river by industrial waste tipping. Housing developments and infrastructure such as gas pipelines had further compromised the floodplain environment and severed the connection between the river channel and the floodplain (River Restoration Centre 1998). This situation is typical of many river systems in the United Kingdom and beyond.

The objectives of the scheme were to restore a 2 km reach of the river and its floodplain in terms of physical features and to deliver on a range of ecosystem services including flood management, habitat diversity, water quality, landscape, and access for the local community. Secondary objectives included the opportunity to apply innovative river restoration techniques within an urban environment and to further the knowledge and understanding of river restoration through a comprehensive monitoring program. The scheme involved remeandering sections of the river; cutting a new channel; backfilling the old, straightened channel; reprofiling the river banks; and lowering the floodplain surface to increase flood storage. The local community were actively engaged in the design, planning, and implementation of the project to ensure that the resultant benefits of the scheme reached the appropriate stakeholders. The monitoring program, established as one of the original objectives, has discovered that the improved urban environment around the River Skerne has provided significant quality of life benefits to the local community. Key benefits delivered to the local community include an increased wildlife, improved landscape quality, and reduced risk of flooding.

In 2002 parts of Germany experienced disastrous flooding in the River Elbe region with 21 people losing their lives and direct damage to property estimated at approximately €10 billion (Monstadt 2008). The 2002 flood event led to intense public discussion on the inadequacies of the existing flood management strategies, the limitations of technical fixes, and the negative effects of canalization of the river. Immediately after the flooding, and partly driven by immediacy of an impending election, the Federal government developed a five-point program proclaiming a policy change towards a more precautionary approach to flood management. Consequently planned construction measures on the River Elbe were suspended. Based on the program, the Federal government developed and passed a Flood Control Act in May 2005. In addition to this, the government announced the funding approval for a large protected area in Lenzen.

The conservation and restoration of the water retaining capacity of the floodplain had been established as a principle within the action plan for the Elbe in the late 1990s by the International Commission for the Protection of the Elbe. However, the magnitude of the 2002 floods catalyzed a policy shift away from the traditional approach of building dams and dikes to one that gave the river more space and which sought to retain precipitation across the wider river basin. The main physical intervention resulting in relocating 7.4 km of dike up to a maximum 1.3 km away from the river channel, and subsequently recreating 420 ha of functional floodplain in the area of Lenzen. This work was completed in October 2008.

One of the key reasons for the success of this floodplain wetland restoration project was that the preconditions were relatively favorable. The area had an integrated regional administration responsible for nature conservation, flood protection, and agriculture, a proactive and supportive farmer owning and cultivating much of the floodplain area and a sparsely populated area which already included protected areas.

Despite these positive drivers, the project also had to address a range of constraints. Whilst there was some initial lack of acceptance of the concept of restoring wetlands to regulate flooding, the implementation of a dedicated public awareness and promotional campaign, complemented by the personal commitment of the major landowner, minimized the local concerns. Despite the ever-increasing public acceptance of the project, the planning process was prolonged, primarily due to the high level of innovation it embraced. Whilst the restoration of a considerable floodplain area has delivered significant benefits, the ability to deliver a truly catchment-oriented approach to floodplain restoration has been limited. Aside from the project at Lenzen, only a limited number of other floodplain areas have been restored and all have been delivered on a site by site basis. Consequently, ecological synergies and the development of an integrated and resilient network of restored floodplain areas have not been delivered. A stark example of this is lies on the opposite bank of the Elbe at Lenzen. The river forms the border between Brandenburg and Lower Saxony, but despite good cross-border relations the floodplain restoration and flood management measures have yet to be extended to the Lower Saxony side of the river.

Catchment-Scale Management

The wise use of wetlands and the sustainable management of other ecosystems, such as forests, grasslands, and agricultural land, can contribute to reduced flood risk. The degradation of soil structure, the loss of organic material, and the intensification of drainage can all increase flood risk (Wheater et al. 2010). However, land management interventions can be applied throughout a catchment to mitigate downstream flooding and also to provide other benefits such as water quality improvement, carbon sequestration, and habitat enhancements.

The catchment of the Charles River is one of the most densely populated river basins in North America. Urban and suburban development from Boston, Cambridge, and surrounding communities has destroyed much of the lower river's wetlands and natural landscapes. This has resulted in a reduction of natural water storage and significant downstream flooding in 1938, 1955, and 1968 causing millions of dollars' worth of damage. The United States Army Corps of Engineers commenced an analysis of the situation in the mid-1960s and discovered that wetlands still played a major role in storing excess floodwaters and reducing the potential for damage on the upper and middle portions of the Charles River. However, despite an understanding of their value, wetlands in Massachusetts continued to be degraded and lost at a rate of up to 1% per annum. The destruction of wetlands in the upper River Charles basin not only extended flooding problems throughout the catchment, it exacerbated flooding in the lower basin, as floodwaters, liberated from the buffering by wetlands, could move downstream more quickly.

In 1972, the Corps of Engineers commenced work to alleviate flooding in the lower basin by replacing the existing dam at the mouth of the river. A new dam and associated pumping station, which could divert high flows to Boston Harbour, was completed in 1978. The Corps' initial proposal for the basin also recommended the construction of levees and a second dam along the middle portion of the Charles River at an estimated cost of $100 million at 1970s prices. However, the 1968 flood had taught the Corps important lessons regarding the capacity of the wetlands to store flood waters. Based on an understanding of the capacity of wetlands to attenuate flooding, in 1977 the Corps began purchasing land and acquiring easements, prioritizing parcels by location, storage capacity, and threat of development. By 1983, the Corps had purchased approximately 1,300 ha and acquired easements on 1,975 ha of private land. The protected area now includes over 75% of all existing wetlands in the Charles River watershed. In addition to the wildlife, recreational, and economic benefits which have resulted from the protection of wetlands, estimates have suggested that the capitalized flood control value of wetlands within the Charles River basin was approximately $5,000 per wetland hectare at 1981 prices (Thibodeau and Ostro 1981).

Institutional Issues

To successfully deliver on the appropriate wetland management to regulate flooding often requires strong cross sectoral institutional structures. Government departments need to recognize their mutual contribution to flooding issues and their shared responsibilities in ensuring that solutions optimize the benefits across different sectors. Sustainable flood management provides benefits across normal institutional boundaries. For instance, the restoration of a floodplain area may provide health, transport, agricultural, tourism, water resource, and recreational benefits. Differing governmental drivers should be considered to ensure a joined-up approach does not exclude opportunities and therefore compromise the delivery of ecosystem services. Often to achieve this requires strong advocates, both from within governments but also drawn from the wider society, to promote multiple interests across a range of fora.

Governments, NGOs, and other interested parties need to promote the multiple opportunities provided by the role of wetlands in flood management. The need for greater application of communication, education, participation, and public awareness (CEPA) extends both within and without government agencies, the private sector, NGOs, and the general public. Demonstration projects can be hugely influential in explaining the benefits of restoring floodplains to a range of previously sceptic stakeholders.

However, wetlands are only part of the solution. Land-use planning, zoning, and control is generally adopted where intensive or inappropriate development, particular on floodplain wetlands, is undesirable. Providing incentives for development to be undertaken elsewhere can be more effective than simply trying to stop development or conversion of wetlands. However, where land is under development pressure, and especially from informal development, land-use control is less likely to be effective. Flood proofing or house raising are most appropriate where development intensities are low and properties are scattered, or where the warning times are short. In areas prone to frequent flooding, flood proofing of the infrastructure and the communication links can reduce the debilitating impacts of floods on the economy.

Development of flood warning systems and the implementation of timely and robust emergency actions once flooding occurs are complementary to all forms of physical intervention, including wetland-related management options. A combination of clear and accurate warning messages, with a high level of public awareness, can be invaluable and set the foundations for the best level of preparedness for self-reliant action during floods. Public awareness and education programs are critical to ensure that the success of warnings intended to prevent a hazard from turning into a disaster. Evacuation is an essential component of emergency planning. Consideration of evacuation routes is essential and will usually include pathways to a flood refuge at a higher elevation or outward, depending upon the local circumstances. Outward evacuations are generally only necessary where the depths of water are significant, where flood velocities are high and where the buildings are vulnerable. Without careful planning combined with an appropriate level of awareness among the wider population of what is required during a flood emergency, flood evacuation and management plans can fail. It is essential that during the planning stage, active community and stakeholder participation forms an integral component to the process and that regular exercises to assess the viability of the system are undertaken to ensure that genuine evacuations are effective. The provision of basic amenities such as water supply, sanitation, and security in areas where refugees gather is particularly important in establishing a viable evacuation system.

Future Challenges

The management of wetlands undoubtedly has a role to play in the regulation of fluvial flooding. Numerous studies across the world have shown that wetlands can store water, delay, and desynchronize flood peaks and protect sensitive communities and infrastructure downstream. In addition to the regulation of flooding, the same wetlands can often provide a range of other benefits (Blackwell and Maltby 2005). The challenge is to ensure that the multiple benefits provided by wetlands, of which flood regulation is but one, are fully understood and integrated into systemic solutions that will deliver sustainable land and water management (Everard and McInnes 2013).

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Surface Water and the Maintenance of Hydrological Regimes

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Contents

Abstract

Wetlands provide essential ecosystem functions and services which are mostly influenced by the specifics of its hydrological regime. Reflecting the spatio-temporal pattern of inflow, storage and outflow of water of a specific wetland, the wetland regime varies with the specific wetland type and the given, often interacting, environmental conditions. In order to remediate, mitigate and prevent human and climate change impacts, the majority of efforts to maintain or restore the hydrological regime focus on the stabilization of water flow dynamics.

Keywords

Hydrological regime · Hydrological functioning · Wetland types · Human impact · Wetland restoration · Frequency · Timing · Extent

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Introduction

Wetlands provide essential ecosystem functions and services (Brandner et al. 2006) ranging from habitat provision to biogeochemical cycling, climate regulation and, in particular, hydrologic functions. With their impact on flood retention, groundwater recharge, baseflow control, and sediment trapping, wetlands also play an important role in the water cycle at ecosystem, basin, regional and even global scale (Bullock and Acreman 2003). However, most ecosystem functions and services provided by a wetland system are influenced by the specifics of its hydrological regime.

The hydrological regime of a given wetland, sometimes also referred to as "wetland water regime" or "hydropattern," characterizes the hydrological dynamics over time and space and, thereby, reflects the spatio-temporal pattern of inflow, storage and outflow of water of a specific wetland type. In contrast, the often used term "hydroperiod" usually refers to the long-term prevailing hydrological characteristics of a wetland only and vaguely reflects whether a wetland is predominantly waterlogged (saturated soil matrix) or inundated (formation of a surface water body), and its general temporal pattern (permanent, seasonal or periodical).

The hydrological regime of a wetland is determined by the specific wetland type and the given, often interacting, environmental conditions, i.e., the climatic conditions, the geomorphological setting and anthropogenic impacts. While coastal wetlands are primarily driven by tidal hydrology, inland wetlands are driven by seasonal changes in climate, although the relevance of individual climate components may vary among climate regions, with higher impacts of atmospheric precipitation and evaporation in regions of warm climate and air temperature in regions with a cold or temperate climate. The timing, frequency and duration of inundation and waterlogging of inland wetlands are either directly or indirectly controlled by rainfall dynamics, snowmelt and losses through evaporation. Additional factors influencing the wetlands' water cycle are the specific landscape position (e.g., sub-/surface water connectivity), soil formation (e.g., water storage capacity) and vegetation pattern (e.g., water uptake). On the other hand, draining wetlands, groundwater withdrawal or reducing inflows by upstream water extraction may also notably affect the hydrological regime and alter major abiotic and biotic wetland functions. Thus, the hydrological regime is a suitable indicator to identify disturbances, but also to assess restoration efforts.

Types of Hydrological Regimes

According to the variety of drivers controlling the spatial and temporal hydrological pattern, water levels of most wetlands are generally not of constant behavior over time, but rather vary inter-daily or daily, seasonally, periodically, or unpredictably.

For example, tidal-driven wetlands can either be permanently (e.g., lagoon systems in south-east Asia), regularly (e.g., salt marshes along the North Sea) and irregularly (e.g., black mangrove swamps along Florida's coastline) flooded (Tiner 1993). In some cases, inflowing freshwater from rivers or groundwater aquifers may additionally affect the hydrological regime of a tidal-driven system as observed, for instance, in the iSimangaliso wetland system, South Africa (Lawrie et al. 2011) or the Guadalquivir River in the Donana National Park, Spain (Serrano, chapter "Balancing Water Uses at the Donana National Park: Maintaining Hydrological Regimes," this volume 2013).

Permanently waterlogged inland wetland types, for example fens or peatlands, are found in sub-/humid environments in Europe or Canada where water inflow from rainfall, groundwater or melting snow/permafrost is fairly stable. Contrary, prairie potholes of North America, being strongly depending only on rainfall and its variability, tend to vary within and from year to year (Johnson et al. 2010). Although located in different landscape settings, the hydrological regime of both high-altitude wetlands like the Tuolumne meadows in the Sierra Nevada (Lowry et al. 2010) and floodplain wetland systems in Central Europe (Heiler et al. 1995) are, among others, dominated by the seasonal, but spatially and temporally varying snowmelt and, thus, show high variations regarding the timing, frequency and duration of inundation. Their hydrological dynamic, in particular the retention and release of water of the individual wetland also varies by the depth and structure of the sediment layer, terrain features, as well as the vegetation pattern. Since river floodplains are dominated by the dynamic of their upstream river system, the hydrological system is also strongly influenced by river regulation (e.g., reservoirs, channelization) and land use management (e.g., irrigation, afforestation).

Flood pulse-driven wetland systems often receive their water from intense rainfall in higher, usually remote regions, for instance the Okavango Delta, Botswana, which receives a seasonal, fairly predictable flood pulse by runoff generated in the Angolan highlands (Steudel et al. 2013). In contrast, flood pulses can also be highly unpredictable when monsoon-driven rainfall, sometimes even combined with snowmelt, cause catastrophic floods like in lower Brahmaputra floodplains (Brammer 1993).

The hydrological regime of man-made wetlands, such as rice fields in many regions, or ponds that are used for fishery, such as cascading pools in Thuringian Mid-Mountain Range, Germany (Müller Schmied et al. 2008), are strongly controlled by the management of the respective wetland (e.g., channelization, drainage) or upstream regulations.

Further prominent examples for hydrological regimes of various wetland types with varying controlling factors are given in Mitsch and Gosselink (2007).

Assessment of Hydrological Regimes

Typical parameters to describe the hydrological regime of a wetland are (i) the timing, frequency and duration of waterlogging/inundation, (ii) the water level depth (magnitude), (iii) the spatial extent and (iv) the variability of these parameters. The timing describes the time of the year (within annual) or between years (interannual) when the wetland is waterlogged or inundated, whereas the frequency refers to wetting and drying dynamics for a given period. The duration considers the length of time of inundation or waterlogging in hours, days, or even years, depending on the wetland type. The depth of water level (sub-/surface water) is a measure to quantify in- and outflows and, therewith, to relate wetland fluctuations to direct inputs through direct rainfall, rivers and groundwater, but also anthropogenic disturbances in the system and beyond. The spatial pattern and extent of inundation and/or waterlogging refers to the intensity of flooding, but also management impacts in the wetland. The variability of these parameters provides information on long-term dynamics and change impacts, and thereby helps to differentiate the natural succession from human-induced or natural disturbances.

Since the hydrological regime of wetlands differ in various climates, landscapes, but also in wetland types, the water regime parameters used to describe and assess a wetland system may vary between wetland systems. Thus, a parameter that is meaningful for one water regime may be irrelevant for another wetland type. For example, the frequency of flooding is per se negligible in permanently flooded wetlands.

Maintaining Hydrological Regimes

Causes for Changes of Hydrological Regimes

The variety of factors driving the wetland water cycle and the complexity of their interactions result in a high vulnerability of wetland systems to changes and disturbances, either in the wetland or in its environment. The majority of threats affecting the hydrological regimes of various types of wetlands are either related to predicted changes in the climatic conditions or human activities. This is also reflected in the significant loss of wetland area during the past century.

Climate change induced threats to existing wetland water regimes are mainly caused by the following phenomena:

- Increasing sea level is expected to alter the hydrological dynamics of coastal wetlands around the world, but the overall impact remains dependent on the magnitude of sea-level rise, climatic zone and the investments being spent to protect coastlines (Morris et al. 2002);
- Increasing temperatures may affect the water regime such as in northern prairie wetlands (Johnson et al. 2010) or the Great Lakes region and increase evaporation in semi-/arid environments (e.g., Okavango Delta) directly, but also indirectly, by changing snowmelt pattern (e.g., melting rates, snowmelt onset), altering timing of water availability (e.g., droughts) or increasing water demands and associated extraction;
- Changing precipitation patterns will affect the length, frequency and magnitude of flooding in wetlands in all affected regions, whereas decreasing precipitation

will reduce the input to wetland systems and, thus, may change the type of wetland. Some types of wetlands, however, will face increasing precipitation, for example depressional wetlands with small watersheds in arid to mesic climates (e.g., prairie potholes, North America) (Johnson et al. 2010) or monsoon-driven systems in East Asia, although the type of wetland may not change.

Human impacts on coastal and inland wetland systems are generally more multifaceted.

Severe impacts on coastal wetland hydrology result from management activities along the coastline (e.g., tourism industry, forming bays, inlets, airports or harbors) or in the wetlands' tributary basin (Kennish 2001; Phillips and Jones 2006). Given ongoing population growth, urban sprawl and predicted sea level rise, many coastal wetland systems are under serious threat and require conservation and restoration efforts (Kennish 2001).

Alterations of the hydrological regime of inland wetlands are found all over the world. In general, changes in water level fluctuation are caused by either direct disturbances in the wetland or indirect impacts by management in upstream areas. In order to utilize the land, common direct disturbances are drainage, infilling, dam construction, channelization, peat mining, road construction, grazing or a combination of these activities, all affecting the spatial and temporal pattern of inundation and having implication for hydrological, hydrochemical and ecological process-interactions. As shown by many examples (e.g., in Mitsch and Gosselink 2007), wetlands are usually drained for agricultural utilization and commercial forestry. By its extent, the transformation of wetland areas into agricultural fields and plantations remarkably contributed to the overall loss of wetlands at global scale over the last century. Filling of wetlands is, in many cases, aiming on preparation of land for building human infrastructures. An overview about the extent and hydrological implications of draining peatlands is given by Holden et al. (2004). While Hill et al. (1998) demonstrated the impact of dam construction on the hydrological regime and its implication for the diversity of shoreline plant communities, the effects of peat mining on wetland stability are described by Nishimura and Tsuyuzaki (2014).

Indirect impacts on the hydrological regime of inland wetlands may result from stream channelization and regulation, dam construction, mining and industrial activities in the respective watershed, but in particular from land use management and change in upstream areas. In many regions, land clearing and development have caused increased erosion in upland areas which led either to increased sedimentation or higher frequencies and intensities of flooding and, therewith, consequently alteration of chemical and hydrological regime of lowland wetlands in a relatively short time (Hilbich et al. 2007). In other cases, land use changes from grassland to commercial forestry increased water uptake by plantations and evapotranspiration in the uplands, which in turn affects basin and wetland hydrology in lower reaches (Helmschrot 2006).

Maintaining and Restoring Hydrological Functions

With regard to importance of the hydrological regime for wetland ecosystem services and functions, many efforts focus on stabilizing the wetland system by remediating, mitigating and preventing human and climate change impacts. Since there is agreement by researchers, practitioners and conservation experts the effectiveness of wetland restoration depends on the re-generation of water flow dynamics and the frequency of re-flooding, the majority of restoration efforts focus on the hydrological regime of wetlands. There are many examples demonstrating that individual activities can help to successfully respond to such impacts.

A wide range of studies throughout the world have analyzed the success of restoration efforts. For example, Acreman et al. (2007) summarize examples from various sites in Europe demonstrating the efficiency and success of restoration methods in different wetland types and its implication for their hydrological functioning. In other regions, restoration efforts of peatlands resulted in a remarkable increase in water levels for instance in peatlands on the Tibetan Plateau (Zhang et al. 2012) or in the southern Rocky Mountains, USA (Schimelpfenig et al. 2014), whereas restoration activities in interdune freshwater wetlands in Mexico helped to re-establish flooding pattern and reduce disturbances by invasive plants (López-Rosas et al. 2013).

Upstream river regulation has proven to be a suitable instrument to restore and stabilize the connectivity of floodplain wetlands along the Missouri River, USA (Galat et al. 1998) and the Murrumbidgee River, Australia (Frazier and Page 2006). Although some studies show that changes in upland land management can reduce impacts on wetland dynamics like for instance in afforested areas in South Africa (Helmschrot 2006) or in agriculturally utilized systems, restoration efforts barely consider a landscape perspective that integrates upland and wetland conservation efforts and spatially relevant linkages (Faulkner et al. 2011).

Case Study: Impact of Afforestation on Wetland Hydrology, Umzimvubu River Basin, South Africa

The landscape of the semi-arid Eastern Cape Province, South Africa is characterized by the occurrence of different types of palustrine wetlands which vary in extent, topographic position and hydrological functioning, thus hydrological regimes and provide various eco-hydrological functions such as flood flow attenuation, groundwater recharge, baseflow control, and sediment and nutrient trapping. However, little knowledge was available regarding their distribution and functioning, nor the impact of human activities on wetland dynamics such as intensive afforestation in the headwaters since 1989 which was expected to cause severe changes in wetland characteristics.

Helmschrot (2006) developed an integrated research combining empirical field studies, laboratory analysis, GIS and remote sensing techniques, as well as process-oriented plant growth and hydrological modeling to study such impacts. By a system

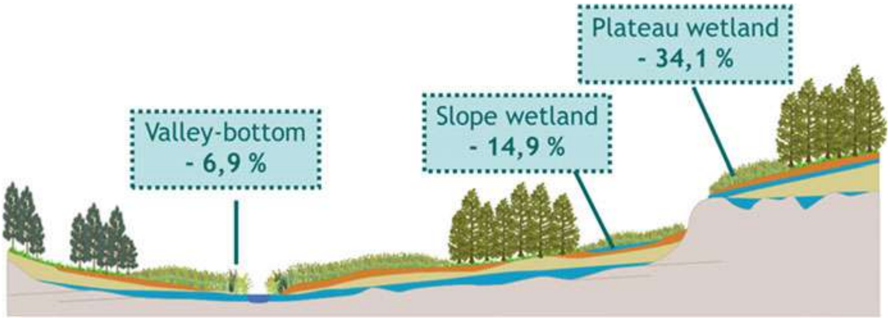

Fig. 1 Average reduction in runoff for different wetland types in the headwater of the Umzimvubu River catchment, Eastern Cape Province, South Africa (Helmschrot 2006)

analysis approach, he identified three main wetlands types (plateau wetland, slope wetland and valley bottom wetland) and a number of subtypes being different in terms of landscape position, soils, vegetation composition and hydrological dynamics. The study showed that two main external drivers control wetland dynamics. Besides the spatio-temporal dynamic of rainfall which affects wetland processes directly, the water availability and release from up-wetland areas are recognized as important factors, but also strongly dependent on the land management. Internal drivers controlling the hydrological regime of wetlands are soil properties and slope gradients. With respect to soil and relief analysis, the clayey-silty soils of valley bottom wetlands show, supported by the flat terrain, a high water retention capacities allowing for flood attenuation during the dry season as well as the control of the low flow dynamics during the dry season. In contrast, soils of plateau and slope wetlands are characterized by relatively good drainage properties. Thus, they are dependent on the subsurface inflow (interflow) and exfiltrating water from the upslope areas.

Based on modeling exercises, Helmschrot (2006) found that afforestation generally affects the hydrological dynamics at catchment and wetland scales. For a small reference catchment, the Weatherley catchment, runoff was significantly reduced by forest plantations by amounts ranging from 13.5% to 21.5%. When examining the flow reduction seasonally, the percentage decline in low flow during dry season (48%) was actually greater than in annual flow (17.5%). At the wetland scale, afforestation influences wetland dynamics as a consequence of total runoff losses in the range of 6.9–34.1% (see Fig. 1), with an average of 17.3%, and losses for individual runoff components varying between 0.3% (groundwater flow in valley bottom wetlands) and 83.6% (surface runoff in plateau wetlands). These losses are addressed to altered recharge/discharge mechanisms as well as reduced base flows and subsurface inflows from contributing areas which are related to increased interception losses, but vary with the wetland type. Plateau wetlands experience a reduction in runoff by 27–48%, but their overall influence on the basin water budget remains small. This is a result of their size, their temporary nature and hydrological process dynamics. Slope wetland runoff losses vary between 11.1% and 19.9% and are mainly caused by a decreased volume of exfiltrating water usually supplied as surface runoff by upper plateau wetlands and the higher water demand of upslope plantations reducing interflow. Because of their seasonal occurrence and due to the lower inflows from upslope areas, slope wetlands will tend to dry out more frequently allowing invasive plants to settle which may support the process of altering into a non-wetland ecosystem. Valley bottom wetlands are less affected (3.9–8.7% runoff loss), since those wetlands are mainly driven by interlinked ground-/surface water dynamics, discharge/recharge processes and direct rainfall input. However, modeling exercises further indicate that changes in forest management and planting practices such as mixing deep-rooting with shallow-rooting species, balancing species compositions, creating buffer zones around the wetlands and reducing ridge planting practices will reduce the runoff loss caused by afforestation and, thus, help to stabilize the hydrological regime.

Future Challenges

Since many regions are expected to face notable changes in precipitation and temperature patterns due to climate change, but also the ongoing sprawl of agricultural areas causing significant wetland losses, maintaining ecosystem functions and services at current levels in a warmer climate and under changing socio-economic conditions will be a major challenge for natural resources manager and decision makers in policy and conservation authorities.

Since assessment studies on wetland restoration rarely follow a landscape perspective, future research should integrate upland and wetland restoration activities and their spatial and temporal interacting processes and feedbacks.

Although some notable efforts have been reached (such as the Ramsar Convention on Wetlands, European Water Framework Directive), a further challenge to policy and decision makers is to develop and implement instruments for the assessment and sustainable management of wetlands in national and international water laws and agreements, and thereby ensuring the functioning of wetlands in changing environments.

A well accepted and integrated measure to identify disturbances and to assess conservation efforts is the concept of environmental flows which integrates the quantity, timing, duration, frequency and quality of water flows required to sustain freshwater, estuarine and near-shore ecosystems including wetland systems with their socioeconomic dimension. Thus, synthesizing the knowledge and experience provided by the high number of individual case studies in order to support and guide the development of environmental flow standards into a scientific framework covering various scales will also be a key challenge of further research.

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176 Mississippi Watershed and the Role of Wetlands in Flood Management

Robert J. McInnes

Contents

Abstract

The Mississippi River is the longest river in North America and the fourth longest in the world. The Mississippi-Ohio-Missouri (MOM) watershed, including the Arkansas River, covers some 4.76 million square kilometers encompassing approximately 40% of the conterminous United States of America. The watershed comprises several major sub-catchments and ultimately discharges into the Gulf of Mexico through the Mississippi Deltaic Plan (MDP). Within the last 100 years, the MDP has experienced relative sea level rise as the surface area has been reduced and wetlands lost. Much of the wetland loss has been associated with wider activities across the watershed that have reduced sediment input to the river and consequently to the wetlands. This, combined with the likely increased intensity of tropical storms as a result of climate change and the ongoing loss of wetlands which offer protection from coastal storm surges, will increase the vulnerability to flooding of near coastal human settlements and infrastructure.

Keywords

Flood control · Sea level rise · Regulating ecosystem service

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Introduction

The Mississippi River is the longest river in North America and the fourth longest in the world. The Mississippi-Ohio-Missouri (MOM) watershed, including the Arkansas River, covers some 4.76 million square kilometers encompassing approximately 40% of the conterminous United States of America (Fig. 1). The watershed comprises several major sub-catchments and ultimately discharges into the Gulf of Mexico through the Mississippi Deltaic Plan (MDP). Driven by a combination of snowmelt and rainfall, the river has a long history of flooding throughout its catchment (Olsen et al. 1999). Similarly, the evolution and development of the MDP is a result of cyclical delta building periods when sediment has been deposited from overbank inundation and river discharges. Within the last 100 years, the MDP has experienced relative sea level rise as the surface area has been reduced and wetlands lost. Much of the wetland loss has been associated with wider activities across the watershed that have reduced sediment input to the river and consequently to the wetlands. This, combined with the likely increased intensity of tropical storms as a result of climate change and the ongoing loss of wetlands which offer protection from coastal storm surges, will increase the vulnerability to flooding of near coastal human settlements and infrastructure, such as the city of New Orleans (Costanza et al. 2006).

Fig. 1 The Mississippi River watershed. (MDP Mississippi Delta Plain). (Reproduced from "Mississippi River Watershed Map" by National Park Service – http://www.nps.gov/miss/riverfacts.htm. Licensed under Public Domain via Commons: https://commons.wikimedia.org/wiki/File:Mississippi_River_Watershed_Map.jpg#/media/File:Mississippi_River_Watershed_Map.jpg)

The Mississippi-Ohio-Missouri (MOM)

Since the mid-1800s, efforts have been made to control flooding across the MOM floodplain (Changnon 1998). The response to flooding by the United States' government for the majority of the twentieth century has been to build levees, dams, and other structures and to straighten the channel throughout the upper Mississippi Basin in order to reduce flood impacts on agricultural land, infrastructure, and human habitation. Despite this, massive floods have occurred frequently over the course of the twentieth and the beginning of the twenty-first centuries. For instance, flooding in 1927 inundated more than 73,500 km^2 of land and economic losses were estimated at US$1 billion (at 1927 prices). The 1993 flood resulted in an area of over 80,000 km^2 being inundated, with some areas remaining under water for more than 200 days, producing an estimated economic impact between US$15–20 billion. The flooding experienced in April and May 2011 represented a 1 in 500 year event and forced the river managers to divert up to 3500 $m^3\ s^{-1}$ of water out of the main channel and into the Atchafalya River Basin in order to protect downstream towns and cities such as Baton Rouge and New Orleans. Without this action, the damage would have been much higher.

Whilst the man-made structures have helped to divert water and reduce impacts to human settlements and infrastructure, there has been increasing interest in alternative approaches to managing flood risk including limiting development in the floodplain, improving flood warning systems, and creating natural buffers, such as wetlands, to reduce potential flood damage (Black 2008). The creation and restoration of natural infrastructure, such as wetlands, is partly in response to the fact that as a result of the construction of the levees, flood damage has been exacerbated by elevating the river stage and increasing flow velocities, resulting in the mean annual flood damage increasing by approximately 140% over the twentieth century (Hey and Philippi 1995); partially as result of the maintenance and repair costs associated with the built flood protection infrastructure (Dierauer et al. 2012); but also as result of the fact that in some states within the MOM watershed between 80% and 90% of the original extent of wetlands have been lost, primarily through artificial drainage (Mitsch and Day 2006).

Hey and Philippi (1995) suggested that the restoration of approximately 5 million hectares of wetlands across the Upper Mississippi River watershed could provide substantial reductions in flood risk, even for 1 in 100 year events, such as the one experienced in 1993. Mitsch and Day (2006) proposed wetlands as an "ecological solution" not just to flooding but also to the reduction of nutrients in the main river, which are a source of chronic water quality issues (hypoxia) in the Gulf of Mexico. As an integrated solution to environmental issues, Mitsch and Day (2006) recommended the creation and restoration of farm wetlands and the diversion of river water into adjacent restored and constructed wetlands along the main river channels during times of flood. Evidence from an agricultural runoff wetland in Logan County, Ohio, USA, demonstrated that a multicelled wetland with a watershed to wetland size ratio of approximately 14:1 was appropriate to buffer storm pulses surface water runoff sufficiently to reduce flood risk downstream (Fink and Mitsch 2004).

Following recurrent floods which occurred in the 1990s, attention was focused on the potential to target wider floodplain restoration, especially in the Missouri River watershed. Galat et al. (1998) suggested restoring a series of floodplain patches, especially along low-lying areas, that had high vulnerability to inundation, and also were at risk from scour and erosion during major flood events. Whilst only 20% of the vast lower Missouri River floodplain may have been amenable to restoration, due to existing infrastructure and agricultural interests, Galat et al. (1998) recommended that wetland managers adopt an approach which targeted specific locations or areas within selected basins to ensure that wetland restoration delivered targeted improvements in flood storage as well as enhancing floodplain biodiversity and facilitating wider recreational opportunities.

Wetland management, and particularly wetland restoration, has a role to play in reducing flood risk in the MOM. The examples demonstrate that where the design process has considered potential synergies there are also opportunities to create integrated, holistic solutions which optimize benefits beyond flood reduction to include inter alia biodiversity enhancements, water quality improvements, and recreational activities.

The Mississippi Deltaic Plan (MDP)

The dynamic Mississippi Deltaic Plain (MDP) has evolved over the last 6000–7000 years at the interface with the Gulf of Mexico. The delta has been dependent on the input and distribution of sediment through a series of channels, distributaries, and crevasse splays. To survive, the MDP has to keep pace with sea level changes. Geological subsidence as a result of the weight of sediment causes a relative sea level change of about 1 cm per annum. Overbank flooding, in-channel dynamics, and reworking from coastal waves and currents have created a dynamic environment characterized by complex wetland systems. However, since 1900, the rate of loss of wetlands in coastal Louisiana has been in the order of 100 km^2 per annum (Britsch and Dunbar 1993). The main cause of this loss is the disconnection of the river from the wider delta environments through the construction of levees, preventing the input of sediment from overbank flooding and crevasse formation. In addition, over 15,000 km of channels have been dredged for navigation, drainage, logging, and oil and gas exploration resulting in a fundamental change in the dynamics of the MDP (Day et al. 2007).

The resilience of the communities living in the MDP to major storms and hurricanes has decreased significantly as a result of wetland loss and the prevailing management regimes in the MDP. Hurricanes are a normal episodic atmospheric force in the MDP. However, in 2005, hurricanes Katrina and Rita, the fourth and fifth most powerful storms to strike the MDP since 1893 with respect to their wind speed at landfall, caused extensive flooding and devastation with over 1800 people losing their lives.

Attention, therefore, has turned significantly towards the potential to restore and manage the wetlands of the MDP in order to increase the resilience of communities in the face of a changing climate and increases in flood risk. Planning for the restoration of wetlands in the MDP requires an integrated effort to ensure the needs of human society are considered in tandem with the sustainable design of coastal ecosystems. This will inevitably require optimizing the combination of traditional engineering approaches, such as flood gates and closure structures, with ecological engineering approaches. Following hurricanes Katrina and Rita, Day et al. (2007) recommended the following four part approach to the future management and restoration of the MDP:

1. Reopening old channels, distributaries, and crevasse-splays to facilitate large-scale reconnection of the river to the wider deltaic plain.
2. Though potentially expensive, pump-dredged sediments over distances of tens of kilometers in order to rapidly create large areas of wetlands in areas which ultimately would be sustained by sediment deposited from re-establishing dynamic distributaries and channels.
3. Using material pumped from offshore, in combination with hard engineering structures such as groynes, rip-rap, and fencing, to create and restore barrier islands in order to reduce wave power and storm surges.
4. Restoring dynamic hydrological processes through a programme of spoil bank removal, backfilling canals, shutting down deeply dredged navigation channels, encouraging sedimentation, and reducing shoreline erosion.

However, the MDP is a complex social and environmental system which is part of the wider Mississippi watershed. The restoration and management of the coastal wetlands needs to be integrated with and cognizant of wider issues across these systems. Therefore efforts to enhance the ability of the wetlands in the MDP to reduce flood risk need to be integrated with improved stewardship of the resources and environment across the MOM to ensure a joined-up and wise-use approach to ecosystem management.

Future Challenges

Across the Mississippi watershed, wetland restoration and management have been considered as being effective in reducing flood risks and increasing resilience of society to climate change. In part, human interventions across the watershed have exacerbated both fluvial and coastal flood risks; and consequently rather than continuing to rely on structural solutions for flood control, attention has turned to the development of a comprehensive flood management strategy that includes using wetlands to intercept and hold precipitation where it falls, store flood waters where they occur and buffer communities from coastal threats. This approach acknowledges that the river is a continuum and activities in the upper reaches of the MOM influence the dynamics of ecosystems in the delta. The future will require the diversity of organization involved in the management of the Mississippi watershed, including the US Army Corps of Engineers, the Federal Emergency Management Agency (FEMA), non-governmental organizations, landowners, scientists, and local communities, to recognize the multiple benefits that can be delivered through an integrated approach to flood risk management that combines wetland management with traditional land management and engineering approaches.

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Water Quality Regulation: Overview

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Jos T. A. Verhoeven

Contents

Abstract

Water quality has deteriorated in many parts of the world with intensive agricultural practices. Nutrients in agricultural runoff are carried towards streams, lakes and coastal areas and cause there serious eutrophication problems, with fish kills and loss of biodiversity as consequences. Wetlands in agricultural catchments have a robust potential to improve water quality by removing nutrients from the agricultural runoff. Creation and restoration of wetlands in the landscape have reduced nitrogen and phosphorus loads downstream by particle trapping, adsorption and nitrification/denitrification processes. Such wetlands should be designed in a way that connects them with surface and subsurface runoff flows. These wetlands have been shown to enhance biodiversity at the landscape scale because they provide habitat for plants, aquatic macrofauna and birds. The wetlands could potentially result in emissions of nitrous oxide and methane, but these do not constitute a major environmental risk at the catchment scale. The benefits in terms of water purification and biodiversity enhancement by far outweigh these risks.

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Nitrogen · Phosphorus · Agricultural runoff · Treatment wetlands

Introduction

Aquatic ecosystems worldwide have become threatened by eutrophication and discharge of heavy metals and organic pollutants. This is particularly true in densely populated regions of the world and in areas with high agricultural activity. Nutrients and pollutants are flushed into water courses and lakes as a result of leaching (diffuse sources) and discharge (point sources). Where wetlands are still part of the landscape, water often flows through riparian buffer zones, instream wetlands, floodplains, or depressional wetlands. It has been widely recognized that wetlands improve water quality at the catchment scale (Mitsch et al. 2001; Verhoeven et al. 2006). This ecosystem service is particularly important in catchments with a dominant agricultural land use, which can be found in many parts of the world (see Fig. 1, MEA 2005). Nitrogen, phosphorus, and pesticides are the most important substances transported from agricultural soils towards streams and lakes, eventually resulting in eutrophication of freshwater bodies and coastal areas (Cho et al. 2010; Lowrance et al. 2007). The functionality of individual wetlands in removing pollutants depends on their size, location in the catchment, and the characteristics of the catchment lithology and soils. The conditions for the most effective removal differ strongly for the two major nutrients: nitrogen and phosphorus. While phosphorus is mostly running off from the agricultural land in particulate form, the major form of nitrogen is dissolved nitrate, which leaches through surface and subsurface runoff

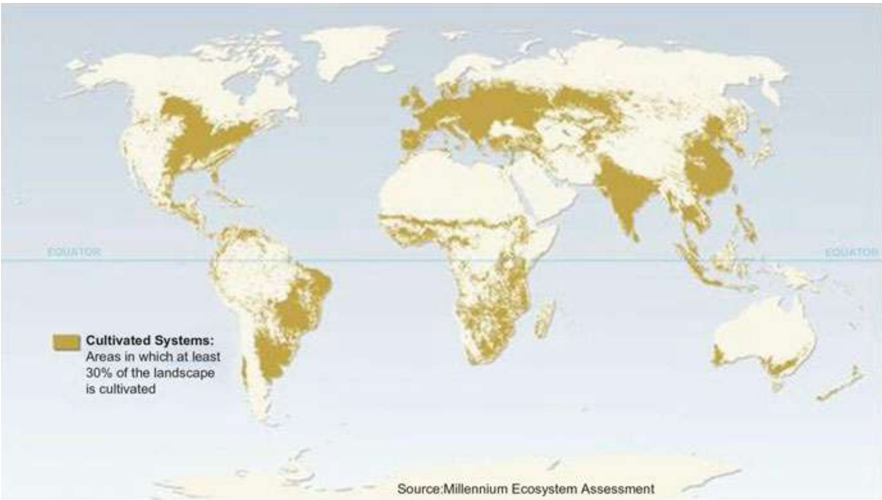

Fig. 1 Cultivated areas of the world. *Brown* regions indicate areas in which at least 30 % of the landscape is cultivated (Reproduced with permission from the Millennium Ecosystem Assessment 2005 (http://www.millenniumecosystemassessment.org), UNEP)

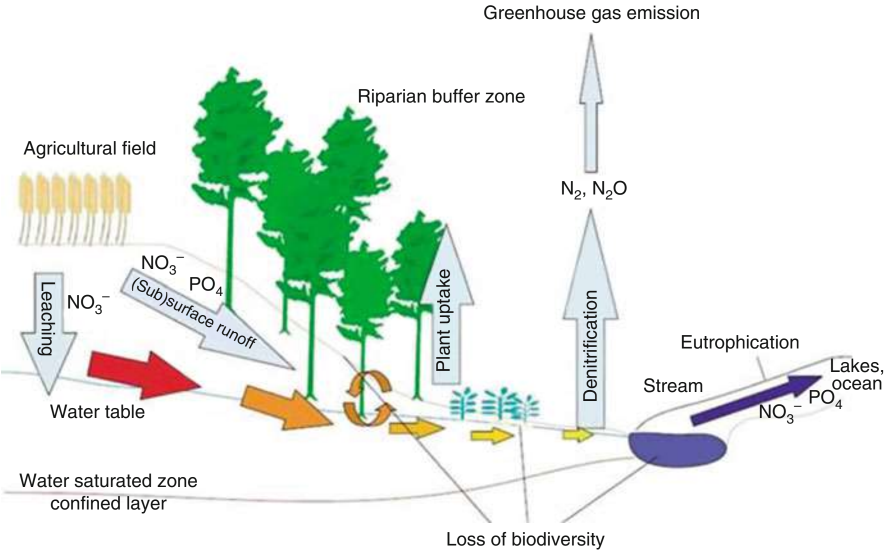

Fig. 2 Cross-section of a riparian wetland showing hydrological fluxes, nutrient processes, and environmental impacts of nutrient loading. *Thicker arrows* with warmer colors indicate a higher nutrient loading rate (Reproduced with permission from Verhoeven et al. (2006))

(Duan et al. 2012; Watson et al. 2010). Therefore, capturing of phosphorus by wetland zones is mostly the result of interception of particulate nutrients from surface runoff, while nitrogen removal is taking place predominantly in deeper, anaerobic zones of the wetland (Duan et al. 2012; Vidon et al. 2010).

Processes

Wetlands are characterized by wet soils or sediments with a predominance of anaerobic conditions. Numerous small-scale redox (reduction-oxidation) gradients exist, mostly associated with radial oxygen loss of roots or biologically mediated water flow through biopores. These conditions result in characteristic biogeochemical functioning, in which redox cycles play an important role. Nutrient removal in wetlands is directly related to these biogeochemical processes and the activity of plants and microbes in the ecosystem. The riparian wetlands and floodplains in many catchments are the zones where all factors required for nitrogen removal coincide, i.e., aerobic microsites for nitrification, anaerobic sites for denitrification and an input of easily degradable carbon and nitrate (Fig. 2) (Hefting et al. 2003; Seitzinger et al. 2006). They are often recognized as hotspots for nitrogen removal and have a measurable effect on nitrogen loading of lakes, coasts, and estuaries (Hefting et al. 2006; Lowrance et al. 2007). The efficacy of wetlands for phosphorus removal is more variable, because phosphates are most mobile in anaerobic soils; however, in catchments with a high proportion of particulate nutrients in runoff they can be quite effective (Knight et al. 2003; Duan et al. 2012). Wetlands have also been shown to remove substantial quantities of pesticides, heavy metals, and polycyclic aromatic hydrocarbons from through-flowing water (Williams et al. 2009; Tromp et al. 2012).

Catchment Scale Effects

Several regional studies have attempted to evaluate the nutrient loading of larger river basins or agricultural areas feeding into coastal areas. Examples are the nitrate loading of the Mississippi basin, which has resulted in dramatic environmental problems in the coastal zone near the river mouth in the Gulf of Mexico (Mitsch et al. 2001; Rabalais et al. 1996). Likewise, there has been major concern regarding the nitrate pollution of the Baltic Sea arising from agriculture in the southern part of Sweden (Arheimer et al. 2004). Restoration of wetlands has been proposed in both regions as a major possible solution to improve the situation. In the Mississippi basin, measures were proposed in the upstream areas, e.g., restoration of riparian zones along lower-order streams and floodplain forests in the lower basin (Mitsch and Day 2006). Hundreds of new wetlands were created in the agricultural landscapes of southern Sweden; these wetlands generally were shown to perform well in terms of nitrogen removal and macrofauna diversity enhancement at the landscape scale (Thiere et al. 2009). In the large-scale restoration of the Everglades wetlands in Florida, major parts of former sugar cane production areas have been transformed into stormwater treatment areas (STAs) to reduce the phosphorus concentration of agricultural runoff feeding into the larger Everglades wetland system (Juston and DeBusk 2006). These large areas of created wetlands and ponds have been shown to lead to a substantial reduction of the phosphate concentrations of runoff water during passage of the cascading system of STAs (Dierberg and DeBusk 2008; Knight et al. 2003). Table 1 gives a number of characteristic ranges of loading rates of nitrogen and phosphorus in different parts of the world.

Loading Limits and Removal Efficiency

Concerns have been raised regarding possible negative by-effects of the loading of wetlands in the landscape with nutrients (Verhoeven et al. 2006). Mesotrophic (semi-) natural wetlands tend to be species-rich and vulnerable to nutrient loading. Another environmental risk is the enhanced emission of greenhouse gases, specifically nitrous oxide and methane. Although nitrous oxide emissions are potentially enhanced in areas with high nitrate runoff, a study in the lower Rhine basin has shown that such emissions primarily occur in low-pH soils in the upper basins of catchments, while wetlands with soil pH >6 have much lower nitrous oxide emissions, even at high levels of denitrification (Hefting et al. 2013). An evaluation of the balance between the nitrogen removal benefit and methane emission risks of constructed wetlands in Swedish landscapes showed that large-scale application of wetlands would result in relatively important reduction of nutrient loading, while the methane emissions were of no significance compared to regional greenhouse gas emissions in general

Table 1 Characteristic nitrogen and phosphorous loading rates of wetlands in agricultural catchments in relation to relevant loading thresholds, expressed per ha wetland per year (After Verhoeven et al. (2006) and Hefting et al. (2013))

Catchment	Location	Wetland type	Origin	N load (kg ha^{-1} year^{-1})	P load (kg ha^{-1} year^{-1})
Liuchahe	PR China	Multipond	Constructed	>500	>50
Regge, Twente	Netherlands	Riparian	Natural	200–1140	
Everglades	USA	Marsh	Natural		2–40
Mississippi	USA	Forested/ marsh	Natural	19–39	2–9
Various	USA	Riparian	Natural	20–155	
Treatment wetlands	USA/ Europe		Constructed	500–9000	100–2000
Maximum load[a]				500	40
Critical load[b]				40	5

[a]Beyond this limit, the wetland will show substantial leaching and associated high concentrations in the outflow
[b]Beyond this limit, wetlands with a species-rich vegetation will show an increase in productivity and change in species composition

(Thiere et al. 2011). At the scale of whole polders (each 200–300 ha), peat meadow areas in The Netherlands were shown to have reduced total greenhouse gas emissions if water tables were raised and fertilization was stopped (Schrier-Uijl et al. 2014). The higher methane emissions were more than offset by net carbon dioxide fixation in peat in the larger wetland areas in the situation with high water tables. It may be concluded that the environmental benefits outweigh the risks. Creation of new wetlands in agricultural landscapes has also been shown to strengthen landscape biodiversity (Thiere et al. 2009).

For wetlands in agricultural landscapes, maximum and critical loading rates have been identified on the basis of literature reviews (Verhoeven et al. 2006; Hefting et al. 2013). Table 1 shows these loading limits. The maximum loads indicate thresholds beyond which the wetland no longer functions properly and starts to leach large amounts of nutrients and to emit N_2O. Wetlands with a low-productive vegetation loaded beyond the critical values may increase in productivity and decrease in species richness.

Future Challenges

There are some major challenges for the maintenance of a healthy water quality in agricultural landscapes in the coming decades. First, food production will have to increase even further to keep up with the needs of the growing world population. This will lead to more reclamation of natural areas for agriculture and will spur intensification of current land use in many parts of the world. The nutrient loading of catchments will increase, so that the role of riparian zones and in-stream wetlands will become more crucial. These zones should be left intact in new reclamation schemes and be restored or broadened in areas which are already in agricultural use. It will be difficult to ensure that these issues become part of overall cost-benefit analyses in such development schemes. A second challenge is the global climate change that is expected to occur. Higher temperatures and higher frequencies of extreme events will impact upon the resilience of water bodies with respect to water quality in a negative way. Eutrophication effects are expected to be aggravated by the changing climate in many parts of the world.

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East Kolkata Wetlands and the Regulation of Water Quality

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Abstract

The East Kolkata Wetlands (EKW), located on the eastern fringes of Kolkata City, India, are a large network of fish farms with their water supply coming from sewage effluents of Kolkata City. The EKW are spread over an area of 12,500 ha and form a part of the extensive delta of the River Ganges. The wetlands sustain the world's largest and oldest integrated resource recovery practice based on a combination of agriculture and aquaculture and have been estimated to provide livelihood support to a large, economically underprivileged population of in excess of 20,000 families. The wetland is a mosaic of landforms including predominantly water-dominated areas (fish farms) to terrestrial usages for agriculture and horticulture. Settlements are interspersed between various land uses. The use of sewage as the basis of aquaculture, agriculture, and horticulture production systems provides the rationale for considering the entire system as a single management unit, also referred as Waste Recycling Region (WRR). Based on the significant ecological and sociocultural importance of the site in 2003, the Government of India declared East Kolkata Wetlands as a Wetland of International Importance (Ramsar Site) under the Ramsar Convention.

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Keywords
Fish farms · Sewage effluent · Kolkata · Integrated resource recovery · Livelihoods · Waste Recycling Region · Ramsar Site

Introduction

The East Kolkata Wetlands (EKW), located on the eastern fringes of Kolkata city, India, represent one of the largest assemblages of fish farms with water supply derived from sewage effluents of Kolkata City. The EKW are spread over an area of 12,500 ha and form a part of the extensive delta of the River Ganges (Fig. 1). The wetlands sustain the world's largest and oldest integrated resource recovery practice based on a combination of agriculture and aquaculture and have been estimated to provide livelihood support to a large, economically underprivileged population of in excess of 20,000 families. Located amid Rivers Hooghly on the west and Kulti on the east between $22^{0}25'$ to $22^{0}40'$ N and $88^{0}20'$ to $28^{0}35'$ E, the wetland is a mosaic of landforms including predominantly water-dominated areas (fish farms) to terrestrial usages for agriculture and horticulture. Settlements are interspersed between various land uses. The use of sewage as the basis of aquaculture, agriculture, and horticulture production systems provides the rationale for considering the entire system as a single management unit, also referred as Waste Recycling Region (WRR). Based on the significant ecological and sociocultural importance of the site in 2003, the Government of India declared East Kolkata Wetlands as a Wetland of International Importance under the Ramsar Convention.

History

The city of Kolkata was established in the sixteenth century and began to struggle with waste water disposal almost immediately. Waste water was originally directed to the Hooghly River, a tributary of the Ganges, but the drainage system proved ineffective, especially during the monsoon season when the increased rainfall elevated water levels in the River exacerbating flooding. Furthermore, by the mid-1850s the Hooghly River was becoming acutely polluted due to a combination of unregulated urban discharges, and outbreaks of malaria, plague, and other diseases were common. Consequently, as the nineteenth century reached its midpoint, the Governor General of India established a committee to oversee improvements in the drainage facility in order to protect the city from flooding and to improve the health of citizens.

In 1857, the drainage committee proposed to abandon discharging to the Hooghly River and to construct a system of sewers, canals, and locks to prevent tidal ingress in order to divert waste water to the River Bidyadhari and hence to the Bay of Bengal. However, within 20 years the river and its associated wetlands, which had acted as a huge storage area for flood and tidal flows, became silted up and lost their connection to the freshwater and tidal environments. The process continued until the

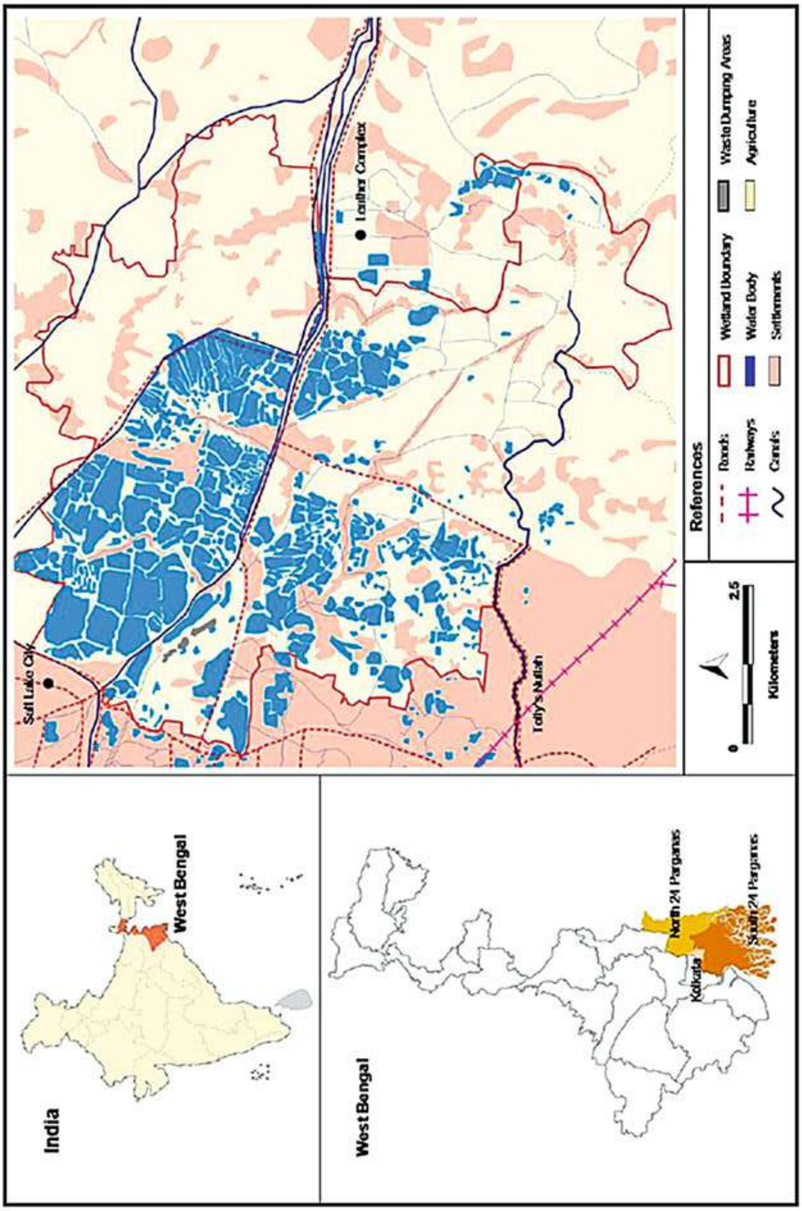

Fig. 1 Location map for the East Kolkata Wetlands (Reproduced from Wetlands International 2008)

end of the nineteenth century, by which time the discharge from the city's waste water system had become the main source of hydrological inputs to the wetlands.

During the late nineteenth and early twentieth century, enterprising local entrepreneurs began to establish waste water aquaculture, horticulture, and agriculture, all dependent on the city's effluent supply. Slowly, the once brackish and seasonally flooded wetlands were transformed into an extensive mosaic of sewage-fed fish farms and paddy fields. Some areas were subsequently reclaimed for urban development as the city grew, especially post-Indian independence in the mid-twentieth century, but the framework of small wetlands fed by waste water and valued for the support they provide for local livelihoods had become well and truly established.

Regulation

Located on the periurban interface of Kolkata City, the East Kolkata Wetlands have been under constant pressures for conversion for settlements and agriculture. In 1956, the Salt Lake Reclamation Scheme to build a satellite township for Kolkata City was approved on nearly 3,800 acres of wetland area. The new townships of Salt Lake City and Patuli emerged on converted wetland areas. Further, construction of Eastern Metropolitan Bypass in 1980 led to conversion of western and northern fringes of wetland area. The current wetland area is therefore a remnant of a large wetland regime extending at least to 20,000 acres in 1940s.

In 1992, a proposal was moved to convert further 800 acres of wetland area for various developmental activities, significant being construction of a World Trade Center. The move received still opposition, leading to filing of a public interest litigation (PIL) by a Kolkata-based NGO, People United for Better Living in Calcutta (PUBLIC). The litigation yielded a landmark judgment by Calcutta High Court which prevented any further reclamation of wetlands by any government or nongovernment agency and also demarcated a boundary. A regulatory framework for wetland in the form of East Kolkata Wetlands (Conservation and Management) Act was passed by the state legislature in 2006, laying the foundation of East Kolkata Wetland Management Authority (EKWMA) as a nodal agency for systematic implementation of wise use principles for management of this Ramsar Site. The Act took explicit cognizance of EKW as a Wetland of International Importance and its various ecosystem services, including ability for regulation of water regimes, wastewater treatment, source of groundwater recharge, and other sociocultural values. The Act defined the land use within the wetland as per revenue records, identifying each land parcel as substantially water dominated, under agriculture, horticulture, or settlements, and banned any further diminution of wetland area, change in ecological character, and overall land use. EKWMA was accorded functions to enable implementation of the Act, which included regulating land use change, preventing unauthorized development, and promoting integrated management of the wetland system.

Water Quality Improvements

Approximately 250 ponds are fed directly with waste water from Kolkata city. The ponds cover a surface area of around 4,000 ha, facilitating a variety of physical, biological, and biogeochemical processes which improve the water quality and act as the basis for a sustainable aquaculture system (Kundu et al. 2008). The Kolkata Municipal Corporation (KMC) generates approximately 600 million l waste water every day and the fish ponds produce in the region of 4.5 t/ha of fish per annum (Little et al. 2002).Underground sewers feed the waste water to several pumping stations towards the eastern boundary of the city from where the effluent is distributed through open channels. Thereafter, the owners of the individual fisheries abstract the waste water and distributed through their ponds.

Organic loading (in the form of biochemical oxygen demand, BOD) on the individual fish ponds varies between 20 and 70 kg/ha/d. The main removal mechanisms for BOD, as well as for nitrogen and phosphorus, are:

Settlement within the sediments of the fishponds
Incorporation into algal biomass
Incorporation into fish biomass
Volatilization into the atmosphere (Kundu et al. 2008)

The removal mechanisms for organic pollutants, and to some degree pathogens, demonstrate their similarities with waste stabilization ponds used elsewhere in warm climates (Mara 1997). In such systems it has been estimated that removal efficiencies increase due to the action of bottom feeding fish which both incorporate biomass into their bodies and also liberate nutrients for uptake by algae and macrophytes (Edwards 1993). The traditional design of the fish ponds of the EKW has produced ponds with a surface area ranging from 0.4 to 162 ha with average depths in the region of 1 m and a range between 0.5 and 1.5 m. Locally these shallow basins are termed *bheris*. Mara et al. (1997) and Bunting (2007) provide further details on the design rationale for BOD removal.

The cumulative efficiency of BOD removal from the waste water influent has been estimated to be on average greater than 80%, with each hectare of fish pond removing approximately 237 kg/BOD/d (Raychaudhuri et al. 2008). The reduction of BOD results from the unique symbiosis between algae and bacteria where energy is drawn from the algal photosynthesis. This further helps in the reduction of pathogenic coliform bacteria where rapid algal photosynthesis, resulting in the demand for CO_2 outstripping its supply from bacterial metabolism, leads to carbonate and bicarbonate disassociation. The resultant CO_2 is fixed by the algae causing pH values to rise to levels lethal to the fecal bacteria as a result of concentration of hydroxyl ions. High light intensities and levels of dissolved oxygen (due to algal photosynthesis) further reduce fecal coliforms (Raychaudhuri et al. 2008). Some estimates have put pathogenic fecal coliform removal as high as 96–99% (Pradhan et al. 2008).

Further water quality improvement is provided by the aquatic plants present in the fish ponds, principally through the fixation of nitrogen, the oxygenation of water, and the accumulation of heavy metals. Within the fishponds of the EKW, water hyacinth *Eichhornia crassipes* is considered to play an important role in bioremediation of pollution (Amann et al. 1995). The plants can leach out heavy metal ions from the water column and the root systems can accumulate further heavy metals. In addition to the water quality improvements provided by water hyacinth, the plants also protect the shorelines and banks from erosion and provide shelter and shade for fish.

Some concerns have been raised regarding the levels of bioaccumulation of heavy metals and other toxins within both the sediments and fish biomass of the EKW fish ponds. For instance, Kumar et al. (2010) demonstrated that Fe, Mn, and Zn concentration in fish tissue were greater than WHO/FAO certified values and recommended that regular monitoring of trace elements should be undertaken from fish tissue collected from the fish ponds. However, the accumulation of a variety of potentially toxic elements (including inter alia P, S, Cl, Cr, Mn, Co, Ni, Cu, and Pb) were determined in vegetables grown in the agricultural areas irrigated by fish pond effluent and levels were considered to be below recommended daily allowances and safe for human consumption (Raychaudhuri et al. 2008).

Future Management Planning Challenges

The EKW are under considerable anthropogenic stress resulting from rapid land use change, pollution, siltation, changes in hydrological regimes, and stakeholder conflicts. However, the area remains a unique environmental and social system which mediates the waste water from Kolkata while providing sustainable food supplies and livelihoods. The management planning approach adopted for the EKW mandates recognition of the values, functions, and attributes of the wetlands and their interlinkages with hydrological, ecological, and social systems where polluted water is transformed into a vital resource for the local communities. Therefore, the goal of present-day management planning has been defined as the *conservation and sustainable resource utilisation the ecological security and economic improvement of stakeholders* (Wetlands International 2008).

The key management strategies adopted at the EKW in order to deliver ecosystem conservation and sustainable resource development and livelihood improvement include:

Management zoning identifying entire wetland area as core zone and direct basin as a buffer zone

Establishing hierarchical and multiscalar inventory of hydrological, ecological, socioeconomic, and institutional features to support management planning and decision-making

Ensuring hydrological connectivity of EKW with freshwater and coastal processes at basin level
Regulating industrial effluent discharge as per relevant standards
Environmental flows as basis for water allocation for conservation and developmental activities
Biodiversity conservation through habitat improvement of endangered and indigenous species
Ecotourism development for enhancing awareness, income generation, and livelihood diversification
Poverty reduction through sustainable resource development and utilization
Formation of multistakeholder groups for planning, implementation, and monitoring of management action plans
Strengthening EKW Management Authority with adequate legal and administrative powers
Capacity building at all levels for technical and managerial skills
Result-oriented monitoring and evaluation at activity, outcome, and impact levels

Following this approach aims to guarantee the delivery of sustainable fisheries development, ecotourism potential, and microenterprise development to improve the quality of life and sustainability of resource management within the EKW.

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Integrated Constructed Wetlands for Water Quality Improvement

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Robert J. McInnes

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Abstract

The integrated constructed wetland (ICW) concept has evolved from traditional considerations of constructed wetlands and is based on the holistic use of land embracing the explicit requirements for wetlands not just to improve water quality but also to demonstrate landscape fit (including physical and social landscapes) and biodiversity enhancement. Pioneered in Ireland, the ICW concept has pursued a paradigm shift through moving away from electro-mechanical treatment of wastewaters in favour of the utilisation of wetlands, and other integrated ecosystems, as natural water management infrastructure. This approach has demonstrated that the achievement of water quality objectives can be attained whilst delivering a range of ecosystem services, distributed more widely across a greater number of beneficiaries and stakeholders.

Keywords

Water quality · Regulating ecosystem service · Multiple Benefits · Waste water treatment

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Background

Whilst the general role of wetlands in the moderation of pollution and improvement of water quality is well established, within the realm of waste water treatment there is also a long history of wetlands being embraced as naturalistic treatment systems. Pioneering work was conducted in Germany in the 1950s on the use of vegetated systems to treat wastewaters. Since then, the work on constructed wetlands has proliferated and constructed wetlands are now firmly established in the tool box of the wastewater manager. It has been estimated that in the United Kingdom there are more than 1,200 constructed wetlands currently in operation. The majority of these systems provide, at best, secondary but more usually tertiary treatment or 'polishing' systems as a supplementary component to a traditional engineered wastewater treatment plant. The historic water treatment discipline-specific focus has often led to the design and management of constructed or rehabilitated wetlands to address discrete problems, overlooking their potential for optimisation to achieve multiple benefits (Everard et al. 2012).

Integrated Constructed Wetlands

A pioneering approach that seeks coherence among the environmental, economic and social dimensions has been developed in Ireland in order to deliver sustainable water management. This approach has built on the decades of research and practical knowledge of constructed wetlands but has assimilated ideas from ecological restoration, social sciences and systems thinking, within an innovative but practical construct. The *integrated constructed wetland* (ICW) concept has evolved from traditional considerations of constructed wetlands and is based on the holistic use of land embracing the explicit requirements for wetlands not just to improve water quality but also to demonstrate landscape fit (including physical and social landscapes) and biodiversity enhancement (Scholz et al. 2007; Harrington and McInnes 2009). The ICW concept has pursued a paradigm shift through moving away from electro-mechanical treatment of wastewaters in favour of the utilisation of wetlands, and other integrated ecosystems, as natural water management infrastructure. This approach has demonstrated that the achievement of water quality objectives can be attained whilst delivering a range of ecosystem services, distributed more widely across a greater number of beneficiaries and stakeholders (Harrington and McInnes 2009).

The ICW concept is a multifunctional, multi-benefit approach which recognises the inextricable linkages between land, people, the atmosphere and, importantly, social needs spanning a wide range of ecosystem services (Harrington et al. 2005; Harrington et al. 2011). The efficacy of ICWs in delivering multiple ecosystems services has been extensively tested and documented in numerous peer-reviewed studies (including Harrington and McInnes 2009). So too has been their efficacy at performing specific roles, such as treatment of contaminated water in agricultural landscapes with not only low environmental impact but crucially, associated ecological gains (Carroll et al. 2005), cost-effectiveness in treatment of this waste water (Culleton et al. 2005), treatment of domestic sewage (Doody et al. 2009) and livestock wastewater (Dunne et al. 2005; Harrington and McInnes 2009; Harrington and Scholz 2010; Scholz et al. 2007) and enhancing wetland biodiversity (Becerra-Jurado et al. 2010).

The ICW approach has been applied across a range of water treatment applications. The Government of Ireland has published a guidance document on their design and management for treating both domestic and farmyard wastewater (DEHLG 2010). A similar guidance document has been produced on behalf of the Scottish Environment Protection Agency (SEPA) for the treatment of farmyard wastewater (Carty et al. 2008). Both of these documents provide the design assurances that need to be considered in order to manage risks.

DEHLG (2010) advocates a risk-based assessment and provides a framework for evaluating and managing pressures and impacts on identified 'receptors' based on a *'Hazard-Pathway-Receptor'* model. Risk is defined as the likelihood (or expected frequency) of a specific adverse consequence. Applied to an ICW system, it expresses the likelihood of impacts arising from the construction or operation of an ICW and its water-vectored pollutants, i.e. the potential *hazard*. A *hazard* presents a risk when it is likely to significantly and negatively affect the status of a valued resource such as surface or groundwaters or natural and built heritage i.e., the *receptor*. An impact can only occur if a significant linkage or *pathway* is established between the *hazard* and the *receptor*. For the hazards associated with water-vectored pollutants from waste waters the key factors to minimise or mitigate the risk are velocity (addressed by having an appropriate wetland configuration), associated residence time (addressed through provision of adequate functional wetland area) and sub-soil physical properties necessary to deliver the required impermeability and isolation from receptors. The key environmental receptors for ICWs are:

- Surface Water
- Groundwater
- Soil/subsoil
- Landscape
- Flora and Fauna
- Air
- Human
- Archaeology

The principal contaminants that constitute the hazard are dependent on the origin and provenance of the water-vectored pollutants. Information on the influent water quality is essential for both design purposes and also for understanding the prevailing hazard. Volumetric information is also vital as the influent water volume may be increased by rainfall. The nature of the influent will affect not just the risk management strategies (as applied during the design stage) but also the subsequent costs and benefits.

Published data has demonstrated the efficiency of ICWs to moderate a range of water-vectored pollutants. For instance removal rates of phosphate from farmyard run-off have been between 81% and 99% (Harrington and McInnes 2009) and 99.9%, 96.7% and 97.9% for *Escherichia coli* bacteria, chemical oxygen demand (COD) and biological oxygen demand (BOD) respectively (Scholz et al. 2007).

Cost effectiveness should be considered as a product of an appropriate cost-benefit analysis (CBA). For this to be effective it is essential that all the benefits (or ecosystem services) associated with implementing a natural wetland solution are recognised. Failure to account for all the benefits can introduce externalities resulting in a skewed CBA. The failure to recognise fully of the range of ecosystem services and their beneficiaries remains a concern in wetland restoration work.

A simple assessment has been undertaken of the costs associated with an ICW constructed in County Monaghan, Ireland (Doody et al. 2009). The ICW was constructed in 2006/2007 to treat combined sewage from the village of Glaslough in North Monaghan. The design capacity was for a population equivalent (pe) of 1,750. The plant became operational in October 2007 and the initial operational load was approximately 800 pe.

Original estimates of the design capacity to treat the village of Glaslough, using a traditional waste water treatment plant, equated to an estimated cost of €1,530,000 (2008 prices) for a plant with a 650 pe capacity. The actual total capital cost of designing and constructing the Glaslough ICW (including land lease and monitoring equipment) was €770,000 including value added tax (2008 prices). The final design was for a system to treat 1,750 pe. Consequently, this simple comparison of costs indicates that when compared with a traditional treatment facility, the ICW provides approximately three times the pe capacity at half the price. Similarly, the operational and maintenance costs of the ICW system were estimated to be less than 5% of those associated with a traditional treatment plant (Doody et al. 2009).

However, this simple equation fails to address the benefit side of the equation. The ICW is not only consistently out performing traditional waste water plants in terms of the final quality of the discharged effluent, but as with other ICW systems, it is also providing a range of ancillary benefits including *inter alia* recreation (horse riding trails and walking), carbon sequestration and carbon costs foregone through the reduced energy costs, biodiversity enhancements and improved landscape aesthetics (Doody et al. 2009; Harrington and McInnes 2009).

Whilst the simple cost formula applied to the Glaslough system cannot be transferred universally, it provides an indication of the cost-effectiveness of using a wetland solution. The costs will vary from system to system depending on a range of variables including the property of the local soils, the need to import materials, distances required for pumping, pipework and other service infrastructure and the cost of land. All these issues should be included in both the capital and maintenance cost assessments. But, as stated before, the same rigour applied to the cost estimates need to be applied to capturing the value of the ecosystem services provided by wetland water management solutions such as an ICW.

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180 Wetlands in the Management of Diffuse Agricultural Run-Off

Mark Everard

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Abstract

Natural wetland functions modify the flows and physico-chemical composition of water, as extensive systems as well as localized pockets distributed across landscapes. Contemporary, heavily modified urban and rural landscapes generate a range of diffuse sources of pollution. Intensification of land use not only generates substantial and diverse pollutants but also displaces wetland functioning naturally mitigating at least some of the problems caused by increased run-off. Retention, restoration of construction of novel wetlands and wetland functioning in agricultural landscapes is of particular importance for the reduction of run-off from farmed land.

Keywords

Anthropocene · Pollution · Agriculture · Land use · Surface sealing · Constructed wetlands · Integrated constructed wetlands · Ecosystem services

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Introduction

Many natural wetland functions occur in localized pockets across landscapes, modifying the flows and physico-chemical composition of water passing through them. However, in the Anthropocene, the current era in which the cumulative impacts of human activities on the ecosystems and cycles of Planet Earth are at a scale significant enough to constitute a new geological epoch (Crutzen and Stoermer 2000), we live in heavily modified landscapes.

Point sources of pollution, from factories, sewage treatment works and other discrete sources, have been subject to increasing control over the past century and more, at least across the developed world. However, diffuse inputs of both storm water and pollution from a range of sources – urban and industrial sites, infrastructure and rural land use – is of increasing prominence (Novotny 2003). Furthermore, intensification of land use displaces wetland functioning that would naturally mitigate at least some of the problems caused by increased run-off.

Retention or restoration of wetlands and wetland functioning in agricultural landscapes is of particular importance for the reduction of run-off from farmed land (Verhoeven et al. 2006).

Characterising Diffuse Agricultural Run-Off

The *Ecosystems and Human Well-being: Wetlands and Water Synthesis* of the Millennium Ecosystem Assessment (2005) outlines the extent to which agriculture has adversely affected the freshwater environment, and been the principal driver of wetland loss globally. Impacts from agricultural run-off are multiple and substantial.

The panning (surface sealing due largely to compaction especially from the footfall of livestock and use of heavy vehicles) of farmed land reduces groundwater infiltration and contributes to elevated run-off of floodwater from farmed landscapes during heavy precipitation. Concentration of flood run-off can increase its erosive power, contributing to soil erosion and consequent water contamination. Attenuating floodwater close to source is then a priority to avert these adverse consequences.

Agricultural run-off bears with it a range of pollutants. These include microbial pollutants of health concern (such as *Cryptosporidium*, faecal coliform bacteria and viruses) as well as 'sanitary determinands' (particularly organic matter with associated biochemical oxygen demand and ammonia), nutrient substances (particularly forms of phosphorus and nitrogen) and suspended solids which also carry with them a range of adsorbed and absorbed pollutants. This diversity of pollutant types can cause substantial changes in aquatic ecosystems, ranging from physical effects such as increasing turbidity and the blinding of river and lake sediments, though to eutrophication and other chemically vectored ecosystem changes as well as microbially vectored problems.

The attenuation of agricultural run-off is then a priority, although of course avoidance or minimization of the generation of run-off and its associated pollutants at source is a more strategic priority.

The Role of Wetlands in the Management of Agricultural Run-Off

Wetland systems, both natural and constructed, can play significant roles in managing run-off from agriculture (Ockenden et al. 2012). The diverse hydrological, physico-chemical and biological functions of wetlands can serve to reduce muddy flooding and other floodwater-related run-off problems, as well as capturing suspended matter, transforming organic and nutrient substances, and reducing loads of microbial contaminants.

Retention of natural wetland systems and processes, be that extensive mire, bog or other natural wetland systems or else more localized gulleys in the landscape which may provide important if often overlooked wetland functions, can radically reduce both flood peaks and pollutant loads exiting farmed land.

Constructed wetlands can also play significant roles in a range of run-off management challenges. Many constructed wetlands are designed to serve narrow purposes, such as retaining contaminated yard run-off or detaining rain-generated peak flows. However, Integrated Constructed Wetlands (ICWs, addressed in detail in another chapter in this Wetland Book) represent a philosophy of wetland design to achieve multiple, simultaneous benefits (Harrington and McInnes 2009). The range of interconnected beneficial outcomes for which ICWs may be designed includes hydrological buffering, carbon storage, nutrient retention for further productive use or else chemical transformation, breakdown of microbial and other contaminants, trapping of suspended sediment, and a range of wider ecosystem services including enhancing landscape aesthetics, providing amenity and supporting wildlife.

Other more localized and strategically placed solutions can restore important wetland functions to farmed landscapes (Heathwaite et al. 2005). Leaving unploughed contour strips (unmanaged bands of land following contour lines) in farmed landscapes, allowing natural vegetation to establish or else deliberately planting them with trees or hedgerows, can have a substantial impact on the reduction of soil loss and associated pollutants, enhancement of natural infiltration into groundwater, as well as supporting desirable wildlife including pollinators and the predators of crop pest organisms. Another particularly useful form of local intervention is the buffer zone, constituting a fringe of land adjacent to a watercourse that is fenced off, planted up or is no longer ploughed. Habitat regeneration in buffer zones can not only attenuate a substantial amount of diffuse pollution from agricultural land, but can promote the diversification of habitat in the watercourse edge with direct benefits including offsetting some of the negative impacts of run-off entering from upstream reaches that have not been protected by buffer zones. A key consideration for buffer zone design is to ensure that water-vectored pollutants can be retained within the buffer zone, and that they do not bypass the buffer zone feature via sub-surface drainage, ditch networks or poorly-sited gates funneling field run-off (Correll 2005).

Wetlands of all scales, ranging from the wetland functions of often overlooked ditches and runnels in the landscape through to extensive or strategically placed wetland systems, have multiple beneficial roles to play in agricultural landscapes, including the attenuation of run-off (Casey and Klaine 2001).

Other Solutions to the Management of Agricultural Run-Off

As noted above, a more strategic approach to avoidance of aquatic pollution is averting the generation of both run-off and associated pollutants at source. This can be achieved by a range of measures, mainly beyond this scope of this brief introductory chapter. Some simple yet effective measures include:

- separation of clean roof water from contaminated livestock yard water, reducing the volume of potential run-off. This measure also preserves the clean water for other beneficial uses, as well as reducing volumes of contaminated yard run-off that may then be more readily intercepted and treated or beneficially reused, substantially minimizing the pollutant load in run-off.
- avoidance of tillage of land and heavy grazing on steep slopes, averting erosion and protecting soil and nutrients in farming landscapes.
- contour ploughing such that furrows follow contour lines, rather than being oriented down-slope which may promote run-off and associated erosion and pollution.
- relocating gates and feeding rings from valley bottoms to the tops of hills or on drier land away from drainage lines, such that mobilized sediment and other pollutants do not flow immediately into watercourses but may instead be attenuated by natural landscape wetland functions.

These measures, to one degree or another, make use of or avoid overburdening wetland processes within farming landscapes. The net result includes better management of flood peaks and aquatic contamination and the conservation of useful resources (particularly soil and nutrients) within the farming system.

They also reflect that, viewed at a closer spatial scale, diffuse pollution may be more logically defined as comprising multiple local point sources of pollution. Wetland solutions, such as buffer zones, attenuation basins and a range of SuDS (sustainable drainage systems) and related rSuDS (rural sustainable drainage systems), may then be applied strategically to address key run-off pathways from farmed land.

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Constructed Wetlands for Water Quality Regulation

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Jan Vymazal

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Abstract

Constructed wetlands (CWs) have been used for wastewater treatment since the 1960s. Constructed wetlands are engineered systems that have been designed and constructed to utilize natural processes involving wetland soils, vegetation, and microbes to treat wastewater. Constructed wetlands may be categorized according to the various design parameters, but three most important criteria are hydrology (surface flow and subsurface flow), type of macrophytic growth (emergent, submerged, free-floating), and flow path (horizontal and vertical). Different types of constructed wetlands may be combined with each other, i.e., hybrid or combined systems, to utilize the specific advantages of the various systems. CWs are used to treat municipal sewage, as well as agricultural and mine drainage, industrial effluents, landfill leachate or stormwater runoff. Constructed wetlands are considered as reliable and robust treatment systems with low maintenance and operation costs.

Keywords

Wastewater · Macrophytes · Organics · Nutrients

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Introduction

Constructed wetlands (CWs) have been used for wastewater treatment since the 1960s (Vymazal 2011a). Constructed wetlands are engineered systems that have been designed and constructed to utilize natural processes involving wetland soils, vegetation, and microbes to assist in treating wastewater. They are designed to take advantage of many of the same processes that occur in natural wetlands, but do so within a more controlled environment. Constructed wetlands may be categorized according to the various design parameters, but three most important criteria are hydrology (surface flow and subsurface flow), type of macrophytic growth (emergent, submerged, free-floating), and flow path (horizontal and vertical). Different types of constructed wetlands may be combined with each other, i.e., hybrid or combined systems, to utilize the specific advantages of the various systems (Vymazal 2005).

Free Water Surface Constructed Wetlands

A typical free water surface constructed wetland (FWS CW) is a shallow basin, containing 20–30 cm of rooting soil, with a water depth of 20–40 cm (Fig. 1). There is no special requirement for the soil quality; the most important role of soil is to support macrophyte growth if emergent or submerged plants are present. If free floating plants such as water hyacinth (*Eichhornia crassipes*) or duckweed (*Lemna* spp.) are used, soil matrix is usually absent. Dense emergent vegetation covers substantial fraction of the surface, usually more than 50%. The most frequently used emergent macrophytes are common reed (*Phragmites australis*), cattail (*Typha* spp.), and bulrush (*Scirpus* spp.). The shallow water depth, low flow velocity, and presence of plant stalks, leaves, and litter regulate water flow, and especially in long, narrow channels, ensure plug-flow conditions (Reed et al. 1995). One of their primary design purposes is to contact wastewater with reactive biological surfaces (Kadlec and Knight 1996). The most important treatment processes occur in the water column and in the litter layer on the bottom. Also, periphyton growing on submerged parts of macrophytes adds to removal of pollutants.

Suspended solids are rapidly removed in FWS systems under quiescent conditions by deposition and filtration. Attached and suspended microbial growth is responsible for the removal of soluble organic compounds which are degraded aerobically in the water column as well as anaerobically in the litter layer. Oxygen is supplied to the wetland water column by diffusion through the air-water interface and via the photosynthetic activity of plants in the water column, namely algae. Nitrogen is most effectively removed in FWS constructed wetlands by nitrification/denitrification. Ammonia is oxidized by nitrifying bacteria in aerobic zones, and nitrate is converted to free nitrogen or nitrous oxide in the anoxic zones. FWS CWs provide sustainable removal of phosphorus, but at relatively slow rates. Phosphorus removal in FWS systems occurs from adsorption, absorption, complexation, and precipitation. However, precipitation with Al, Fe, and Ca ions is limited by little

Fig. 1 FWS constructed wetland at Otorohanga, New Zealand, planted with *Eleocharis sphacelata* for tertiary treatment of municipal sewage (Photo credit: Jan Vymazal © copyright remains with the author)

contact between water column and the soil (Vymazal and Kröpfelová 2008; Kadlec and Wallace 2009).

FWS CWs are used for all types of polluted water, but the most common use is for tertiary treatment of municipal sewage, stormwater runoff (e.g., urban, highway, airport, agriculture, golf course, nursery), mine drainage (Fig. 2), and agriculture drainage waters, landfill leachate, and industrial effluents such as pulp and paper, refineries, or abattoir (Vymazal 2011a; Vymazal and Kröpfelová 2008; Kadlec and Wallace 2009).

Constructed Wetland with Horizontal Subsurface Flow

In horizontal CWs (HF CWs), the wastewater flows slowly through the porous medium under the surface of the sealed bed in a more or less horizontal path until it reaches the outlet zone where it is collected before leaving via level control arrangement at the outlet. During this passage, the wastewater will come into contact with a network of aerobic, anoxic, and anaerobic zones (Cooper et al. 1996). The aerobic zones occur around roots and rhizomes that leak oxygen into the substrate. At present, coarse porous media (size fraction cca 5–20 mm) with high hydraulic permeability such as washed gravel or crushed rock (Fig. 3) are used in order to ensure subsurface flow. The major roles of plants in HF CWs include insulation of the bed surface during cold periods, provision of substrate for growth of attached bacteria on roots and rhizomes, oxygen release from roots to the adjacent

Fig. 2 FWS constructed wetlands for treatment of coal mine drainage at Monastery Run, Pennsylvania, USA, planted with *Typha latifolia* (Photo credit: Jan Vymazal © copyright remains with the author)

rhizosphere, release of antimicrobial compounds into the rhizosphere, and nutrient uptake. The most frequently used plant worldwide is *P. australis*, but many local plants have been used (Brix 1997; Vymazal 2011b).

Organic compounds are very effectively degraded aerobically as well as anaerobically by bacteria attached to the plant's underground organs (i.e., roots and rhizomes) and media surface. However, due to a constant saturation of the filtration beds, anoxic/anaerobic processes prevail. Suspended solids are removed from wastewater by filtration and sedimentation. The major removal mechanism for nitrogen in HF CWs is nitrification followed by denitrification. (Vymazal 2007). However, the field measurements have shown that the oxygenation of the rhizosphere of HF constructed wetlands is insufficient and, therefore, the incomplete nitrification is the major cause of limited nitrogen removal. Phosphorus is removed primarily by precipitation with Ca, Al, and Fe. However, media used for HF wetlands (e.g., pea gravel, crushed stones) usually do not contain great quantities of Fe, Al, or Ca, and therefore, removal of phosphorus is generally low.

HF CWs are used for all types of wastewater including municipal sewage, industrial and agro-industrial effluents, landfill leachate, and stormwater runoff. However, most installations have been used to treat municipal sewage (Fig. 4). The investment costs of HF CWs are higher than those for FWS CWs. The difference is made up by the cost for the liner and for the selected filtration material and its transportation.

Fig. 3 Filtration bed filled with crushed rock with distribution zone filled with stones at HF CW Čejkovice, Czech Republic (Photo credit: Jan Vymazal © copyright remains with the author)

Constructed Wetlands with Vertical Subsurface Flow

Vertical flow (VF) constructed wetlands comprise a flat bed of graded gravel topped with sand planted with macrophytes. The size fraction of gravel is larger in the bottom layer (e.g., 30–60 mm) and smaller in the top layer (e.g., 6 mm). The depth of the filtration bed is usually 1–1.2 m. VF CWs are fed intermittently with a large batches of wastewater through the distribution system on top of the bed (Fig. 5). Wastewater then gradually percolates down through the bed and is collected by a drainage network at the base. The bed drains completely free and it allows air to refill the bed. This kind of dosing leads to good oxygen transfer and allows for nitrification (Cooper et al. 1996). The major purpose of macrophytes in VF CWs is to help maintain the hydraulic conductivity of the bed.

The removal processes are the same as in HF CWs, but VF units are predominantly aerobic, and therefore, the removal of ammonia is very high. On the other hand, there is little or no denitrification present. Removal of organics and suspended solids is high, but removal of phosphorus is generally low unless media with high sorption capacity are used. This, however, would increase the investment costs and therefore waste products with high sorption capacity such as blast and electric arc furnaces steel slags have been evaluated (Vohla et al. 2011).

VF CWs are very often used for on-site treatment of domestic wastewater, but their use for other types of wastewater such as landfill leachate, dairy, cheese factory, abattoir, airport runoff, or refinery process waters have been recorded (Vymazal and Kröpfelová 2008).

Fig. 4 Constructed wetland with horizontal subsurface flow for 800 PE (population equivalent) at Čistá, Czech Republic planted with *Phragmites australis* (Photo credit: Jan Vymazal © copyright remains with the author)

Fig. 5 On-site vertical flow CW at Bexhill, NSW, Australia, shortly after plantation (Photo credit: Jan Vymazal © copyright remains with the author)

Hybrid Constructed Wetlands

Various types of constructed wetlands may be combined in order to achieve higher treatment efficiency, especially for nitrogen. The concept of hybrid CWs was developed as early as during the 1960s in Germany and consisted of a series of VF beds followed by a series of HF beds (Seidel 1965). In aerobic VF beds ammonia is oxidized to nitrate, and in the following anoxic HF beds nitrate is reduced via denitrification. During the 1990s, HF-VF CWs were introduced; these systems need recirculation of nitrified effluent from aerobic VF bed to the HF bed inflow where high concentration of organics can support denitrification in anoxic HF bed. Recently, hybrid constructed wetlands of both types have become very popular, and they are used for many types of wastewater including municipal wastewater, landfill leachate, diaries, fish, and shrimp aquaculture recirculating systems or wineries (Vymazal 2011a).

Future Challenges

After five decades of research and implementation, CWs have been recognized as a reliable wastewater treatment technology around the world and, at present, they represent a suitable solution for treatment of many types of wastewater. Constructed wetlands have drawn the attention because of their high treatment efficiency and very low operation and maintenance costs. Most former concerns regarding their safe and reliable application have been refuted, namely, the concern over the performance during winter periods in cold climate. The ongoing research is focused on improvement of phosphorus removal, modeling of treatment processes aimed at the optimization of design parameters, and characterization of microbial assemblages responsible for water purification.

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Managing Phosphorus Release from Restored Minerotrophic Peatlands

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Abstract

The nutrient pollution of water bodies is a globally distributed environmental problem. An important strategy to mitigate the non-point phosphorus and nitrogen pollution is to restore minerotrophic riparian peatlands (also termed 'fens'). Originally natural fens served important functions as sinks for nutrients and as hydrological buffers for downstream systems leading them to be referred to as the 'kidneys' of glacial landscapes in the Northern Hemisphere. However, long-term drainage and intensive agricultural use of such peatlands, in some European countries more than 90% have been drained, has induced severe changes in their physical and geochemical soil properties. Today, in face of pollution of water bodies, dramatic loss of animal and plant species and expected global warming there exist major attempts to restore the "kidneys of the landscapes".

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However a full rehabilitation of lost ecological functions can be retarded for several decades (or centuries) in particular if degradation of upper soil layers is advanced. This paper provides empirical evidence based on field and lab experiments in Germany on the implications and effectiveness of different restoration strategies.

Keywords
Helophytes · Rewetting · Topsoil removal · Water pollution · Wetlands

Peatland Restoration and Processes Controlling Phosphorus Release

The eutrophication of freshwaters is a globally distributed environmental problem. An important strategy to mitigate the eutrophication of freshwater systems due to nonpoint phosphorus (P) pollution is to restore minerotrophic riparian peatlands (also termed “fens”). Originally natural fens served important functions as sinks for nutrients and as hydrological buffers for downstream systems leading them to be referred to as the “kidneys” of glacial landscapes in the Northern Hemisphere. However, long-term drainage and intensive agricultural use of such peatlands (in some European countries more than 90% have been drained) has made it unlikely that the original P sink function and low-nutrient conditions can be reestablished within a human time perspective (Zak et al. 2010). Drained fens, or those which experience strong drying/rewetting cycles, show elevated decomposition rates because of aerobic conditions resulting in a nonreversible transformation of formerly preserved substances, including P bound in refractory organic compounds, to more mobile inorganic P forms (Zak et al. 2008). In addition, significant changes can be observed in peat properties, such as a high degree of peat decomposition, elevated dry bulk densities, and lowered hydrological conductivities. These changes are most notable in the upper soil layer (approximately 0–40 cm) of fens, since this layer is mostly influenced by desiccation/aeration and agricultural use. The process of peat mineralization/peat loss and subsequent risk of eutrophication of surface waters (and the loss of stored carbon) can only be mitigated through the rewetting of drained fens, i.e., through the reestablishment of permanent waterlogged soil conditions (Lamers et al. 2014). However, following rewetting, a rapid lowering of redox potential within a few days or weeks, due to anaerobic microbial respiration, leads to a dissolution of P bound to redox-sensitive iron (Fe) compounds, and hence, to an increase of P concentrations in pore waters of rewetted fens (Lamers et al. 2014). In addition, two other reactions can be important: the release of either loosely bound P as a chemical equilibrium reaction between peat and pore water or of organic bound P due to hydrolytic cleavage of particulate organic matter (Robinson et al. 1998).

However, the level of P concentrations in pore waters of rewetted fens can vary widely. Values between 0.04 mg P/L and about 13 mg P/L are reported (Rupp et al. 2004; Zak et al. 2004; Tiemeyer et al. 2007). In particular, high P concentrations can be expected in fens with highly decomposed peat in the upper soil layers (Zak and Gelbrecht 2007). These are characterized by a high content of P bound to redox-sensitive Fe(III) compounds (Zak et al. 2008). The high P concentrations in rewetted fens interfere with the restoration goal to reestablish conditions for nutrient storage and can even create an additional P load to downstream watercourses (Tiemeyer et al. 2007). However, high internal P mobilization in rewetted fens cannot be simply equated with a high P load to downstream freshwater systems. Comprehensive investigations on P retention on the fen surface (the redox interface) have shown that an enhanced P export to adjacent surface waters can only be expected if the molar Fe/P ratios are lower than 3 in anoxic fen pore water (Zak et al. 2010). Accordingly at high molar Fe/P ratios in anoxic pore waters (>3), P will be almost completely retained in previously heavily drained fens. In such cases, the high initial P concentrations in anoxic pore waters (360–15300 μg/L) decreased to concentrations lower than 30 μg/L at oxic conditions, consequently corresponding to a "good water quality status" according to the European Water Framework Directive. That means a substantial P export to waterbodies can be widely excluded. The determination of Fe/P ratios of peat can be used as a predictive tool for the assessment of either Fe/P ratios in pore waters of rewetted peat soils or net P fluxes at the peat/surface water interface.

As a rule of thumb, if molar Fe/P ratios in peat are high (larger than 10) then high molar Fe/P ratios in pore waters (>3) are expected, which promote low net P fluxes in rewetted fens. In this case, the soil surface acts as a "P barrier" as long as this zone remains oxic (Fig. 1). Such coherence is well known for lake sediments and is often applied for lake restoration by adding iron compounds to reduce net P fluxes from sediments to water bodies (e.g., Jensen et al. 1992). However, analogous to lake sediments, net P fluxes can rise greatly during anoxic conditions in overlying water bodies. This situation can be found occasionally in large-inundated areas which have experienced severe soil subsidence and peat loss (up to 1 m and more) due to long-term drainage and intense agricultural use (Lamers et al. 2014).

Another important retention process in rewetted fens can be the P uptake by helophytes. Due to their nutrient removal capabilities, helophytes are often used in constructed wetlands to purify wastewater (e.g., Álvarez and Bécares 2006). However, most of the plant P stock will be released after dieback at the end of the growing season through leaching and mineralization (Koerselman and Verhoeven 1992). Consequently, helophytes contribute to the eutrophication of inundated peatlands by "smuggling" P from the rooted soil layer across the redox interface at the soil surface into the overlying surface water. Mowing and removal of plants would interrupt the P recycling process after dieback of plants. Whether this measure is effective in restoring rewetted fens as low-nutrient systems depends on the P uptake of the dominant helophytes in relation to the amount of P available in the upper, highly decomposed peat layer. Investigations have shown that the P uptake by different helophytes (*Phragmites australis*, *Typha latifolia*, *Glyceria maxima*, *Carex acutiformis*, *Carex riparia*, and *Phalaris arundinacea*) was in the range of the P mobilization rates found in highly decomposed peat soils (range: 0.8 – 15.6 g

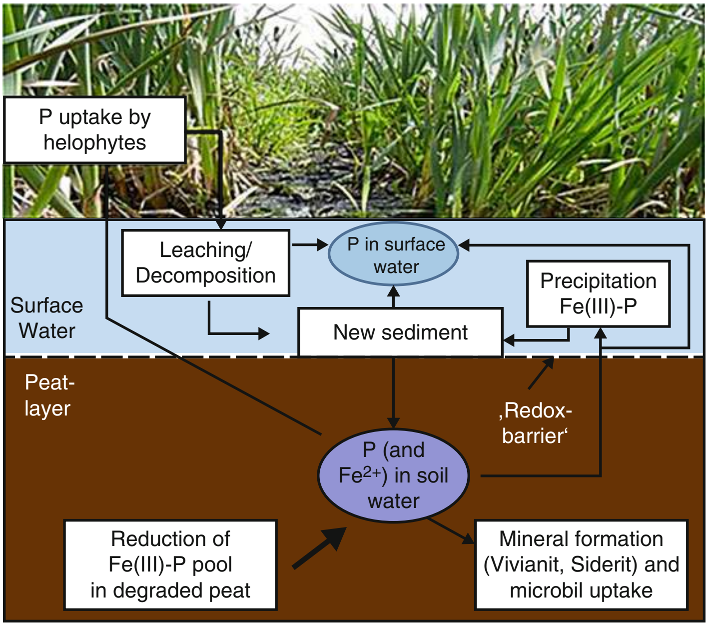

Fig. 1 A conceptual scheme of different phosphorus (P) pools and processes controlling the internal P release in rewetted (inundated) fens (original figure based on information in Zak et al. 2014)

P m^{-2}, n = 30) but four- to 10-fold higher than diffusive net P fluxes at the interface between soil and surface water (Zak et al. 2014). Accordingly, helophytes are able to compensate for the high P mobilization in degraded peat soils during the growing season by incorporating this P into biomass (Fig. 1).

Management Perspectives

Considering the potential side effect of high P release in waterlogged peat soils, it is recommended to combine the rewetting of fens with other measures to accelerate fen restoration and to mitigate the risk of P pollution of downstream aquatic systems.

1. Harvesting of helophytes, for instance, would be most effective at the beginning of flowering since biomass and P concentrations are at their peak (Meulemann et al. 2002). This suggestion might pose difficulties in mixed species assemblages. However, if, for example, *Phragmites australis* is dominant, the time of harvesting should be based on this species. However, calculations for *Phragmites australis* show that about 30 years would be required to exhaust the plant-available P at a highly degraded peatland site. Mowing therefore will often be the only realistic measure unless there is a long-term commitment to biomass harvesting or if an economic use can be found for the harvested plant material.

2. To enhance P sequestration, harvesting (before flowering) without plant removal might be also an option. If above-ground biomass causes rapid oxygen depletion at the inundated soil surface, then the decomposition of organic matter would be slowed. This in turn would accelerate the silting up of these shallow lakes on degraded fens, the storage of organic P in newly formed mud layers, and finally also the recolonization of peat-forming plants adapted to weak or noninundated conditions (Steffenhagen et al. 2012; Cabezas et al. 2014; Zerbe et al. 2013).
3. Depending on restoration targets, the high internal P release might be a constraint to the restoration of other fen ecological functions, including the recolonization with lost peat-forming plants such as low sedges or brown mosses. The literature suggests that removing the highly decomposed peat layer or sod-cutting is probably the most time-effective measure to restore fens, often producing positive results within a couple of years (Klimkowska et al. 2007), but it also can incur significant costs; however, the further use of the removed substrates, e.g., for commercial purposes, still needs detailed consideration. There is some evidence that top soil removal is an effective method to minimize the risk of internal P mobilization in rewetted peat soils as well as the P export to downstream systems. Laboratory experiments have shown that P release in rewetted fens with top soil removal can be reduced by a factor 100 (Cabezas et al. 2014).This method is recommended if highly sensitive (regarding P input) freshwater systems are present downstream.

Conclusions for Fen Restoration

The knowledge of the processes controlling P release (Fig. 1) and their relative importance over time is essential for securing the best management option to control P release in rewetted fens. While this remains a theoretical approach, it can be essential to support water authorities and landscape managers deliver sound environmental decision making. Additionally, other practical aspects such as economical and social issues which may vary strongly, even on a local scale, and restoration measures must be implemented as one element within an overall realistic management plan (e.g., Comin et al. 2014; Lamers et al. 2014). Based on the existing knowledge and also considering the realities of financial constraints on restoration projects, the following approach is recommended:

1. The large-scale rewetting of fen areas, particularly in intensive agricultural catchments, is an important measure to reduce the nonpoint pollution of freshwaters. Therefore a permanent waterlogging of peat soils is necessary to stop the fast aerobic decomposition of peat soils and to support peat-forming processes as long-term P sinks. This usually involved abandoning the former polder system and rewetting large areas by damming drainage ditches, opening dikes, and decommissioning water pump stations. In the case of highly degraded fens, which are characterized by substantial soil subsidence due to peat shrinkage, such waterlogged conditions can be only maintained under permanently flooded

conditions. The recolonization by peat-forming plants can be retarded due to intolerance to high water tables; however, incomplete decomposition of above ground plant biomass of helophytes as well as of submerged or floating plants can lead to the formation of mud layers which also contribute to the long-term storage of P (Steffenhagen et al. 2012; Cabezas et al. 2014).

2. Preliminary investigations of different particulate P fractions of upper degraded peat soils can be a useful tool to asses both the P mobilization potential as well as the risk of higher P export to downstream aquatic systems before fen rewetting (Zak et al. 2008, 2010). Such preinvestigations are highly recommended if the upper peat soils of fens are highly decomposed and/or in the case of sensitive downstream receptors such as nutrient-poor lakes. The assessment of the P mobilization potential of the highly decomposed peat layer is also useful to estimate the theoretical increase of P concentrations in adjacent waterbodies with a known nutrient status. For a general assessment of the impact of fen rewetting, all P fluxes in the catchment must be taken into account.
3. Harvesting of helophytes as temporal P sinks, along with submerged or floating macrophytes, in inundated fens can be useful to mitigate both the internal P release and the P export and also to accelerate the reestablishment of nutrient-poor conditions in degraded peat soils. Such measure might be in particular meaningful if costs can be reduced or even compensated by a commercial use of the harvested material. Top soil removal is highly effective but also carries high costs. This measure should only be applied if a risk of high P export cannot be excluded, which is the case in "iron-poor fens" (i.e., molar Fe/P ratios of peat are smaller than 10), or if nutrient-sensitive waterbodies are present downstream. The addition of iron to the peat surface instead (akin to practices common in lake restoration) might be useful in the short term; however, it can be expected that the internal P release will be maintained for much longer periods (centuries instead of decades) due to the repeated dissolution of redox-sensitive Fe(III)-P compounds, and finally iron (but also calcium) may accelerate the peat decomposition and thus release organic P.

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Managing Urban Waste Water

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Sally Mackenzie

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Abstract

Urban wastewater can be defined as domestic waste water or the mixture of domestic wastewater with industrial waste water and/or runoff rain water. For centuries, wetlands have been used to informally receive wastewater relying on their natural remediative abilities to remove pathogens and breakdown nutrients. However, unregulated and unmanaged discharges to urban wetlands, which are generally subject to a range of over pressures such as drainage for agriculture, reclamation for urban expansion, or exposure to high-strength industrial effluents, can lead to a loss of natural wetland functioning and decline in their capacity to process wastewater as well as provide other vital services. Integrated management of urban wetlands needs to take into account all the ecosystem services that a wetland provides to people through stakeholder engagement and community engagement and should be designed to ensure that the wetland is managed to function effectively.

Keywords

Wastewater treatment · Functions · Pathogens · Nutrients · Integrated management · Stakeholder engagement

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Introduction

Urban wastewater can be defined as domestic waste water or the mixture of domestic wastewater with industrial waste water and/or runoff rain water (Urban Wastewater Treatment Directive 91/271/EEC). For centuries, wetlands have been used to informally receive wastewater relying on their natural remediative abilities to remove pathogens and breakdown nutrients. However, unregulated and unmanaged discharges to urban wetlands, which are generally subject to a range of over pressures such as drainage for agriculture, reclamation for urban expansion, or exposure to high-strength industrial effluents, can lead to a loss of natural wetland functioning and decline in their capacity to process wastewater as well as provide other vital services.

Integrated management of urban wetlands needs to take into account all the ecosystem services that a wetland provides to people through stakeholder engagement and community engagement and should be designed to ensure that the wetland is managed to function effectively.

The East Kolkata wetlands are the world's largest example of integrated management of wetland and urban wastewater treatment. Wastewater generated from the city is fed through a series of fish ponds and also used for agricultural irrigation, and the wetlands in turn provide livelihoods for around 20,000 families. The East Kolkata Wetland Management Authority arose from the passing of an Act in 2006 to protect the wetland from urban pressures and ensure it continues to function effectively and provide ecosystem services (more information available at: http://www.ekwma.com and http://www.ramsar.org/pdf/KUMAR_UNHABITAT_East_Kolkata_Wetlands_PDF.pdf). The wetland was declared a Wetland of International Importance under Ramsar Convention in 2003 and is also cited as an example of "Wise Use of Wetlands."

At the tenth Conference of the Parties to the Ramsar Convention (COP10), the Parties adopted Resolution X.27 Wetlands and Urbanization. This resolution notes that "urban wetlands" are those wetlands lying within the boundaries of cities, towns, and other conurbations and that "peri-urban wetlands" are those wetlands located adjacent to an urban area between the suburbs and rural areas, and recognizes that wetlands in urban and peri-urban environments can deliver many important ecosystem services to people, such as wastewater treatment.

The resolution further notes the important role that urban and peri-urban wetlands can play in communication, education, participation, and awareness for urban communities about wetlands, and provide safety nets for the communities living in these areas, both through the buffering effect of wetlands in riverine and coastal areas and through the role of wetlands in reducing impacts associated with climate variability.

Ramsar is concerned that many wetlands in urban and peri-urban environments are, or are becoming, degraded through encroachment of surrounding populations, pollution, poorly managed waste, and infilling or other developments and that these activities have diminished both the ecosystem services that urban wetlands can provide and the recognition of their value and importance by both decision makers and urban communities. This lead to the adoption of Resolution XI.11 on Principles for the planning and management of urban and peri-urban wetlands at COP11 in 201.

Urban Wetland Management for Sustainable Integration of Wastewater Treatment Objectives

The Ramsar Handbook, Managing Wetlands (2010) gives useful guidance for developing an integrated management plan. The main components of a successful integrated plan are:

- To identify the ecosystem services the wetlands provide
- Consideration of wetlands within urban planning which should also incorporate water resource management, development of transport infrastructure, and agricultural production
- Alternatives for urban development so that wetlands are not degraded or lost through development

Case Study

The That Luang Marsh at 20 km^2 is the largest urban wetland in Laos. Situated on the outskirts of the capital city, Vientiane, the services it brings to the people living in urban and peri-urban areas has been valued at just under $5 million (Gerrard 2004) and include flood control, sanitation, and livelihood provision. Around $70,000 of this is for wastewater treatment services. The marsh is under pressure from agricultural expansion and urbanization which in turn affects local livelihoods and vital services of water treatment and flood storage (Kyophilavong 2008). In addition much of the wetland biodiversity in the area has been lost including Siamese crocodile, herons, otters, and snakes.

It was therefore necessary to integrate the sustainable management of wetlands into the larger urban planning and decision-making process of the city. Building on the initial valuation study, WWF, STEO, and the Wildfowl & Wetland Trust (WWT) secured a grant from the European Commission to examine the feasibility of creating a constructed wetland treatment system to treat domestic and industrial wastewater while maintaining the important functions of flood detention and food web support. The project – Wastewater Treatment through Effective Wetland Restoration (WATER) – contributed to the capacity and understanding of the need to integrate wetlands into urban development plans by decision makers in Vientiane.

The project activities included training over 20 technical staff in participatory processes in wetland planning, management and monitoring, constructed wetland design and operation, and promoting the scope and value of wetlands to urban environments to over 15 high-level policymakers through stakeholder consultation, involvement in the project steering committee, and study tours in the region. The

City of Vientiane Capital is now reviewing the constructed wetland designs and looking for donor support. In conjunction with this, the Food and Agriculture Organization (FAO) has proposed a That Luang Marsh Master Plan – a document written in collaboration with local residents and government ministries which will determine the future of the That Luang Marsh ensuring that all development is in concert with the unique cultural, historical, socioeconomic, and ecological assets of the area (Gerrard 2010).

The case study of Vientiane demonstrates the need to integrate the principles of wetland management across different sectors of local government and across wider societal stakeholders.

Future Challenges

The main challenge to successful management of urban wetlands is to balance the demands placed on the services of the wetland provides. A lack of capacity to carry out an assessment, for example, using an ecosystems-based approach, means that services may be missed or undervalued.

To ensure all information on wetland functioning is captured, all stakeholders must be included in the consultation and decision-making process.

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Section XII

Management of Cultural Services

Mark Everard

Cultural Aspects of Wetland Management: An Overview

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Thymio Papayannis and Dave Pritchard

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Abstract

Wetlands have given rise to a rich variety of human cultural values, belief systems and associated practices. Safeguarding this heritage, and the role of culture in ensuring future resilience of human societies, depends on protection and wise management of the wetland environments that produce it. Conversely, a better understanding of these cultural aspects, and an integrated approach to management, is key to the successful conservation of most wetlands. This chapter examines the ways in which this can be achieved.

Keywords

Cultural values · Cultural services · Integrated management

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Introduction

Global experience during the past three decades has demonstrated that effective nature conservation depends critically on human beings through their individual and collective attitudes and actions. It has also been understood and broadly accepted that positive conservation results cannot be achieved and sustained without parallel care for human wellbeing, and good understanding of the relationship between the two (Howard and Papayannis 2007).

Nowhere is this more evident than in the case of wetlands. These sensitive ecosystems, which help to maintain other ecosystems (notably through regulation of the water cycle), support rich biodiversity and provide invaluable services to humanity, are nevertheless being degraded rapidly as a result of anthropogenic pressures. Total losses during the twentieth century are estimated to be as high as 50% (MA 2005). Societies throughout history have lived intimately with wetlands, have developed invaluable knowledge for their management and made good use of their multiple services, while incorporating wetlands and water in their cultures. This has however altered radically in the case of many contemporary societies, which often view wetlands negatively, for example as a source of disease – in particular malaria – or as empty, flat spaces to be occupied for other purposes, such as agriculture, industrial installations, urbanization, tourist facilities, and transport infrastructure.

The cultural aspects of wetlands arise from the diverse interactions between human beings and these sensitive and highly productive ecosystems. Much of this is intimately related to the ecosystem services provided by wetlands to humanity, which underpin the wide range of values associated with these systems. These values are in turn the basis for arguments in favor of wetland conservation, but they need to be better recognized, maintained, and enhanced. This is particularly true in the case of cultural values, which are often less visible than the provision of water, food, and fiber, and/or arise in more indirect ways from the physical functioning of the ecosystem.

This necessary breadth of perspective however does inform the way in which the Convention on Wetlands (Ramsar 1971) has incorporated cultural values and services into its core concept of the "wise use" of wetlands, which is synonymous with sustainable use (Finlayson et al. 2011). During the past 15 years, the Convention has given more focused attention to this question. In 2002 and 2005, the governing Conference of Contracting Parties (COP) approved two formal Resolutions (VIII.19 at COP8 in Valencia, Spain, and IX.21 at COP9 in Kampala, Uganda) on the incorporation of cultural aspects in the management of wetlands. In 2006, the Convention established a Culture Working Group to coordinate activities on this subject and, in 2008, it issued a major guidance document on Culture and Wetlands (Papayannis and Pritchard 2008). The Convention also has formal cooperation arrangements with UNESCO, including special attention to sites designated under both the Ramsar Convention and the World Heritage Convention and/or UNESCO's Man & the Biosphere Programme, thus further institutionally recognizing the interplay between nature and culture.

The Variety of Cultural Values and Services

An extremely wide range of human activities is related to wetlands in various ways, and some structuring of this variety is a prerequisite for coherent and prioritized management strategies. Key categories include habitation, primary and secondary exploitation of wetland resources, and more intangible aspects. The large majority of these activities generate culture in its broadest sense. Not all, however, are positive for conservation. For example, certain traditional hunting and fishing practices, although embedded in the culture of specific societies, are destructive to wild fauna.

Based on the classification included in the Ramsar Guidance on Culture and Wetlands (Papayannis and Pritchard ibid.), wetland-related activities can be grouped into four categories for reasons of convenience.

Habitation: As demonstrated by archeological research in prehistoric sites, human beings for millennia have lived in wetlands or in close proximity to them, making use of their natural resources (Current Archaeology 2001; European Archaeological Council 2000). Lacustrine settlements – as found in the Palafitte network in Central Europe or in Dispilio on Lake Castoria in Northern Greece – were built over the water on piles, probably for reasons of security. In historic times, security of settlements continued to be an important factor, along with the availability of freshwater and the facilitation of marine transport. Thus, quite a few urban centers grew along rivers and shorelines, or in lakes and lagoons (such as Boston, Budapest, Tunis, Bangkok, Dar es Salaam, and Alexandria).

The combination of the natural elements of wetlands with the works necessitated by human habitation has created unique cultural landscapes, as indicated by the iconic case of the city of Venice. Wetland natural elements include habitats based on moist soils, water (with its variety and mobility), and often rich biodiversity, while anthropogenic works – sharing the same landscapes – comprise archaeological sites of various periods, villages and towns, historic buildings, infrastructure facilities (such as roads, harbors, water transportation and distribution works, airports, and other more recent constructions). Dynamic seasonality (including alternating dry and wet seasons, or cyclical patterns of inundation) is a common feature. Thus, wetland landscapes frequently exhibit a great diversity and complexity of human habitation-related uses and require careful spatial planning and management to maintain both their cultural and natural values (Plieninger and Bieling 2012).

The values associated with this have been acknowledged by the UNESCO World Heritage Convention, which has recognized and included a number of wetlands in its list of sites qualifying under the Convention's criteria for outstanding universal value (OUV).

The global demographic trend towards increased city dwelling has led to an increased urbanisation of many of these wetland environments, compounded by the growing need for space for infrastructure, industrial and commercial facilities, tourist installations, and other human uses. Wetlands have often been perceived as offering large flat areas with few impediments to such constructions, and this has gone hand in hand with destruction for other reasons, such as attempts to reduce the incidence of malaria. Thus, urbanization has become one of the main threats to wetlands, and a cause of loss of cultural as well as natural values and services.

In the efforts for wetland conservation and wise use, cultural aspects (such as recognized archeological sites, historic buildings, traditional knowledge, and sustainable water management practices) as well as international designations (UNESCO's World Heritage sites and Biosphere Reserves) provide strong reasons for integrated responses that draw on all of these aspects of ecosystem-based value, and not only biodiversity-related evidence.

Wetlands are highly productive ecosystems and not only from the biodiversity perspective. They produce resources that are of considerable use to humanity and are critical for subsistence in many societies worldwide. The exploitation of these resources has also in many cases led to the development of knowledge, traditions, and practices which in themselves constitute values of a cultural character.

Abiotic resources include mainly water, salt, sand, and minerals. Water from wetlands is used for irrigation and domestic consumption. Its distribution has often required ingenious constructions that have become part of wetland cultural landscapes, along with some sophisticated management arrangements which play a strong role in distinctive cultural characteristics. Salt extracted from coastal wetlands through traditional salinas has resulted in the formation through the years of unique cultural landscapes, celebrated in certain cases by salt museums, as in Sekovlje Soline in Slovenia (Fig. 1). Salinas in turn have distinct conservation values for a number of rare species. The extraction of sand, notably for admixture in concrete, must be managed with great prudence, as excessive sand removal tends to impact negatively on wetland systems. The same can be said of other minerals. Gold extraction from streams has been a traditional activity with its own cultural aspects, but has often been unfavorable to wetland conservation (Hill 1999).

Biomass cultivated in wetlands has provided fresh fodder for domestic animals and grazing in wet areas is associated with indigenous breeds of cattle and of buffaloes. Reeds in particular can be used as fodder when fresh, while their shoots can be dried for the winter months; and when grown they are an important building material. More recently, through conversion into pellet form, new ways of using reeds in energy production are being found.

Of the edible products cultivated in or around wetlands, rice is the most important, with field flooding from wetland water having been harnessed over centuries for this purpose. Rice cultivation is found in many parts of the world and has been the driver for the creation of several distinctive forms of cultural landscapes and practices (Fig. 2).

The medicinal qualities of some wetland flora species has been recognized by ancient and traditional societies, as far back as Hippocrates and ancient civilizations in China and the Amazon, and continuing today to feature in ethnomedicinal practices.

Many species of fish, molluscs, crustaceans, amphibians, mammals, and a large variety of waterbirds that breed in wetlands, reside there outside the breeding season or pass through on migration, are hunted by humans either for subsistence or for sport, or are exploited commercially, through a multitude of fishing, hunting, and

Fig. 1 Sekovlje Soline saltworks in Slovenia (Photo credit: P Hieng © copyright remains with the author)

harvesting practices. Some of these practices, although they may appear unsympathetic to some eyes in terms of environmental sensitivity or animal welfare, are deeply embedded in local culture. For example, the use of hawks for hunting birds or small mammals is a venerated tradition in the Middle East. In a similar way, ligatured cormorants are used for fishing in certain areas of South-East Asia. Excessive exploitation of wetland resources undermines conservation objectives. Thus a key priority of wetland management is to establish an appropriate balance in the use of resources and to ensure that it is environmentally sustainable in the long term.

Resource utilization: Wetland resources and their exploitation can give rise to secondary uses, representing a further set of ecosystem services. In their traditional forms, many of these secondary uses incorporate invaluable knowledge that must be recorded and maintained, as it can provide guidance for the present and the future.

The processing of wetland-derived food products through the ages and throughout the regions of the world has created a rich gastronomy, which constitutes one form of cultural service. Salt from wetland salinas, as an essential element in this, has been linked throughout history with the development of human communities (Papayannis and Pritchard 2011). The case of the Neretva Delta in Croatia, in which wetland dishes have been incorporated in a tourist package, indicates how culinary values can be exploited in a sustainable manner for other purposes too (Viñals 2002).

Fig. 2 Rice field in South China (Photo credit: G Saxton, NESDIS, NOAA © copyright remains with the author) [TP1]

A number of highly sophisticated crafts have been developed based on wetland resources. Many of these relate to boats and fishing methods. Traditional boats for wetland fisheries have been constructed using material extracted from the wetlands themselves, including wood from riparian forests. A characteristic example is the boats constructed from reeds in Lake Titicaca in Peru. Unfortunately, traditional boats in many regions are being replaced with standardised mechanical forms, with the traditional forms in some places being retained only for rare use for visitor transport (for example in the Neretva Delta (Fig. 3)). Fishing gear itself often involves culturally distinctive crafting skills, including the use of local materials, as illustrated by the ingenious fish traps used in the Niger River in Mali (Fig. 4). Agricultural and construction tools also feature frequently as valuable historical artefacts.

Wetland materials, and especially reeds, have been used for centuries in building construction, and roof thatching is still an existing practice, for example in some parts of Europe. Construction with reeds reached its pinnacle in Southern Iraq among the Ma'adan people, also known as the Marsh Arabs, and it has given rise to an architecture of extreme lightness and beauty, perfectly adapted to wetland conditions (Fig. 5). Unfortunately, this tradition is being rapidly lost, along with the wider cultural heritage of the Ma'adan.

The three cultural aspects mentioned above (gastronomy, traditional crafts, and special constructions) are very well able to complement tourist packages that focus

Fig. 3 Traditional boats in Neretva Delta, Croatia (Photo credit: T Papayannis © copyright remains with the author)

Fig. 4 Fish traps on Niger River, close to Mopti, Mali (Photo credit: T Papayannis © copyright remains with the author)

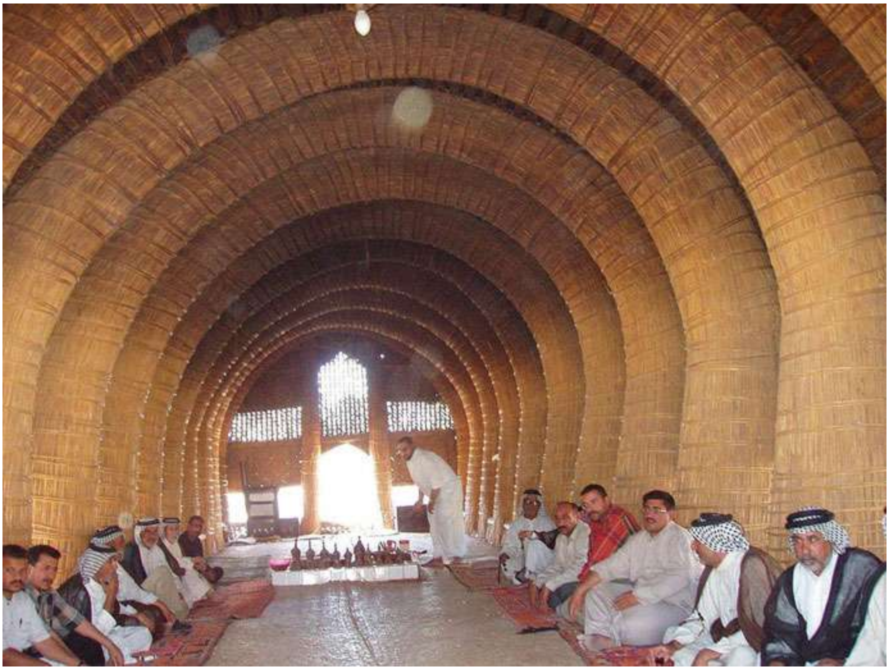

Fig. 5 Ma'adan reed architecture in southern Iraq (Photo credit: H Janali, US Army Corps of Engineers © copyright remains with the author) [IL2]

on wetlands, with the addition of visits to adjoining archeological sites and to historic buildings related to wetland activities. Unfortunately, such well-integrated approaches to tourism are not common, as the attraction potential of wetlands has not been sufficiently explored. It is hoped, however, as demand for special forms of tourism grows, that much better use will be made of sensitive promotion of the wetland heritage.

This possibility can be enhanced by various forms of sporting and leisure activities that are compatible with wetland conservation. Canoeing and sailing for example may often be acceptable on this basis, but the scope is less where power boats and jet-skis are concerned. Hunting has often been presented as a cultural endeavor, managed sustainably in favor of wetland conservation. In some sites however, although it may provide economic benefits for a time, it can ultimately have a negative long-term impact on wildlife populations if disturbance intensities or rates of take are too high. Perhaps the most positive leisure activity that wetlands make possible is visiting on foot and experiencing the natural and cultural heritage in low-impact ways.

Traditional wetland management practices include those concerned with the allocation of water resources, and the intricate but effective management of water systems in the Sahara oases and the maintenance of the *foggara* networks are an example: these have withstood the pressure of time for many centuries but their sustainability is now being challenged by younger generations and modernization trends.

Wetland-related activities have also given rise to public social events, often focusing on fisheries, such as the Tarabusco Festival in Orbetello, Italy.

Intangible benefits: At a more intangible level, spiritual, aesthetic, and other cultural significance has often arisen in situations with a specific focus on wetlands, and often in close association with traditional wetland management practices. Though less tangibly manifested, these values can nonetheless have profound impacts on social cohesion, local economies, political support, and other key areas of benefit.

Throughout human history, wetlands have been an important focus for the evolution of knowledge concerning the functioning of the natural world. In various parts of the world, there are research centers focusing on wetlands, for example at Tour du Valat in the Camargue, France [www.tourduvalat.org], Howard T. Odum Center for Wetlands, Florida University [www.cfw.ufl.edu], and the Greek Biotope – Wetland Center in Thessaloniki, Greece [www.ekby.gr]. The knowledge that is developed can be communicated to the public through visitor centers, and these have been established in many wetland sites. Such centers also play a significant role in education on nature for various learning levels, from primary to university. Research and education activities in wetlands should where possible include their cultural aspects, from archaeology to ethnology and moral philosophy.

Research on traditional knowledge and practices concerning wetlands, including the management and use of their services, is especially important. A few systematic studies exist in this area (Gros and Frithz 2010). From the information available, however, it appears that traditional knowledge of wetlands is extensive and can provide significant insights into the environmentally sustainable approaches increasingly sought by policy-makers at all levels.

The spiritual values of water and wetlands have been well documented in the many publications on sacred natural sites (see Verschuuren et al. 2010 for an overview). In most religions, water plays an important role. Baptism is a key sacrament of the Christian religion, originating from the baptisms in the Jordan River by Saint John the Baptist. Ablutions before prayer are a daily practice among Moslems, and immersion in the waters of the Great Ganges River is a major yearly event for Hindus. Blessing of the waters is a practice not only of mainstream religions but also of many indigenous peoples, as shown by the joint ceremony during a mission of Ecumenical Patriarch Bartholomew in the Amazon in July 2006 (Fig. 6). Religious events, with the participation of large numbers of the faithful, are held even today in many parts of the world, such as the yearly pilgrimage to the Virgin of El Rocío through the wetlands of the Doñana National Park in Spain, the religious procession on boats in the Albufera marshes of Valencia, also in Spain, and the Loy Krathong festivals in various parts of South-East Asia.

On the artistic side, wetlands have been traditional sources of inspiration because of the beauty of their landscapes, their diversity and dynamism, acoustic properties, and three-dimensional access possibilities. All of these have for example been well recognised by film directors, among others. Historically important paintings that

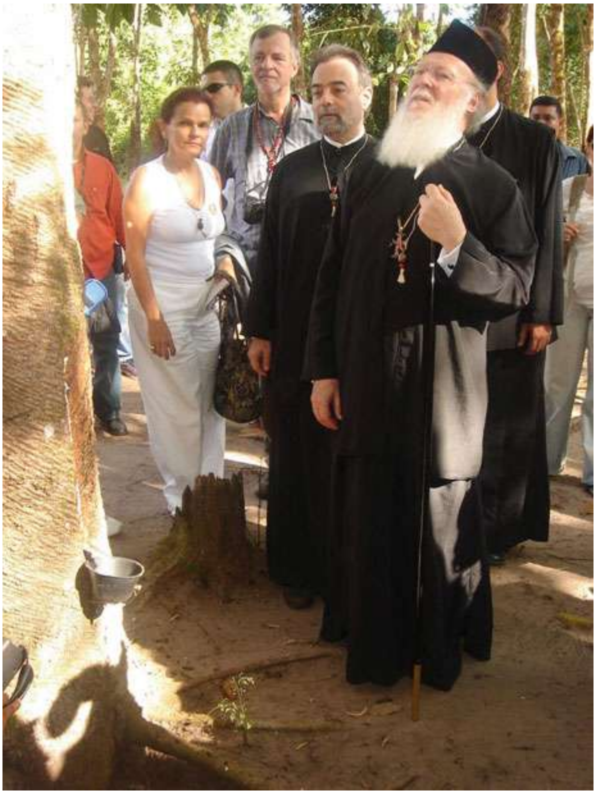

Fig. 6 HAH Ecumenical Patriarch Bartholomew in the Amazon, July 2006 (Photo credit: T Papayannis © copyright remains with the author)

have been inspired by wetlands include Raphael's "The Miraculous Draught of Fishes" (1515) and "Moses Saved from the Water" (1518), Pieter Breugel's "Landscape with the Fall of Icarus" (1558), Pierre-Auguste Renoir's "Boating on the Seine" (1879–1880), Georges Seurat's "Bathers at Asnières" (1884), and Claude Monet's "Water Lilies" (250 oil paintings, 1897–1926).

Many works of music similarly have strong associations with water and wetlands, such as "Waters of Babylon" by Johann Sebastian Bach (1730), "In the Lake" by Johannes Brahms (1888), "Fleur des Eaux" by Claude Debussy (1883), "Water Music Suites I, II, III" by Georg Friedrich Handel (1715–1736), "Jeux d'Eau" by Maurice Ravel (1901), "Auf dem See" by Franz Schubert (1817), "Swan Lake" by Piotr Illych Tchaikovsky (1875), "The Water Sprite" by Jean Sibelius (1888), "Water Music" by John Cage (1952), "Curlew River" by Benjamin Britten (1964), and many others.

Wetland literature includes a variety of works that use these ecosystems as background. Among them are "The Merchant of Venice" by William Shakespeare (1596–1598), "Great Expectations" by Charles Dickens (1860–1), "The Adventures of Huckleberry Finn" by Mark Twain (1884), "Lorna Doone" by Richard Doddridge Blackmore (1869), "Heart of Darkness" by Joseph Conrad (1899), "The Hound of the

Baskervilles" by Arthur Conan Doyle (1902), "The Secret Garden" by Frances Hodgson Burnett (1910), "Death on the Nile" by Agatha Christie (1937), "The Pendle Witches" by Walter Bennett (1993) and the poems "I Wandered Lonely as a Cloud" by W. Wordsworth (1804) and "The Lady of the Lake" by Sir Walter Scott (1810).

Through the increased awareness and emotional engagement which results among wider audiences, these aesthetic and inspirational properties have contributed to public support for wetland conservation. There is great scope, including through use of ever-diversifying forms of new digital media, to make better use of artistic creativity in wetland conservation efforts, for example by motivating tourists to visit, providing interpretation on wetland values and vulnerabilities, and engaging local people in participatory planning and management activities.

Cultural Aspects in Wetland Management

Ancient societies and indigenous peoples tend to show much less of a dichotomy than our own between the cultural aspects of nature and the management and protection of natural resources, with all of these aspects often being the responsibility of the same individuals and structures, and/or being closely integrated in a philosophical and practical sense. Examples of this existing today include those Saharan oases which retain sophisticated communal forms of land and water resources management (Papayannis 2008).

In most contemporary societies, however, responsibilities for culture, nature conservation, and use of natural resources have become institutionally much more fragmented. Thus, in many countries the overall responsibility for wetlands may be divided under the central services of Ministries of Culture, Environment, Agriculture, and Forestry, each with their own priorities and policy mandates, often posing challenges for coordination. Similar divisions of responsibility may also be seen at other more local levels of public administration.

Traditional knowledge, however, more often tends to encourage an integrated management of natural ecosystems – including their cultural dimensions – so that the services they provide can be of benefit to human communities, while their functioning is safeguarded and sustained. Applying this knowledge in a contemporary context, three main approaches might be seen as particularly appropriate.

In the first, a loose and occasional coordination would be established between the experts managing natural protected areas, and particularly wetlands, and the scientists and technical experts – mainly archaeologists – responsible for safeguarding cultural heritage. While far from an ideal system, this could be viewed as one step closer to a fully integrated approach. In any case, a high degree of flexibility and persistence would be required to make it effective.

In a second approach, management bodies for protected natural areas – wherever they exist – would include, in both their governing bodies and executive staff, individuals with appropriate expertise on cultural aspects. This would allow gradually the development of harmonized attitudes and streamlined practices.

Those responsible for ensuring wetland conservation and wise use must be given appropriate training in this field. This need extends beyond wetland management experts to other sectors that have an impact on wetlands, such as spatial planning, water management, agriculture, forestry, and fisheries. Other stakeholders such as sacred site custodians, culture experts, scientists and educators, traditional local leaders, and artists need to become more fully aware of the specific role played by wetlands as the source of the services they already value. Wetland visitor centers, which exist in many major wetlands, can play a key role in documenting and interpreting information and understanding about cultural ecosystem services for the wider public.

In the third, fully integrated management bodies would be established, with multidisciplinary staff having responsibility across the whole of the conservation of both the natural and the cultural dimensions of each specific site. This would be closer to traditional practices in which responsibilities are integrated. Societal control of the management process must be ensured, so that decisions taken by the appropriate organs are recognized by the population in general and are felt to be just and equitable, which is key to ensuring adequate implementation and enforcement (Viñals et al. 2005). The ancient Water Tribunal in Valencia, Spain, is a characteristic example of how this has been achieved.

It is clear that the third approach offers the best prospects for effective and long-lasting results, but is not easy to implement as it would involve a somewhat radical and innovative change to current practice in most countries. In any case, the three approaches outlined above could be regarded as sequential steps towards a progressively more ideal and mature situation.

Future Challenges

At the policy level, nationally and internationally, there is a great need for synergy in the conservation of wetland natural and cultural heritage, and in the effective maintenance of the capacity of wetland ecosystems to continue to deliver the benefits to humans (ecosystem services) for which they are so important.

Perhaps the key initial step is to bridge the epistemological divide between the natural and the human sciences. Quite some progress has been made in recent years with publications that analyze critically the interface between humanity and nature, with some of them even becoming bestsellers (Jared 2005). Rapid progress would also depend on the development of more of a common vocabulary for use in integrated research and actions related to wetland systems.

Overcoming these intellectual obstacles would then make it more feasible to set common policy goals at the national and international levels. The Ramsar Convention, during its 46 years of existence, has played a major role in this direction. Its "wise use" approach to wetlands, which it has developed and promoted globally, is in essence an integrated approach. It has undergone a continuous process of fuller interpretation to take account of developing wisdom in wetland policy and management, including the cultural dimension. The message of the Convention has been passed through its Contracting Parties to the national level, where it has nevertheless not been always been well understood or effectively implemented. Greater efforts will be required to reach a satisfactory level of integrated management of wetlands at the national level.

Broader international collaboration would help. The growing relationship between the Convention on Wetlands and the World Heritage Convention is a positive sign in the direction of seeking coherence, as it focuses more and more on joint, concrete activities. International nongovernmental organizations – such as IUCN, Wetlands International, and WWF International – although focused predominantly on the conservation of the natural environment, have developed a growing sensitivity to human and cultural aspects. Consequently, as partners of the Ramsar Convention, they are able to contribute to the strengthening of its policies in this respect.

Once integrated management has been agreed, a number of key steps are required.

Traditional knowledge has generally embraced all aspects of wetland ecosystems – physical, cultural, and spiritual – as well as the relationships between them. In contemporary developed-world societies, however, knowledge is often sectoral and compartmentalized. Thus, collecting sound data about each river basin, coastline and wetland site and all its aspects is a prerequisite. This must be done through widely accepted scientific protocols, so that the results can be objective and comparable.

The real challenge though is to identify the relationships between the various parameters and to express these in measurable and operational terms. Analyzing relations and impacts between cultural and natural aspects and uses is highly sensitive, as there is limited experience in such complex transdisciplinary endeavors.

In traditional societies, it may be easier to understand and to agree on common goals because of increased social cohesion. To stimulate an equivalent understanding in a contemporary context would require agreement on mutually acceptable targets between conservation and cultural heritage sectors as a prerequisite of management planning. Such targets should include optimum conditions for public use of wetland resources and other wetland ecosystem services. This is not sufficient, however, as it would be imperative – in accordance with traditional knowledge and experience – to obtain public support for the targets, a process of raising public awareness and involving local communities. In the case of small rural communities, this might be more feasible. In large urban settlements, however, the process of sensitization, participation and building consensus would be much more laborious and uncertain, requiring large-scale and persistent efforts.

After targets have been set and agreed, activities should be planned for achieving them. These would involve not only the body in charge of wetland management but also bodies responsible for sectors such as spatial planning, agriculture, fisheries, water supply and distribution, tourism, archaeology, and others.

For all such activities that concern the public use of wetland resources and services – both natural and cultural – sustainability guidelines and regulatory measures need to be instituted by the public sector. The whole process of effective implementation of management actions and the enforcement of regulatory measures must be carefully designed. Balance must be established between allowing a *laissez-faire* attitude under which everything is permitted – thus leading to wetland degradation – and repressive measures that may provoke negative public responses. According to traditional knowledge, this can be achieved best through mechanisms of public involvement and consensual control.

In such an integrated approach, a monitoring system for ensuring feedback and evaluation of management initiatives – individually and collectively – must be established and applied responsibly. Normally, such monitoring is carried out by bodies responsible for wetland management. Evaluating one's own action is, however, not straightforward. In a traditional context, local communities and users of wetland resources and services would provide the necessary feedback. In a contemporary context, such assessments are more typically entrusted to institutions, including academic bodies and nongovernmental organizations (NGOs). Their role must be defined and made explicit within the wetland management process.

Finally, it should not be forgotten that wetlands, in their multiple aspects, must be enjoyed by the public, both local inhabitants and visitors. Securing this indefinitely into the future is perhaps one of the greatest cultural services we should strive to maintain.

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Cultural Services: The Basics

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Mark Everard

Contents

Abstract

The Millennium Ecosystem Assessment classified ecosystem services into four major categories: provisioning services, regulatory services, cultural services, and supporting services. Cultural services are defined as "…the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences". Heterogeneous ecosystems influence the diversity of cultures, spiritual and religious values, knowledge systems (traditional and formal), educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, and recreation and ecotourism. There is also a degree of feedback from cultural uses into ecosystem structure and function than can, in turn influence the services provided by ecosystems. Spiritual and other cultural values are as important as other services, but have been significantly compromised by ecosystem degradation, recreation and ecotourism placing pressures on ecosystems but also serving as potentially influential levers for their conservation. It is important to balance management for and uses of cultural services provisioning, regulatory and supporting services to ensure overall resilience and contributions to human wellbeing, including their contribution to poverty alleviation.

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Keywords
Nonmaterial benefits · Spiritual · Religious · Cognitive development · Reflection · Recreation · Tourism · Aesthetics · Knowledge systems · Poverty Alleviation

Definition

The concept of ecosystem services, describing the diverse benefits that the natural world provides to people, has been emerging as a pedagogic and management tool since the late 1980s. Since that time, disparate classification schemes have been developed often addressing specific habitat types (wetlands, coral reefs, rangelands, croplands, forests, etc.) and/or bioregions.

One of the many contributions of the UN's Millennium Ecosystem Assessment program (Millennium Ecosystem Assessment 2005a) was the harmonization of these prior schemes into a consistent classification system suitable for comparison of major habitat types on a global basis. The primary division within the MA classification scheme was the grouping of ecosystem services into four major categories: provisioning services, regulatory services, cultural services, and supporting services.

Cultural services are defined by the Millennium Ecosystem Assessment as "...the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences". These include the influence of heterogeneous ecosystems on the diversity of cultures, spiritual and religious values, knowledge systems (traditional and formal), educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, and recreation and ecotourism. There is in this a degree of feedback in that cultural factors also influence human wellbeing, as well as influencing the way that ecosystems are valued and used.

A number of subglobal assessments informing the MA found that the spiritual and cultural values provided by ecosystems were as important as other services, such as provisioning and regulating, for many local communities, both in developing nations (for example the importance of sacred rivers in India) and industrial countries (such as the importance of water features in urban landscapes). However, another conclusion of the MA was that 70% of regulating and cultural services are being degraded or used unsustainably across the world, highlighting the threats to continuing human wellbeing from ecosystem degradation. "Spiritual and religious values" were found to be significantly compromised by, for example, the rapid decline in sacred groves and species, while aesthetic values too were in sharp decline due to decreases in the quantity and quality of "natural" lands. The situation for recreation and ecotourism was equivocal, with some declines but other enhancements as more areas became accessible while many became degraded. Intensive exploitation of cultural services can have significant impacts on the environment, such as tourism impacts or disturbance of species of cultural significance, while the cultural significance of some fish and other wetland species can be a powerful lever for conservation of the habitats that support them (Everard and Kataria 2011).

A Millennium Ecosystem Assessment synthesis specifically considering global wetlands and water (2005b) (including lakes, rivers, marshes, and coastal regions to a depth of 6 m at low tide but acknowledging that many wetland types were underrepresented) found that "*. . .more than 50% of specific types of wetlands in parts of North America, Europe, Australia and New Zealand were destroyed during the twentieth century, and many others in many parts of the world degraded*". Nevertheless, wetlands produce a diversity of all categories of ecosystem services, including significant cultural services such as ". . .significant aesthetic, educational, cultural, and spiritual benefits, as well as a vast array of opportunities for recreation and tourism". The economic value of these cultural services may be significant, for example, with recreational fishing generating considerable income. For example, the Wetlands and Water synthesis report notes that, "35–45 million people take part in recreational fishing (inland and saltwater) in the United States, spending a total of $24–37 billion each year on their hobby. Much of the economic value of coral reefs—with net benefits estimated at nearly $30 billion each year—is generated from nature-based tourism, including scuba diving and snorkelling." The report also highlights how different types of wetlands produce a different balance of cultural and other services, so the heterogeneity of wetland types across landscapes is of importance for the conservation of these important services.

The Ramsar Convention's "wise use" concept (www.ramsar.org/handbooks4/) recognizes the need to balance the use of wetlands for the production and use of cultural services and all other provisioning, regulatory, and supporting services contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation.

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Cultural, Aesthetic, and Associated Wetland Ecosystem Services

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Thymio Papayannis and Dave Pritchard

Contents

Abstract

The chapter refers to the multiple ecosystem services provided by wetlands, both tangible and intangible, which include cultural, spiritual and aesthetic services. Examples are given of specific sites in which these services are particularly prominent. The challenges faced in their maintenance and appropriate use are also mentioned. Finally, the role of these services in wetland management, conservation and wise use are briefly explained.

Keywords

Ecosystem services · Cultural services of wetlands · Spiritual services of wetlands · Cultural values of wetlands · Wise use of wetlands

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Introduction

The concept of "ecosystem services" dates from the mid-1960s (Hernández-Morcillo et al. 2013), but came into much wider use during the 2000s principally through the work of the Millennium Ecosystem Assessment (MA). The MA's conceptual model divides ecosystem services into four categories: "provisioning," "regulating," "supporting," and "cultural" (MA 2005a).

Broadly speaking, ecosystem services are defined as the benefits that people derive from ecosystems. These benefits may be tangible or intangible. This "service"-oriented approach represents one kind of effort to address the widespread under-appreciation of humankind's dependence on healthily functioning ecosystems. It has, however, been challenged by those who champion "intrinsic" or "existence" values of nature, or who otherwise reject overly utilitarian views of nature's significance to humanity. These objections perhaps overlook the fact that such values are themselves human constructs (i.e., it is people who are recognizing and championing nature's intrinsic right to exist), and to that extent they are addressed by the way in which "intangible services" are included in the MA's conceptual model as part of the "cultural" category.

In the case of wetlands, all this thinking has been formally embraced in the context of the implementation of the Ramsar Convention on Wetlands (see for example MA 2005b). With such institutional endorsement, an improved vocabulary and methods for revealing and evaluating various kinds of intangible significance, the cultural services generated by wetland ecosystems have become progressively better appreciated, and in turn this understanding is, in places at least, strengthening the motivation for wetland conservation.

Characteristic Wetland Cultural Services

Again following characterizations developed for the Millennium Ecosystem Assessment, the following categories of cultural ecosystem services, all of which are applicable to wetland systems, can be recognized:

– *Spiritual and religious*: A number of societies attach spiritual and religious values to particular ecosystems or to components of them, such as certain sacred trees or animals. In wetlands, spiritual aspects are often related to the purifying presence of water, whether in a psychological sense or metaphorically through rituals of baptism and cleansing.
– *Recreation and ecotourism*: People often choose where to spend their leisure time based partially on the characteristics of landscapes and place-settings, which may involve natural features, cultural features, cultural associations, or very often a mix of all three. The diversity of wetland landscapes, with the complex fascination of the dynamics of water, along with often rich biodiversity, contributes greatly to the attractiveness of these places to visitors.

- *Aesthetic*: Individual human beings find aesthetic value in various aspects of ecosystems, as demonstrated by support for parks and other created greenspace, scenic drives, and preferences in selection of housing locations. Deep fascination with the aesthetic properties of water in the landscape again is a strong factor in this, as far as wetlands are concerned (although the large degree of inherent subjectivity in aesthetic perceptions can mean for example that large open expanses are viewed negatively by some and positively by others).
- *Inspirational*: The natural world provides a rich source of inspiration for art, folklore, national symbols, and advertising. Wetlands have been a constant inspiration for the visual arts and literature, while the flow and sounds of moving water have inspired diverse works of music.
- *Sense of place*: Ecosystems in general are a central pillar of "sense of place", a concept often used in relation to those characteristics, in particular the less tangible ones, that make a place special or unique, and those that foster a sense of authentic human attachment, identity, and belonging. Wetlands have a powerful presence and thus contribute strongly to the creation of a sense of place, as in the cases of London and the Thames, the Bay of Tunis, the Venice Lagoon, and a host of individually distinctive and "atmospheric" lakes, rivers, marshlands, and forested wetlands around the world.
- *Cultural heritage*: Many societies place high value on the maintenance of historically important landscapes ("cultural landscapes") and culturally significant species. The diversity of ecosystems is one factor contributing to the diversity of cultures. Wetlands in particular have a strong history of this kind of close interdependence between nature and culture: the broad range of human activities they support have been a key determinant of cultural diversity throughout the ages, and they continue to do so today, as contemporary culture and "living heritage" continues to evolve.
- *Educational*: Ecosystems and their components and processes provide the basis for both formal and informal education in many societies. In addition, ecosystems may influence the types of knowledge systems developed by different cultures. In the case of wetlands, educational possibilities are enhanced by their high diversity and frequent ease of access.

Some specific examples of these different types are described in the paragraphs which follow.

Water and wetlands play a major role in indigenous spiritual beliefs, for example among the tribes inhabiting major river basins such as the Niger and the Amazon. The purifying element of water is solemnized in mainstream religions such as Hinduism, Christianity, and Islam, and is associated with particular wetlands such as the Ganges River in India, the Jordan River in the Middle East, and the Al Hawizeh marshes in southern Iraq. "New Age" movements, mainly in Europe and North America, have found a new source of spirituality in the experience of nature, often with a focus on water and wetlands.

A notable spiritual and social event related to a major wetland is in Spain, where an annual pilgrimage on horseback and carriages takes place through the Doñana

National Park in Andalucía, to the village of El Rocío (Papayannis 2008). With roots both in the Catholic Church and in human subsistence links with the wetland environment, the pilgrimage is dedicated to the Virgin of the Dew (Rocío). The pilgrimage attracts many thousands of people every year and helps in promoting both the cultural values of the Guadalquivir area and the natural values of the National Park.

The Camargue is an important wetland complex associated with the Rhône River in the South of France. During the nineteenth century, a rich cultural tradition developed in the area, when the local community took up ideas which had been initially put together by a small group of intellectuals to celebrate aspects of the area's distinctiveness (Mathevet 2004). These mix local elements with Spanish and Roma influences, and focus on bulls (*toros*) and horses and their herders (*gardians*). Traditional knowledge related to agriculture and stock breeding has been incorporated into these Camargue traditions. It was promoted through the visual arts (Van Gogh was a resident of Arles) and through music, literature, characteristic costumes, and gastronomy. Social events have played a key role, focusing on a special version of acrobatic games with bulls, closely related to similar practices in Minoan Crete more than 3,000 years ago, and in which the bulls are not killed. These Camargue traditions have proven to be a strong attraction for visitors and have accordingly been maintained as part of the value of the area for tourism.

Also in the Camargue is found the Tour du Valat Research Institute for Mediterranean Wetlands. Established in the 1950s by Swiss zoologist Luc Hoffmann in a traditional *mas* (farmstead), which is now a nature reserve, the institute's strong team of researchers has been producing scientific knowledge of great value in the conservation and wise use of wetland ecosystems in the Mediterranean Basin and beyond. The research undertaken at Tour du Valat includes social science studies, embracing traditional knowledge in the area and helping to foster a more holistic approach to wetland management.

In addition, the Tour du Valat Institute plays an important role in education and training, maximizing the dissemination and use of the knowledge it has developed over half a century of work, through publications, seminars, and hosting postgraduate students.

Wetland landscapes have many distinctive qualities, and their ecological diversity and the contrasts it produces are often a part of this. It is predominantly however the element of water which typically marks the special character of most wetlands, in terms of practical life and in terms of the human imagination. It is the water that enlivens the dynamics of these landscapes, often in constant flux from a flat mirror surface to ripples, to waves in a storm. Water gives wetlands their infinite color palette, along with shadows, reflections, and the possibility of profound changes of state (liquid presence, evaporated or seasonal absence, and solidly frozen ice). It can be silent or the source of a variety of sounds. Thus, it is only natural that wetlands should provide a cultural service in the shape of inspiration for various forms of art and creative expression.

Well-known painters inspired by water and wetlands include Brueghel, Monet, Raphael, Renoir, and Seurat. Music composers with similar inspiration include

Bach, Britten, Cage, Debussy, Handel, Ravel, Schubert, Sibelius, and Tchaikovsky. Among writers who have used wetlands as background to their works are Bennett, Blackmore, Christie, Doyle, Dickens, Hodgson Burnett, Shakespeare, Twain, and Wordsworth.

Challenges

As indicated above, wetlands provide significant cultural services that are of considerable benefit to humanity. There are, however, two prerequisites for safeguarding the delivery of these services. First, the cultural contexts within which they are defined must be recognized and themselves safeguarded where necessary. Second, the beneficiaries of such services must perceive the value of them, the role of wetlands in their delivery, and the factors that may jeopardize this delivery.

For example, traditional boats in wetlands embody invaluable knowledge on mobility in water, fishing methods, and use of materials. Maintaining that knowledge may not be feasible today, as these boats have been replaced by more technically developed and less expensive craft. However, one way to do so arises when the old boats find new uses in wetland visitor activities, as in the Neretva Delta in Croatia.

The maintenance of cultural services needs to be embraced where applicable in management and awareness regimes for wetland sites. Adopting explicit management objectives and implementing formal management plans is an expected practice for the most important sites; and interdisciplinary management expertise may be necessary in these processes in order to ensure that cultural aspects are properly addressed alongside attention to the delivery of other types of ecosystem service.

Sensitizing the recipients and beneficiaries of wetland cultural services is required at all levels. The general public – be they local inhabitants or national and international visitors – needs to be better informed about the various aspects of wetlands, including culture, so that they can appreciate the full range of services which they provide. Thus, culture must be incorporated in communications, education and public awareness (CEPA) activities targeting wetlands, as already initiated by the Ramsar Convention and its regional activities (such as the MedWet Initiative).

There is a particular challenge in finding imaginative ways to develop appropriate awareness of the more intangible, indirect and long-term types of services wetlands provide. Techniques for evaluating cultural services, and for monitoring change, need continual refinement. Spatial mapping of service delivery may be particularly important in helping to reveal the intricate, multilayered, and long-range pathways involved. Sociology, ethnography, psychology, ethics, and aesthetics all have a part to play in this, alongside ecological and hydrological science.

Systematic, society-specific work of this kind will in turn support a positive stance by decision-makers, and the adoption of policy changes that will better sustain the capacity of wetlands to continue delivering their wealth of cultural services in future.

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187 Learning for Life and Educational Services

Mark Everard

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Abstract

From the fringes of oceans and major rivers and lakes to the smallest of urban pools, wetlands offer a wide range of educational opportunities. These span multiple forms of learning, ranging from formal educational centers at or relating to wetlands through to informal learning opportunities afforded by proximity to wetlands.

Keywords

Learning · Education · Informal Education · Lifelong Learning · Wetland Learning and Behavior Change · Wetland centres

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Introduction

From the fringes of oceans and major rivers and lakes to the smallest of urban pools, wetlands offer a wide range of educational opportunities. These span multiple forms of learning, ranging from formal educational centers at or relating to wetlands through to informal learning opportunities afforded by proximity to wetlands.

Wetland Education

Many formalized wetland visitor centers have educational facilities, and some field study sites are wetland-based. Two English case studies are noted below exemplifying the kinds of formal educational facilities offered by such centers worldwide.

The London Wetland Centre

The London Wetland Centre is a visitor center operated by the ► Chap. 92, "Wildfowl and Wetlands Trust" (WWT) in south west London (http://www.wwt.org.uk/wetland-centres/london/). The center itself is based on the site of a former water treatment works. Investment from the sale of land to developers enabled the WWT to develop a complex of wetlands and visitor facilities, accessible to a dense population on the periphery of London. The purpose of WWT is to save wetlands and their wildlife, raising awareness of the issues that affect their survival and, through its UK network of visitor centers, enrich people's lives through learning about and being close to nature. Education is a central part of WWF's mission, and the London Wetland Centre offers facilities for both formal and informal education about birds and wetlands. Similar educational opportunities are presented at the Trust's other centers across the UK.

The Field Study Council Study Centre at Slapton Ley

A long-established example of a wetland-based study center is the UK Field Study Council's center at Slapton Ley (http://www.field-studies-council.org/centres/slapton/slaptonley.aspx), on the south Devon coast, adjacent to both a shingle sea coast and the freshwater lagoon from which the center takes its name. Slapton Ley is itself a National Nature Reserve, the ley constituting the largest natural freshwater lake in the South West of England, separated from the sea by a barrier beach and shingle ridge known as Slapton Sands. The surrounding coastline, nearby coastal towns and Dartmoor National Park, provide unrivalled opportunities for field courses, including a variety of freshwater and saline wetland types. The Slapton Ley study center offers outdoor classroom courses tailored to a range of secondary and tertiary education needs. Other courses are offered for teachers, for individuals and families, and for professionals.

Informal Education

The term "informal education" generally relates to that which occurs outside of a standard school setting or beyond structured home-schooling. Informal education can entail informal educators who work in many different kinds of settings, with individuals and groups who choose to engage with them. Some of the learning experiences provided by wetland centers constitute informal learning. Other informal learning happens via mass media routes. More structured environments for informal learning include museums, libraries, zoos, after-school groups, and other community-based organizations and cultural institutions. Age is no limitation of when people can engage in informal learning.

Wetlands offer a variety of informal educational opportunities. These range from scientific observation and investigation, through to learning arising from productive uses of wetlands, recreational activities, spiritual and artistic significance, and indeed learning about the full range of ecosystem services provided by wetlands as well as their dependence on wetland structure and functioning.

Lifelong Learning

Lifelong learning comprises the *ongoing, voluntary, and self-motivated* pursuit of knowledge for either personal or professional reasons (Department of Education and Science 2000). It thereby enhances social inclusion, active citizenship, and personal development, but also self-sustainability, rather than competitiveness, and also enhances employability (Commission of the European Communities 2006). Lifelong learning recognizes that learning is not confined to childhood or the classroom, but takes place throughout life and in a range of situations. Scientific and technological innovation and change has had a profound effect on learning needs and styles and also the media through which people can educate themselves. As such, learning can no longer be regarded as belonging solely in formalized places and at inflexible times to acquire knowledge, for example, in a school, or to apply it, such as in the workplace (Fischer 2000). Rather, learning is reframed as occurring through ongoing interactions with others and with the world around us.

"Learning for life," though also the title of a more formalized United States school and work-site-based youth program (http://resources.learningforlife.org/) subsidiary to the Boy Scouts of America, is also a more generic term synonymous with "lifelong learning."

As for informal education, wetlands offer diverse and rich resources for lifelong learning across a range of linked topics and in the systemic interactions between them. Indeed, reading this Wetland Encyclopaedia and exploring its cross-links can itself constitute a form of self-guided lifelong learning.

Wetland Learning and Behavior Change

Public education is increasingly being added to regulatory "toolkits" as a means to promote behavior change, as an alternative to enforcement (legal obligations) and inducements (subsidies). For example, the US Environmental Protection Agency has a program of education about water including wetlands education (US EPA n.d.) as part of an outreach and communication program. This is tailored to be accessed in structured learning, informal learning, and lifelong learning situations, with the aim of greater appreciation of wetlands and natural processes and hence influencing more sustainable behaviors.

Conclusions

The educational opportunities provided by wetlands are as diverse as the types and locations of wetlands themselves, ranging from formalized educational institutions and courses through to informal access and self-guided learning opportunities. There are multiple opportunities for learning spanning the full spectrum of ecosystem services that wetlands provide, as well as interactions between these services and the processes producing them.

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Educational Benefits of Wetlands

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Sandra Hails

Contents

Abstract

Wetlands are highly diverse ecosystems, ranging from lakes, rivers, and marshes to coral reefs, salt marshes, mangroves, and more. In addition to their natural beauty and wealth of biodiversity, they are also recognised for the services they provide such as food, timber, water purification, and as places for recreation and experiential learning. Wetland visitor/education centres and the wetland environment itself can be used to great effect for learning about wildlife values and sustainable living. Well-designed activities in wetland centers and within wetlands can promote sustainable lifestyles and wetland conservation. Some examples include citizen science projects, the experiential learning in wetlands for tourists as well as local schools, parents and teachers; wetland management training in wetlands that includes working with wetland users such as fishermen, farmers and local communities.

Keywords

Wetland recreation · Wetland education · Teacher training · Experiential leaning · Citizen science

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Introduction

Wetlands are highly diverse ecosystems, ranging from lakes, rivers, and marshes to coral reefs, salt marshes, mangroves, and more. A sense of the outstanding beauty and wealth of biodiversity of wetlands is evident simply by listing a few wetlands such as the Great Barrier Reef, the Okavango Delta, Victoria Falls, or the Everglades. From a more analytical perspective, the high *value* of wetland ecosystems has been calculated through assessing the diverse services they provide for people – food, timber, water purification, water storage, a place for recreation, education, and so on – confirming that wetlands are vitally important across the world for human survival and of significant economic value (Russi et al. 2013) rivalling or exceeding that of other ecosystems.

The importance of wetland education is recognized in the Ramsar Convention's adopted Programme on Communication, Education, Participation and Awareness (CEPA) (Ramsar Convention 2008). The Programme requires countries to integrate CEPA principles and practices into managing wetlands, recognizing that CEPA is an essential part of sustaining wetland health and that engaging all people in conservation is central to success.

Under the CEPA Programme, education has been defined as "*. . .a process that can inform, motivate, and empower people to support wetland conservation, not only by fostering changes in the way that individuals, institutions, business and governments operate, but also by inducing lifestyle changes*". It may take place in both formal and informal settings and, in its broadest sense, is a life-long process.

Formal and Informal Education

Wetland education is helpfully divided into formal and informal education. The former is typically associated with programs within schools, colleges, and universities. The latter is associated more with wetland visitor centers but also with a broad range of noncenter-based educational opportunities often associated with a broad range of activities in wetlands, including:

(i) Wetland-focused recreational activities such as bird-watching, fishing, whale-watching, canoeing, nature-walking, diving and snorkelling, etc.;
(ii) Wetland data-gathering through citizen-science wetland projects that bring the general public and wetland scientists together in conducting basic research
(iii) Working with local communities living in or near to wetlands including specialist users such as fishermen, small-scale farmers
(iv) Working with a broad range of other specialist groups whose activities can have an impact, both positive and negative, on wetlands, such as journalists, national and local government planners, NGOs, businesses, etc.

Formal Education

There are many universities and colleges across the world delivering courses about wetlands, including how they function, their values, and their management. Other training institutions deliver more practical courses on wetland management and management planning. Some of these courses include site-based visits and practical work to reinforce the theoretical work. This might include monitoring physical and chemical aspects of wetlands, assessing plant and animal communities, or working with local stakeholders.

Informal Education at Wetlands

Wetland visitor/education centers located at wetlands use their wetlands to inform people about the flora and fauna of wetlands, about the wetland ecosystem services they deliver to people, and about impacts of people on wetlands covering topical issues such as fresh water availability, pollution, climate change, and so on. Such centers provide a variety of exhibitions, audio-visual presentations, hands-on experiential exhibits, themed restaurant menus related to wetlands, and many other imaginative and innovative displays and materials designed for visitors with diverse interests, knowledge, and needs. More information on wetland centers and educational benefits are discussed in a separate chapter addressing Wetland Visitor Centres.

Apart from wetland centers situated at wetlands, wetlands themselves can directly provide a learning environment for all sectors of society. Various examples below demonstrate the diversity of target groups and educational experiences that wetlands can provide directly through managed site visits.

Sydney Olympic Park, an urban park in Australia, covers 650 hectares of parklands including woodlands, open areas, and a variety of wetlands such as saltmarshes, mangroves, swamp forests, and waterways (Paul 2013). The Park authorities run over 30 excursion courses in the park for over 30,000 students providing an educational experience that feeds into the national curriculum in geography, science, and mathematics. In addition, the Park provides state-recognized professional development programs at the wetlands for teachers from these schools to assist them in updating and further developing their professional skills.

The International Training of Trainers course on Integrated Water Resources Management (run by Wageningen University in The Netherlands) brings together a range of professionals working in the wetland, water, and sanitation sectors, including government managers and planners and relevant NGOs, offering a perfect opportunity for cross-sectoral sharing of experiences and learning in management planning (Ramsar Convention 2013). Organized every two years by the Wageningen UR Centre for Development Innovation in The Netherlands, the three-week course is endorsed by the Ramsar Secretariat. While two of the three weeks take place in a more formal classroom setting with lectures and hands-on exercises, one week is on-site at a wetland. The fieldwork involves visiting and interviewing a broad range of stakeholders from the wetland area, including local government staff from the water board and state forestry service, farmers, fishermen, hoteliers, and local NGOs. The structured questions collect essential information for the further development of a management plan for the wetland area, and the outcomes are presented to all stakeholders for further collective discussion.

In the Philippines, an innovative project was targeted at final year architecture students, challenging them to design a wetland center at the Candaba Wetlands for local communities and other visitors, reflecting the role the center could play in improving the understanding of the value of wetlands (Ramsar Convention 2012). The project was also designed to reflect sustainable design principles in the building itself. An important component of the project was a visit to the wetland to understand more about the location, the wetland itself, and its local communities, thus setting the context for the design. The winning design was used in the final development of the center, and the overall project provided an experiential understanding of the wetland environment. In addition, this project had an impact on the architectural program at the urban university and was widely reported by the press, taking the wetland message even further.

In Finland, following the opening of a new wetland visitor center at the Liminganlahti Bay Ramsar Site, a day of activity was held for 100 local schoolchildren from two schools, their parents, local community members, local tourism enterprises, and journalists (Ramsar Convention 2012). The children had collected wicker materials in advance and then spent time creating wicker bird sculptures with their parents at the site. A workshop organized by university students helped explain more about the bay area ecology, a local environmental association guided people at the viewing tower and walkways, members of the village community provided drinks and snacks, and the press reported on the event. This whole-community action led by the local schoolchildren and teachers provided an ideal learning opportunity for the whole community about the bay area where they lived, bringing them into the bay for a day of activity. The long-term goal of the activity was to encourage more visits to the wetland and center by the local community and to promote future use of the wetland by the schools as a source of art, knowledge, and recreation.

Wetland symposia are often considered as meetings for wetland scientists to talk to other wetland scientists. However, New Zealand's national biennial wetland symposia have found a practical formula to broaden the reach by providing a mix of presentations, technical sessions, workshops, and field trips to local wetlands (Ramsar Convention 2012). The symposium typically brings together scientists, wetland managers, NGOs, community groups, iwi (Maori groups), landowners and businesses, and the mix of presentations in a formal setting as well as wetland field visits ensures the sharing of varied experiences, interests, and knowledge. The site visits with their informal settings and practical management demonstrations help place the knowledge from the presentations into perspective, providing a perfect setting for discussions about the constraints and the realities of wetland management faced by people on the ground.

Citizen science projects in wetlands provide tremendous educational opportunities and potential benefits through the participation of citizens in many different nonexpert ways in assisting scientists in monitoring the environment (Science Communication Unit 2013). This can be very diverse, with citizens gathering data such as water quality, garden bird counts, butterfly counts, beach watches. These are usually led by scientists or NGOs who know the data required and have developed simple techniques for accurate data gathering by nonscientists. This not only provides large amounts of valuable data sets for the scientist that are used as indicators of environmental change, but it also helps to reconnect people with nature and improves their knowledge and understanding of the environment, encouraging lifelong support for conservation. The benefits can also extend to the scientists themselves: research has shown that citizen science projects can encourage scientists to improve their communication techniques as they have to communicate the results to keep citizen scientists engaged so must make their science understandable at the layman's level. For wetlands, amateur bird-watchers are almost certainly the most frequent contributors to citizen science projects, providing an invaluable source of data on the state of waterbirds globally. A project in the Caribbean in 2014 recorded counts from 281 citizen scientists in 13 countries, recording data on 93 species of waterbirds and providing a valuable dataset on 2014's wintering waterbird populations in the Caribbean (eBird Caribbean n.d.).

A further source of educational opportunities comes through wetland tourism. A publication from the Ramsar Convention on wetland tourism (Ramsar Convention and UNWTO 2012a, 2012b) revealed the broad diversity of tourist activities in wetlands. Tourists may visit a wetland and have an enjoyable experience, but leave without really having learned much about wetlands, their wildlife, utility, and fragility: much depends on the materials and information available at the site on the wetland itself and on broader environmental issues. The case studies for the publication – all from Ramsar Sites – reported an amazing diversity of activities including whale-watching, scuba diving and snorkelling, boat cruises, bog-walking, bird-watching, fishing, canoeing, horse-riding, and many more. At many sites, the role of trained guides extended beyond controlling numbers of tourists and restricting where they could go, to sharing information on the conservation of the area, its plant, and animal species, and even wetland-related information concerning sustainable lifestyles. Most tourist sites also include interpretive trails and signage to educate visitors about the wetland and its wildlife. Both the guides and the interpretative materials can also provide information and advice about sustainable lifestyles and promote individual behaviors to support conservation in its broadest sense.

Conclusion and Future Challenges

This chapter provides some examples of how wetland ecosystems are being used for formal and informal education about wetlands, looking variously at the diversity of target audiences that can be brought to wetlands to learn more about the value of wetlands and the ecosystem services they provide to sustain not just wetland biodiversity but also human lives and livelihoods. Wetlands and their natural beauty and utility can do much to sustain the vital connection between people and nature, a connection that is diminishing as more human populations in all continents settle in urban areas, and thus become less connected physically and mentally to the natural environment. Sustaining this connection with nature requires continued innovative approaches by wetland centers and those working for wetlands at local and national levels to increase the support for wetland conservation.

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Wetland Visitor and Education Centers 189

Chris Rostron

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Abstract

Wetland Visitor Centres, or Wetland Education Centres, can be defined as activities or facilities at or near a wetland which deliver CEPA (Communication, Education, Participation and Awareness Raising) activities to the public. The center may be a large, complex set of buildings that delivers a range of activities, such as the Hong Kong Wetland Park with over half a million visitors per year or, at the other end of the scale, a small community group which supports regular volunteering activities, as examples.

Introduction

Wetland Visitor Centres, or Wetland Education Centres, can be defined as or regular activities at or near a wetland which deliver CEPA (Communication, Education, Participation and Awareness Raising) activities to the public. The center may be a large, complex set of buildings that delivers a range of activities, such as the Hong Kong Wetland Park with over half a million visitors per year or, at the other end of

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the scale, a small community group which supports regular volunteering activities, as examples.

Wetland centers first emerged during the early twentieth century, for example, with WWT Slimbridge in southern England welcoming its first visitors in 1946. Sir Peter Scott, a pioneer of wetland education, championed this approach, and environmental education centers also appeared in the Americas during the same time, for example, through the activities of the National Audubon Society. Now, wetland centers are increasingly popular across the globe, reflecting the wealth of wildlife found in wetlands, and the increasing interest of the general public. In particular, the opportunity to get "hands-on" with and close to wetlands and their wildlife is a key part of the attraction of these centers.

The importance of Wetland Visitor Centres is in their capacity to provide a structured experience to those visiting wetlands, either through "self-guided" panels and written information, or with trained staff that are able to run targeted public sessions. Often, people visiting wetlands for the first time will be unaware of the importance and significance of the habitat and its species. Now that 80% of the Western population lives in urban areas and has limited access to nature, and many people are not confident in visiting or spending time in "wild" areas; having a wetland center with staff and facilities (such as toilets and cafés) will attract a much wider audience than a wetland on its own might.

Having said that, it is of course essential that the wetland center does not impact negatively on the wetland itself. In some cases, it is not appropriate to have a large, built facility with thousands of visitors, if the wetland or the species it supports are fragile or heavily protected. The long-term goal of wetland conservation should always be kept in mind, and whilst this includes raising awareness of their importance, it should not be at the expense of wetlands or their wildlife.

CEPA and Wetland Visitor Centers

The signing of the Ramsar Convention on Wetlands was an important milestone in the global protection of wetlands, signed in 1971 and with 168 signatory countries and over 2,000 designated Ramsar sites (as of July 2014). Part of the Ramsar Convention's strength is in its approach to CEPA (Communication, Education, Participation and Awareness Raising), recognizing the importance of people's support in the conservation of wetlands and their species. Some of the delivery of Ramsar's CEPA program is through wetland centers, and this in turn is reflected in an agreement with WWT (the Wildfowl and Wetland Trust) to support the Wetland Link International (WLI) program. WLI is a support network of around 300 key centers across the globe that work together to share information, develop tools and resources, and improve delivery of onsite CEPA activities at wetland centers.

Key to the ethos of wetland centers are Sir Peter Scott's ideas around Conservation, Education, and Recreation and getting the right balance between these three activities. Clearly, this will differ according to the setting of the center, the facilities at its disposal, and the cultural issues depending on where in the world the center is located. Whatever the context of the Wetland Visitor Centre is, it is important to ensure that the areas of conservation and education are not outweighed by pure recreation, particularly at sensitive or vulnerable sites. Where artificial wetlands have been created for the express purpose of recreation, there are still many opportunities for education and conservation activities which should not be overlooked, and these are essential in the mix of what any Wetland Visitor Centre does.

Wetland Visitor Centre Activities

What types of activities take place at a Wetland Visitor Centre? These could be broadly categorized as comfort (café, toilets, car parks), conservation (species and habitat monitoring and management), and communication (education, awareness raising, and messaging). Of course this is as vast simplification of what actually goes on, so let's break it down a bit further.

The first things that a visitor is likely to want to know are largely practical: for example, how to get to the center, where they can park and whether there are toilets, a café, and a shop at the site. These are important to attract the broadest range of visitors possible and to ensure that they have a good experience when they are on site (and therefore want to come back). It is also important to ensure "access for all," welcoming people that have disabilities, visual or hearing impairments, or those that have access issues due to age, illness, or a range of other reasons. Making the site easy to access helps everyone.

Conservation activities are an essential component and often take place out of the sight of visitors. However, even when these activities are not visible, it is useful to use staff and interpretation to communicate them (see below). Conservation can include monitoring and survey work, habitat management and creation, captive breeding and reintroduction programs, among a range of other activities. Sometimes visitors and conservation are not compatible, due to the disturbance that visitors can create. However, using creative means of shielding visitors from birds or other wildlife, good visual access can still be maintained. Well-designed bird hides, sunken walkways, use of optics like telescopes and binoculars can all give visitors a thrilling experience, while ensuring that the wildlife they have come to see remains relatively undisturbed.

CEPA activities cover an extremely wide range of activities, and they need to be diverse to get messages across to the broad range of visitors that a Wetland Visitor Centre will attract. Whether the visitor is 9 or 99, is an expert wetland birder or a city dweller with no wildlife experience, or is more interested in the shops than the bird hides, there should always be opportunities to communicate wetland messages. Trained staff that are used to dealing with visitors and running activities are essential – they need a set of skills that is specialist, and the Centre should invest in regular training and must recruit the right people.

Interpretation can be defined as "methods to communicate sometimes complex messages to the public" and can include signs and information boards, interactive exhibits, structured activities run by staff, leaflets and other printed materials, audio-visual displays, and a range of other types, both inside and out. Interpretation should reflect seasonal differences, and although there is likely to be a set of permanent interpretation materials reflecting the key attributes of the center, there is also a need to regularly change interpretation and activities. Many visitors are likely to come several times a year, and it is very important to provide new attractions so that they will continue to find new and interesting things to justify their visits.

Staff and volunteers are possibly the most important element of a visitor center. Staff need to be accomplished at welcoming and inspiring visitors and communicating conservation messages, as well as managing the site. At Oak Hammock Marsh, Canada, their mantra is "recruit for attitude, train for the rest," indicating that outgoing people with good social skills are vital in these roles. At smaller centers, staff and volunteers may play a number of these roles, and it is important that these people are well equipped to answer questions and ensure the safety of visitors and particularly to have a good knowledge of the conservation value of the site. Volunteers can be an essential part of delivering Wetland Visitor Centre activities, from administrative support, to conservation work and visitor management. However, volunteers, like staff, need support and training; without this, they can feel unsupported, out of their depth and undervalued. Investing in volunteers will pay dividends, but if done badly, can have a negative effect on visitors and the site.

Wetland Visitor Centre Audiences

Visitors come to wetland centers for a range of reasons. Some will already be interested in wildlife and be attracted to see specific species and habitats. Some will stumble across the center as a tourist in the area and be primarily interested in the "fun" element of the center, often looking for something to entertain their children. Others will be locals, use the site to meet friends, use the café, or spend time with their children.

Whatever the reason, the primary purpose of the center is still about communicating messages about wetlands. This is what makes them different from other attractions such as theme parks, fun-fairs, or even shopping centers. It is therefore important to identify the different types of visitors that are coming. Recent work by WWT has looked at the range of visitors and identified them by their interests and activities. These can loosely be divided into:

- **Families** with children may be visiting with the motivation to entertain their children, or to educate them. In many cases, it will be a bit of both. In terms of their characteristics, they will want good access to toilets and a café, look for fun and stimulating things to do, and also seek activities that can involve adults and children.
- **Schools and formal education**. These groups are often coming as part of a wider structured program of education. They will look for activities that are relevant to their syllabus, knowledgeable staff that can work with students, and often hands-on activities that they cannot do in school or college.

- **Birders/Naturalists**. In many countries across the world, bird watching and visiting natural areas are becoming an increasingly popular leisure activity. The role that nature plays for many of us is central and particularly important for those that live in urban areas and have limited access to green space and wildlife. Wildlife enthusiasts can range from specialists that can tell many species apart and recognize the importance of wetlands already, across a broad spectrum to those with a very basic knowledge of wildlife who want to learn more. These two groups have different needs and need to be treated differently, offering information at both a very detailed and also an accessible level.
- **Social visitors**. People often like to spend with friends, family, and workmates and will use a visit to a wetland center as part of their regular social activities. They are not necessarily interested in wildlife; the important element is spending time with other people. For them, the café, shop, and unusual events are important, whereas spending hours trying to spot a rare bird is not likely to attract them. Providing them with interesting things to visit and talk about is important and will often lead to them developing a deeper interest in wetlands and wildlife.
- **Sensualists**. Many of us, particularly those living in cities, will relish the opportunity to get away from the daily urban routine to escape into a more natural environment. Being surrounded by water, trees, dragonflies, and birds is a great tonic to balance out the stresses of daily life. For these visitors, peace and quiet and being surrounded by nature is the most important element of their experience.

Future Challenges

With the increasing development of visitor centers across the globe, their success is largely reliant on having developers that can plan and design a center that works well and access financial resources that will sustain the center's activities. While a wealth of information is available (such as the *Handbook on Best Practices for the Design and Operation of Wetland Education Centres*, available at www.wli.org.uk/resources getting it to the right people can be difficult. Those working for local governments, NGOs, or even private sector are unlikely to find out about it unless it is well signposted on the Internet, or publicized through networks such as WLI and the Ramsar distribution lists.

Finding staff to run the center will also form a major challenge. The right mix of skills is difficult to recreate, with many likely candidates either trained as educators or as naturalists, with very few having a mix of both. Some efforts have been made to run distance learning courses (such as that London's South Bank University), but funding shortages have resulted in no long-term course being sustained. Training materials and courses, particularly for countries creating new centers (such as currently in China), would be useful, particularly where training in environmental education is not widespread.

Pressure on wetlands themselves is also threatening the existence of wetland centers – indeed, the poor siting or design of some centers can threaten the very resources that they were built to promote. Pressures such as climate change, human development, agriculture, invasive species, pollution, or over-use of water are all putting intense pressure on wetlands. Degraded and disappearing wetlands will cease to hold an attraction for visitors, and this puts even more pressure on those that remain.

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Education Centers in Australia and New Zealand

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Abstract

The term "wetland center" encompasses a range of facilities, ranging from small centers to larger centers with a range of facilities, possibly including a dedicated visitor center, classroom facilities, and curriculum-based education services for young children to postgraduate students. They may be owned or operated by a variety of public or private organizations, including charitable trusts, governments, and environmental groups. In many cases they support formal curricula but not do necessarily make use of the wise use material provided by the Ramsar Convention. The following practical topics may serve as a basis for educational or training courses and could guide the development of suitable teaching materials for specific courses and practical demonstrations: wetland conservation; wetland restoration and creation; understanding the value of wetlands; stakeholder engagement; and integrated planning.

Keywords

Education centre · Visitor centre · Education · Training

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Introduction

The term "wetland center" encompasses a range of facilities, ranging from small centers, possibly with paths and interpretive signs but no formal visitor center, to larger centers with a range of facilities, possibly including a dedicated visitor center, classroom facilities, and curriculum-based education services for young children to postgraduate students. They may be owned or operated by a variety of public or private organizations, including charitable trusts, governments, and environmental groups.

A survey of wetland education centers in Australia and New Zealand has been reported in Finlayson et al. (2013) based on information collected in 2010 and discussed with the Secretariat of the Ramsar Convention on Wetlands and with Wetlands Link International.

Survey of Wetland Centers

The results from the survey of 21 wetland education centers in Australia and New Zealand, largely in urban settings, are summarized in Table 1 and discussed in the text below. They provide an overview of the features of the centers and the educational and training work that they undertook.

The majority of the centers included in the survey were owned and managed by local government authorities (38%) and state parks or wildlife services (24%), indicating that local governments and state conservation services had taken the lead in establishing and supporting wetland centers. Many of the centers were able to cater for visitors, with some encompassing basic (38%) or comprehensive (33%) on-site facilities, although some (19%) did not have a visitor center. The size of the buildings, including facilities for visitors, ranged from <100 m^2 (about 25%) to

Table 1 Summary of the main features of the wetland centers (From Finlayson et al. 2013)

Feature	Most common outcome
Ownership/ management	Run by local government (38%) Run by State parks or wildlife service (24%)
Type of facilities	Some visitor facilities have basic (33%) or comprehensive (38%) on-site facilities, including 43% with buildings >300m^2
Annual visitation	45% >30,000 visitors, comprised of local visitors (54%), individuals (49%), and school groups (39%)
Staffing	1–3 paid staff (57%); no volunteers (29%), more than 100 volunteers (14%)
Annual operating costs	<AUD $50,000 (14%) and >AUD $350,000 (19%)
Teaching styles	Formal classroom (81%), outdoor (76%)
Teaching aids	Static displays (67%), audio-visual (62%), microscopes or other equipment (57%), and computers (29%)

more than 300 m^2 (43%). The facilities generally included an interpretive center (67%), indoor and outdoor classrooms (48% and 38%, respectively), as well as multiuse rooms (48%) or a lecture theatre (29%), with very few (5%) having none of these.

Visitor numbers varied, with almost half (45%) receiving more than 30,000 visitors per year and approximately a third (35%) receiving 15,000–30,000. More than half of the visitors came from the local region with visits by individuals (49%) and school groups (39%) dominating.

The number of full-time staff at the centers was generally small with many (57%) having three or fewer full-time equivalent paid staff. Volunteer staff were engaged by many centers with some (14%) having more than 100, although many more (29%) did not have any. The differences in facilities and staffing levels are also shown by the annual operating budgets with some (19%) spending more than AUD$350,000 (Australian dollars) with a similar number (14%) spending less than AUD$50,000.

Despite these differences, a large proportion of the centers provided curriculum-based education programs, with most conducting both outdoor (81%) and classroom (76%) activities with hands-on (48%) and laboratory-based (43%) lessons. These lessons made use of a number of teaching aids, including static displays (67%), audio-visual presentations (62%), microscopes (57%), and computers (29%).

Education and Training Activities

While the above information provides a summary of the main features of the wetland centers, including staffing and operating funds, it does not describe the education and training activities or how these connected with the formal state education curricula. Nor is there an indication as whether the copious amount of information and educational materials developed by the Ramsar Convention have been used as resource materials in education, training, and awareness programs. Most centers relied on their own staff or volunteers to conduct the education and training activities, with more expert instruction being provided by invited lecturers or researchers and learned members of the local community.

Given the development of an explicit focus on urban wetlands through the Ramsar Convention and other international organizations, there is an opportunity to enhance the education and training programs with more explicit information about urban wetlands. Specifically, the following practical topics may serve as a basis for educational or training courses and could guide the development of suitable teaching materials for specific courses and practical demonstrations:

1. Wetland conservation
2. Wetland restoration and creation
3. Understanding the value of wetlands
4. Stakeholder engagement
5. Integrated planning

Many of these subjects are already included in courses run through established wetland training centers, such as the WET workshops run by the Sydney Olympic Park (Finlayson et al. 2013), and through more informal processes (see, for example, Finlayson 2000), and in support of capacity development efforts for local communities, such as those that have for many years been promoted through the Ramsar Convention (Finlayson et al. 2001). It is very likely that these topics have been or could be included in training courses or exercises undertaken by other wetland training programs.

The potential for increased exchange of educational and teaching materials and experience has been encouraged through informal local networks, including links with government education departments, as well as through the international network facilitated by Wetlands Link International (http://wli.wwt.org.uk).

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191 Traditional Knowledge and Wetlands

Mark Everard

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Abstract

People have coexisted with wetlands since prehistory, making use of their multiple beneficial services and averting unintended consequences. It is therefore unsurprising that a depth of traditional knowledge has been deduced, shared, and subsequently evolved. Modern pressures on wetlands have culminated in international collaboration and knowledge-sharing to avert resource degradation, most evidently under the Ramsar Convention, seeking the wise and sustainable use of wetlands. However, many such sustainable management practices are encoded in traditional knowledge and evolved outside of the contemporary appropriative, industrialized world view. It is with good reason that Principle 11 of the Ecosystem Approach recognizes that "The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices," and that Principle 12 states that "The ecosystem approach should involve all relevant sectors of society and scientific disciplines". Not only is inclusion of technical, traditional, and other forms of knowledge valuable for informing more durable management strategies, it is also essential to support more equitable and economically efficient decisions that take account of the interests of all in society.

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Keywords
Traditional knowledge · Ecosystem approach · Indigenous · Equitable · Paddy · Common pool resource

Introduction

People have coexisted with wetlands since prehistory, making use of their multiple beneficial services and averting unintended consequences. It is therefore unsurprising that a depth of traditional knowledge has been deduced, shared, and subsequently evolved. Modern pressures on wetlands have culminated in international collaboration and knowledge-sharing to avert resource degradation, most evidently under the Ramsar Convention, seeking the wise and sustainable use of wetlands. However, many such sustainable management practices are encoded in traditional knowledge and evolved outside of the contemporary appropriative, industrialized world view. It is with good reason that Principle 11 of the Ecosystem Approach recognizes that "The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices," and that Principle 12 states that "The ecosystem approach should involve all relevant sectors of society and scientific disciplines" (Convention on Biological Diversity undated). Not only is inclusion of technical, traditional, and other forms of knowledge valuable for informing more durable management strategies, it is also essential to support more equitable and economically efficient decisions that take account of the interests of all in society.

Traditional Wetland Stewardship

Community-based groundwater use and recharge in semiarid regions of the world provides examples of how people have collaborated and innovated to sustain their livelihoods where water is a principal limiting resource. Pandey et al. (2003) consider the innovation of strategies such as traditional Indian water harvesting structures, which detain water during monsoon rainfall enabling it to percolate into the ground and recharge aquifers for year-round storage and access, as representing societal and cultural responses to prolonged local aridity, matching palaeoclimatological evidence for climate change during the Holocene with archaeological and historical records.

► Chap. 192, "Traditional Knowledge Applied to the Management of Small Tank Wetland Systems in Sri Lanka" reviews small tank (recharge and reservoir)-based food production, dating back to 500 BC and still forming a vital component of both the natural and modified landscape in the dry zone of Sri Lanka. Given the central importance of tank systems for sustaining livelihoods, they also feature in ritual and ceremony throughout Sri Lanka. Their community-based design and management consciously supports multiple connected functions and human benefits ranging from irrigating rice paddy, to supporting a subsistence fishery, to providing water and fodder for livestock and wildlife, as a domestic water supply, for flood control, to promoting groundwater and subsurface water recharge in the surrounding landscape that supports forests, ayurvedic medicines, home gardens, and drinking water wells (Vidanage et al. 2005).

Rice paddy (or padi), addressed in far greater detail in ▶ Chap. 142, "Rice Paddies" have endured throughout centuries and across broad swathes of the tropical and subtropical world, sustaining successive civilizations. Rice paddy systems commonly depend on community-based management, supporting not merely subsistence needs but also potentially cash crop production, linked production of other food including fish reared in inundated paddies during the rice-growing season, straw production, water and soil retention, and (in recent times) ecotourism. Given their central role in sustaining societal needs, rice cultivation constituting the principal activity and source of income for about 100 million households in Asia and Africa (Umadevi et al. 2012), rice paddy management has also commonly become an organizing principle in cultures and livelihoods, with rice and traditional rice-growing systems also incurring notable spiritual significance stemming for the bonding of people around common stewardship of natural productive capacities. The role of traditional knowledge in management of these wetland systems is significant, particularly in their efficient adaptation and management to local climatic, topographical, soil, cultural, religious, and socioeconomic contexts.

The loss of traditional knowledge in wetland and water system management can result in severe problems, including a widespread cycle of interlinked ecological and socioeconomic decline. This is particularly significant for water systems in semiarid and arid environments where locally adapted infrastructure depends on local and traditional knowledge, and on participatory decision-making and management, to sustain wetland, water, and other ecosystems upon which livelihoods depend. Loss of locally adapted traditional knowledge and consequently of both social and technical infrastructure contributes to unsustainable livelihoods in some semiarid regions, culminating in ecosystem degradation and population dislocation, cultural separation, habitation abandonment, and societal collapse (Pandey et al. 2003).

However, groundwater and other ecosystem resources can be responsive to positive management of wetland systems, potentially safeguarding or restoring a range of benefits to dependent communities. For example, Wani et al. (2009) report significant groundwater rises in "treated areas" where community-based participatory methods have been developed at benchmark sites in India, Thailand, Vietnam, and China. These initiatives, which bring together institutions from scientific, nongovernment, government, and farming sectors, have been found to improve productivity by up to 250%, as well as to restore groundwater levels and to reverse the degradation of natural resources (Wani and Ramakrishna 2005; Wani et al. 2006), and community empowerment has also been found substantially to improve the livelihoods of poor people in 368 experimental watersheds across Asia. Nevertheless, local participation in exploitation of regenerated natural resources is also critical, and for this purpose traditional knowledge about locally appropriate governance arrangements is also essential.

Ostrom (1990, 1997) identifies generic principles observed in successful examples of community management of common pool resources (CPRs), including wetland systems. CPRs comprise natural resources that are physically structured in such a way that it is difficult to keep alternative potential users from enjoying or abusing them. Wetland and water systems commonly comprise such “nonexcludable” CPR resource. Attributes identified by Ostrom that increase the likelihood that productive governance arrangements will be formed and will perform better than either markets or states include: clear bounding of common resources, congruence between appropriation and provision rules, collective-choice arrangements, effective monitoring, graduated sanctions, conflict-resolution mechanisms, minimal recognition by the state of rights to organize, and nested layers of management arrangements. The inclusion of traditional knowledge in management arrangements is thus critical to the success of common pool resource management and can therefore be essential to sustainable wetland use and management.

Future Challenges

Understanding the common pool resource nature of wetlands, it is then essential to integrate traditional knowledge into management decision-making institutions and processes. This may entail empowerment and inclusion of a diversity of local people into decision-making institutions as well as, in many cases, rediscovery of preexisting durable management arrangements that can be integrated into contemporary uses and management frameworks. Traditional knowledge held by formerly marginalized communities living most directly resource-dependent lifestyles may be of particular importance in informing sustainable decisions.

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Traditional Knowledge Applied to the Management of Small Tank Wetland Systems in Sri Lanka

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Jayne Curnow and Sanjiv De Silva

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Abstract

The foundation of rice production in the dry zone of Sri Lanka is a hydraulic civilization spanning at least 2,000 years, and based on constructed small irrigation tanks. Dotted across much of the dry zone, and often constituting cascades, the traditional management of these tanks for dry season irrigation water brought together sophisticated engineering skills, deep ecological knowledge and social organisation around the practical need for cooperation and spiritual belief systems. While these small tanks are often referred to with respect to their centrality to irrigating rice, the staple crop, the use of both natural and built elements in managing the tanks and their surrounding landscapes in fact constitute a remarkable multi-functional system that has provided a range of ecosystem services for human well-being. Despite its ancient roots, the ecological principles inherent in the traditional knowledge shaping this system resonate closely with modern concepts around natural resource management such as wise use, sustainability, social ecological systems and green infrastructure.

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Keywords
Sri Lanka · Small-tanks · Constructed wetlands · Water management · Irrigation · Traditional knowledge · Culture · Multi-functionality · Wise use · Social ecological systems · Green infrastructure

Introduction

Small tank (reservoir) based food production is a vital component of both the natural and modified landscape in the dry zone of Sri Lanka. Tank-based irrigation dates back to 500 BC and has been a lynch-pin in the organization of village life and livelihood strategies hence the characterization of Sri Lankan society as a "hydraulic civilization" (Wittfogel 1957; Leach 1959). Human settlements and livelihoods consequently were, and continue to be, orientated to tanks which is reflected in ritual and ceremony. According to the Ramsar Convention, the central role of water in this system defines it as a wetland. In addition to the tanks themselves, rice fields are characterized by the presence of a standing water body. Both offer temporary and seasonal aquatic habitats, across as much as 12% of Sri Lanka's land area (Kotagama and Bambaradeniya 2006).

Small-tanks have an irrigated command area of 80 ha (32.3 acres) or less in a dynamic multifunctional system (Fig. 1). There are four distinctive zones: (i) the tank and tank bund; (ii) associated irrigation channels and paddy fields; (iii) protected forest in the catchment and rain-fed uplands; and (iv) the higher elevation hamlet living area. The tank itself has multiple uses and functions including irrigating rice paddy, supporting a subsistence fishery, providing water and fodder for livestock, as a domestic water supply, for flood control, as well as promoting groundwater and subsurface water recharge in the surrounding landscape that supports home gardens and drinking water wells (Vidanage et al. 2005). Rainfed upland plots under shifting cultivation provided elasticity to the system, enabling households to expand or contract their operational landholding in response to intrahousehold demographic pressures. Moreover, shifting cultivation provides additional means for earning cash along with rice, the dietary staple. Community-managed land use systems facilitate resource access and use, which in turn makes the system viable and enabled its continuity.

Management

As the focal resource that flowed through the system, the main objective of water management was to optimize the conditions necessary for the proper function of the overall ecosystem. Thus the sluices of the tanks are constructed to regulate paddy irrigation and ensure adequate supplies for domestic purposes, livestock, wild animals, and the environment. Other features of the system such as the catchment forest and ponds enable water regulation and conservation in the soil, gradual release

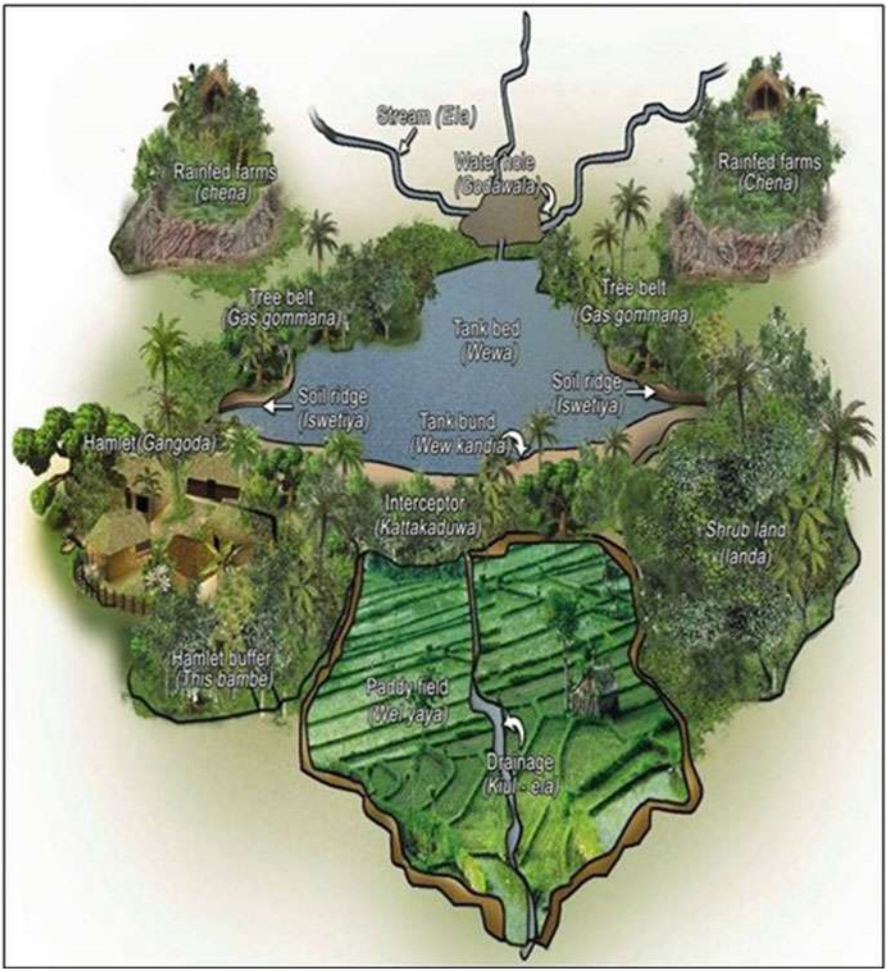

Fig. 1 Schematic of a typical small tank system (Dharmasena 2010)

of water to the main tank during the dry season, water purification, and maintenance of the groundwater table. This is critical given the soil moisture deficit during the annual dry season, the effects of desiccating winds, recurrent droughts, and the low water retention capacity of the soils prevalent in much of the dry zone (Geekiyanage and Pushpakumara 2013).

What is especially notable about many of the components of the system is its multifunctionality. While critical to maintaining the water regime, the catchment forest also provides other services to the farming communities such as timber and medicinal herbs for ayurvedic preparations. Ponds constructed inside the catchment forest are primarily silt traps whilst also augmenting groundwater, regulating the water flow to the main tank, and supplying drinking water for wildlife. The belt of large trees in the upper inundation area along the tank acts as a wind barrier, reducing the occurrence of waves in the tank thereby suppressing evapotranspiration. During the wet season, the protected tank also provides habitats for fish and other aquatic species. Meanwhile check dams have been used to retain incoming sediments from the upper catchment and reduce the velocity of overland water flows, while supporting semiaquatic fauna and flora.

This consciously designed multifunctionality clearly supports a rich floral and faunal diversity that directly and indirectly is supported by the system's water management regime, which itself sustains aquatic habitats. For example, the Interceptor area (Fig. 1) alone was found to contain 77 tree and plant species (Dharmasena 2010). At the Anaiwilundawa Ramsar Site in North-west Sri Lanka, a cascade of seven small tanks was found to be linked to a mosaic of artificial and natural habitats including fresh water wetlands (rivers, irrigation canals, marshes, paddy lands), saltwater wetlands (mangroves, salt marshes and maritime grasslands, brackish water canals, beach and sea shore vegetation), and seasonally inundated vegetation around tank fringes which altogether supported 290 plant and 281 vertebrate species (Perera et al. 2005).

Given that over 12,000 small tanks are estimated to exist throughout Sri Lanka's dry zone, the collective contribution to biodiversity becomes clear. Of particular note is that the distribution of small tanks in the dry zone landscape is key to the survival of many of Sri Lanka's larger faunal species, especially elephants for which the tanks become a major if not the only source of water in the dry season. An additional value of this biodiversity is the function of some species as biocontrol agents benefiting farmers by consuming pests such as insects, crabs, and rodents.

While system design demonstrates the confluence of impressive engineering and ecological knowledge, its efficacy relies on social organization, cultural practices, and associated spiritual beliefs that maintain the physical biosphere. For both Buddhist and Hindu farmers, the principles of nonviolence and respect for all living things cast humans as the guardians of nature and not its owners. This is perhaps best embodied in the nonviolent approach to protecting ripening paddy from foraging birds by leaving an unharvested strip of cultivated paddy land adjacent to a tank bund or paddy fields for bird feed, thus minimizing avian crop damage. For Sinhalese farmers, Theravada Buddhism is complimented with village gods such as Aiyanayake, Pulleyar, Pattini, and Kataragama, all reminiscent of Hindu deities, who are worshipped and revered along with Lord Buddha. This religious syncretism finds expression in the pot turning ceremony, *Mutti Mangalyaya*, which pays homage to Buddha and the Gods to ensure protection and prosperity for humans, animals, and the environment. Central to the ritual is the framing of water as a sacred and holy substance and the desire for sufficient rainfall. Locally specific but commonly enacted across the dry zone, rice is the primary offering, with all farmers contributing an amount proportional to their landholdings. Karunanayake's (1978) short, functional explanation of the ceremony highlights the affirmation of communal ties along with the signification of practices of water efficiency and adherence to irrigation rules. This emphasizes the ethics of equity in water distribution and sustainability of the ecosystem.

Conclusions

In conclusion, these wetlands can be viewed as living embodiments of a unique human culture whereby a traditional understanding of hydrology, ecology, and engineering have combined with social organization, institutions, beliefs, and ritual to sustain human life in a challenging ecoclimatic context. It is also striking that these ancient systems of Sri Lanka's dry zone, developed well over 2,000 years ago, accord with some of the central concepts driving contemporary discourses on human development and its relationship to the natural systems that support life on earth. Moreover, the interactions between the landscape and human food production systems through natural and modified processes also make these arguably one of the oldest surviving examples of social ecological systems that sought to optimize utilization of land and water resources based on ecological principles. Furthermore, the juxtaposition of artificial and natural components of this landscape demonstrates the use of built as well as natural or green infrastructure well before these terms were coined in modern development discourse.

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Archaeological Resources and the Protection of Cultural Services

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Benjamin Gearey

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Abstract

This chapter outlines the exceptional potential of wetland environments to preserve organic archaeological remains and associated evidence of past environments (the palaeoenvironmental record) that rarely, if ever, survives in terrestrial contexts. The future preservation of these records are closely tied to the fate of wetlands environments and processes such as erosion, development, drainage and pollution that impact negatively on these ecosystems represent a threat to the long term survival of the resource. This problem is exacerbated by the fact that our knowledge of the archaeological potential of wetlands across the world tends to be focussed on areas that are already threatened by processes that have exposed archaeological sites, for example by peat cutting. Organic archaeological remains and deposits tend to be very fragile and vulnerable, and further research is necessary to understand the prospects for the long term preservation and protection of different sites. Whilst there are examples of efforts to protect threatened sites and landscapes, significant challenges remain in terms of ensuring that

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wetland management, policies and conservation strategies take account of the particular value as well as the specific threats to the archaeological resource in different wetland environments the world over.

Keywords
Archaeology · Palaeoecology · Ecosystem services · Heritage management

Introduction

The waterlogged, anoxic character of wetland environments can preserve important organic archeological remains, which rarely survive in dryland contexts. It has been estimated that an archeologist working in "dryland" conditions may be fortunate to find 10% of what was once there, whilst an archeologist working in a wetland environment might find 90% of the material culture of past communities. Some of the most evocative archeological discoveries of recent years have come from peatlands, for example, the Iron Age "bog bodies" of Tollund and Grauballe Man from Denmark or the 4th millennium BC wooden pathway known as the Sweet Track from the Somerset Levels, southwest England. Other wetland environments such as lakes and estuaries have yielded a remarkable diversity of finds, ranging from entire settlements such as the prehistoric "Lake settlements" of the Alpine forelands, to the footprints of prehistoric peoples preserved in the silts of the Severn Estuary.

Wetlands can also be valuable archives for records of past environmental (paleoenvironmental) change in the form of subfossil material such as pollen, plant, and insect remains preserved in undisturbed accumulations of waterlogged sediment. These records provide critical information regarding vegetation change, human impact on the environment, and Holocene climatic fluctuations (e.g., Chambers et al. 2011). Archeological and paleoenvironmental records (archeo-environmental) constitute "scientific, educational and heritage/cultural" aspects of the Cultural Services category of Ecosystem Services (CICES 2013), but also have relevance for the symbolic and potentially the "sacred and/or religious" service class (Gearey and Fyfe in press).

Threats

The fate of wetland archeological resources the world over is closely tied to the condition and long-term health of the wetlands themselves (e.g., Van de Noort et al. 2001). Loss and destruction of archeological resources can occur through direct processes such as erosion in coastal areas and river valleys, development, agriculture, pollution (changing soil chemistry), or peat cutting which, though diminishing in some countries, remains a significant source of fuel in countries including the Republic of Ireland. Indirect impacts can have an equally deleterious effect on organic remains. Indirect threats may be defined as any process which may lead to conditions inimical to the long-term stability and preservation of the archeological and paleoenvironmental resource (see Gearey and Fyfe in press). For example, de-watering as a consequence of drainage of peatlands can lead to desiccation and degradation of the peat matrix and thus the deterioration and destruction of any sites preserved within this matrix. Other processes can also be indirectly damaging, such as rising sea levels that threaten coasts resulting in the loss of coastal wetlands and associated archeological resources (see for example, Van de Noort 2013).

Policy for Protection of Archeological Resources

Preservation *in situ* has long been the preferred management option for archeological sites across Europe and is recognized as a principal tenet of the European Charter for the Protection and Management of Archeological Heritage (1992), commonly known as the "Valletta Convention". Although protection under national legal frameworks may recognize the value and importance of specific sites, this cannot in itself always protect the resource; the deteriorating burial environment of the internationally important prehistoric wetland site of Star Carr, Yorkshire, England, means that preservation by record is the only feasible option (Vorenhout 2012). It has been estimated in England that around 10,450 sites have been destroyed in lowland peatlands over the last 50 years (Van de Noort et al. 2001). Brunning (2007, p. 46) has stated that despite the supposed legal protection of many wetland sites: "The well proven, extensive and rapid destruction of waterlogged archaeological deposits in European peatlands should be regarded as a significant crisis".

Protecting and Managing the Resource: Examples

There are significant current and future challenges associated with the protection of the resource. In some cases, these have been recognized and responded to, as for example in the case of eroding shorelines threatening the Alpine lake villages in western Switzerland where management interventions, such as the construction of fences and emplacement of other protective measures round certain "at risk" areas in Lake Neuchatel, have proven successful in deflecting the force of waves and hence stabilizing shorelines (Ramseyer 2013). The assignation by UNESCO in 2011 of the "Prehistoric Lake-dwellings around the Alps" to the World Heritage List should also be regarded as another positive step towards protecting this landscape.

In other wetlands sites can be protected by informed management. The only wetland archeological site in the Somerset Levels, southwest England, that appears secure from the threat of desiccation is the section of the Neolithic Sweet Track that benefits from a pumping system, which keeps water levels high in the Shapwick Heath National Nature Reserve (Brunning 2013). Similar efforts to stabilize the burial environment are underway on Bourtanger Moors, northeast Netherlands, where the surviving section of the Nieuw Dordrecht Neolithic timber trackway is threatened by desiccation.

Future Challenges

Whilst there are examples of projects and initiatives aimed at protecting the archeological resource in wetlands, significant problems and challenges remain. The archeological resource is finite, nonrenewable, and fragile and can be easily damaged or destroyed by any process or activity which impacts on the wetland ecosystem within which they are preserved. A number of future challenges can therefore be identified to ensure improved protection of archeological resources. A major problem remains the fact that known sites tend *de facto* to be concentrated in areas that have been most heavily disturbed, and hence are already at significant risk. Given the difficulty of remotely identifying (using geophysical techniques for example) archeological sites in many wetland contexts, it is often the case that sites are only discovered during the course of interventions or other disturbances. This leads to significant problems in terms of ensuring sites are subsequently protected or, if this is not possible, that resources are made available to recover and/or record the threatened sites/deposits. Although there have been significant advances in knowledge over the last decade or so, understanding the preservation environment of different wetland contexts and hence prospects for the long-term *in situ* survival of associated archeological resources may be highly complex and locally variable.

Another challenge concerns promotion of the value of the wetland archeological resource to both the general public and to conservation bodies and other organizations. Whilst there are examples of successful cooperation between different agencies and stakeholders to ensure optimal protection of sites, there remains a general lack of appreciation of the value and significance of the cultural aspects of wetlands, how best to avoid deleterious impacts and how to design appropriate mitigation strategies (Olivier 2013, p. 689).

Gearey et al. (2010) observed that, whilst legislation may be available to enable the protection of the best and most valued wetland archaeological resources it is only as effective as the knowledge underpinning resource selection and the capacity consistently to apply such legislation. The principle of preservation *in situ* means that, wherever possible, sites should be preserved in the ground rather than be excavated. However, resources can only be protected if they can be located, characterized, and demonstrated to be stable within their associated burial environment; despite advances in recent years, these issues can be highly problematic. Without input from the historic environment sector, management actions that might impact directly or indirectly on wetlands pose a risk to the *in situ* integrity of the record. For example, conflicts may arise when certain conservation measures that do not require planning consent are carried out, which may preclude input from the historic environment sector and possibly lead to the discovery of and/or unintentional damage to the archeological and paleoenvironmental resource.

Olivier (2013, p. 693) has recently stated that: "...we should re-think our primary approach to cultural heritage in wetlands, and move from existing orthodoxies of protection and preservation to a flexible approach in tune with prevailing attitudes to sustainability and environmental change". Successful management of the archeological resource in wetlands is therefore dependent upon the development of consistent policies and guidance, supported by a better understanding amongst all stakeholders of the fragile and threatened nature of the resource.

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Recreational Management and Wetlands 194

Mark Everard

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Abstract

Wetlands support a diversity of recreational uses (cultural services) in addition to a far broader, interlinked set of ecosystem services. One of the key challenges for wetland managers is to sustain all of these values, without detriment to the natural character and functioning of the wetland.

Keywords

Recreation · Angling · Water sports · Disturbance · Conflict · Planning

Introduction

Wetlands support a diversity of recreational uses (cultural services) in addition to a far broader, interlinked set of ecosystem services. One of the key challenges for wetland managers is to sustain all of these values, without detriment to the natural character and functioning of the wetland.

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Interactions Between Wetland Recreational Activities

Among the many recreational services provided by wetlands are angling (▶ Chap. 149, "Recreational Fishery Case Study (UK)"), boating in its various forms, bird watching and contact with nature, quiet pursuits such as painting and internal reflection, and walking. This far from exclusive list of recreational activities varies substantially in character and in terms of requirements.

There are some clear conflicts between some of these wetland recreational activities. For example, anglers and canoeists are in frequent conflict where opportunities are limited and where agreements have not been reached (Angling Trust 2013). Equally, enjoyment of quiet recreation (painting, bird watching, walking) may be compromised by noisy recreational activities (such as jet-skiing or power-boating).

Interactions with Other Services

However, the recreational value of a wetland is far from the only important consideration. To make balanced judgments in wetland management, as in any form of ecosystem management consistent with the Ecosystem Approach, potential antagonisms and synergies across the full framework of ecosystem services need to be considered.

Fundamentally, the natural character of the wetland itself needs to be set as a priority. Without this as a basis, including the full range of functions and supporting ecosystem services typical of the wetland type, the wetland will be likely to be degraded through use. Loss of wetland integrity has consequences for recreational services, but also for the continued functioning of the wetland and its capacity to provide other important services. These supporting services include such important aspects as "habitat for wildlife," "soil formation," and the cycling of water, nutrients, and carbon. This immediately places some constraints on the timing and extent of certain activities, be that walking or any form of access in sensitive bird nesting or loafing areas, or the more obvious disruptive consequences of wash and noise from motorized water sports.

The broader provisioning, regulatory, and cultural services supported by the wetland need also to be considered, as synergies and antagonisms will also be found between them and various recreational activities. These interactions are very much multidirectional.

Conflicts over resource use are all too common, for wetlands and other habitat types. Uneven appropriation or trespass by any party assuming untrammeled rights is likely not merely to compromise the integrity and function of the wetland, but also the rights of others.

Managing Conflicts

Principles 11 and 12 of the Convention on Biological Diversity (n.d.) Ecosystem Approach refer, respectively, to taking account of all forms of knowledge and the inclusion of stakeholders in decision-making. Other principles of the Ecosystem

Approach touch upon management as a matter of societal choice (1), decentralized decision-making (3), consideration of adjacent effects (3), conserving ecosystem structure and functioning (5), and natural limits (6). This then highlights the importance of putting the integrity of the wetland resource first in planning, but also involving the broadest constituencies in integrated, locally based planning.

The Ramsar Management Plan (Ramsar Convention 2002) process recognizes the Ecosystem Approach principles, addressing integrating wetland site management within broader-scale environmental management planning, engaging stakeholders including local communities and indigenous people, taking a precautionary approach, and identification of management units that may be both used and protected through measures such as zonation and buffer zones.

This approach to integrated wetland management, directed by the Ramsar Management Plan process, directly by the Ecosystem Approach or developed by learning, finds a range of manifestations in wetland and in integrated land and water management. In the USA, Special Area Management Plans (SAMPs), specified in the Coastal Zone Management Act (CZMA), are used to address specific management goals within specified geographical locations (National Oceanic and Atmospheric Administration 2007). The SAMP process can refine or tailor existing policy in situations where more general coastal policies do not adequately address the specific conditions found in a particular area, serving as a tool for adaptive management when necessary. They can also improve coordination between adjacent jurisdictions. It is implicit in the SAMP planning process that key participants are identified and included in the planning process, including appropriate, timely, meaningful stakeholder and public participation in the development and implementation of the plan.

In Europe, these principles can be found in progressive legislation such as the EU Water Framework Directive (EU 2000) (▶ Chap. 71, “European Union Water Framework Directive and Wetlands”) and the Marine Strategy Framework Directive (EU 2008).

Nationally, strategies such as the UK’s Marine Protected Area process (JNCC n.d.) and the Catchment Based Approach (Defra 2013) in England apply this approach at regional and catchment scales. Local implementation may, for example, zone areas for particular activities such that different recreational uses and the wider needs of the wetland itself and the other services it provides can be accommodated harmoniously.

The most effective plans work by consensus, rather than enforcement (though some limitations on recreational activities may be limited by law). This then has parallels with Ostrom’s (1990, 1997) theory for the management of “self-organized, common-pool resource regimes (CPRs)” encompassing attributes including defining the boundaries of the common, developing appropriate appropriation and provision rules, participation of most appropriators in the decision-making process, effective monitoring, graduated sanctions, conflict-resolution mechanisms, local organization, and, where necessary, a spatially nested approach.

Conclusions

Wetlands support diverse recreational uses. However, there are potential conflicts not only between various of these recreational uses but also other categories of wetland ecosystem services. One of the key challenges for wetland managers is to sustain all of these values, without detriment to the natural character and functioning of the wetland. The Ecosystem Approach principles, in addition to the related Ramsar Management Plan process and other systemic mechanisms, emphasize the importance of considering wetland structure and functioning as a core consideration but also planning on the basis of engagement and consensus-building. Adaptive management is also an important principle given the unpredictability of outcomes.

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Sustainable Wetland Tourism

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Mark Everard

Contents

Abstract

As one billion tourists were expected to cross international border in 2012, with an anticipated rise to 1.8 billion by 2030, the environmental impact of tourism is far from inconsequential. Wetlands range in type from beaches and coasts to coral reefs, swamps, lakes, rivers, marshes, and a diverse range of other types, all of them potentially supporting an equally diverse range of tourism activities. However, tourism in turn places pressures on wetlands and local communities. "There is therefore a need to manage wetland tourism wisely through sound policies, planning, and awareness-raising."

Keywords

Tourism · Benefits · Conflict · Participation · World Tourism Organization

Introduction

As one billion tourists were expected to cross international border in 2012, with an anticipated rise to 1.8 billion by 2030, the environmental impact of tourism is far from inconsequential (Ramsar Convention and World Tourism Organization 2012).

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Wetlands range in type from beaches and coasts to coral reefs, swamps, lakes, rivers, marshes, and a diverse range of other types, all of them potentially supporting an equally diverse range of tourism activities. However, tourism in turn places pressures on wetlands and local communities.

There is therefore a clear need to manage wetland tourism wisely through sound policies, planning, and awareness-raising. In recognition of the interdependence between sustainable tourism and the conservation and sustainable management of wetlands, the Ramsar Convention and World Tourism Organization (UNWTO) signed a Memorandum of Cooperation in February 2010 to enhance cooperation between both organizations (Ramsar Convention 2010).

Promoting the increasing focus on tourism, the Ramsar Convention (2011) took as a theme for World Wetlands Day 2012 the slogan *Wetlands and Tourism*. Another outcome of the collaboration between Ramsar and the UNWTO was production of the guide *Destination wetland: supporting sustainable tourism* (Ramsar Convention and World Tourism Organization 2012), aiming to put the principles of sustainable tourism into action in wetlands managed for tourism benefits. Much of the text in this article derives from that guide.

Interactions Between Wetlands and Tourism

Wetlands are of considerable value for tourism, and the economic benefits that tourism can bring can also support the management of wetland sites. Benefits for wetlands and associated communities include promotion of nature conservation goals, poverty alleviation (through improvement of local livelihoods), enhancement of regional and national economies, and support for local cultures. However, there are also associated risks and impacts when tourism is not well managed. These include impacts both from development and operation of tourism facilities, such as degradation of wetland areas for extraction of building materials, infrastructure, overabstraction of water, inappropriate waste disposal, noise pollution, excessive trampling, disturbance of wild species, and a range of other pressures besides. Mutually supporting values can only continue if sustainable practices are carefully planned and observed.

Coastal and beach tourism remains the dominant segment in terms of number of tourists, though the tourism industry is diversifying. Consequently, the natural and cultural resources of coastal and island destinations experience increasing pressure from the ever increasing demands of tourism activities. These activities are often highly concentrated in time and space (seasonality, infrastructure, and tourism operations occurring in a narrow coastal zone). There are therefore important linkages to be made between tourism development and Integrated Coastal Zone Management practices, achieved through local, regional, and international collaboration. Among other measures addressing this challenge is the Mauritius Strategy (UN World Tourism Organization 2005), aimed to promote the sustainable development of small island developing states (SIDS), through a set of actions under 20 broad headings many of which have implications for the tourism sector.

The *Destination wetland: supporting sustainable tourism* guide also recognizes a wide range of other wetland types in 14 case studies including, as examples, a high altitude freshwater lake and connected marshes, coral reefs, a connected mangrove and mud flat system, a shallow alkaline lake in an enclosed basin, rivers and a river delta, a raised bog, and an enclosed urban woodland. All of these wetland types offer an array of tourism opportunities.

The value of recreational angling in wetlands, the subject of a separate case study in this Wetland Encyclopaedia, has substantial tourism value. Practices such as catch-and-release fishing can reduce the impact of this tourism activity on wetlands. Furthermore, development of angling and related ecotourism that involves and ensures direct benefits to local communities can serve mutually beneficial outcomes for the wetland, tourists and local people (Everard and Kataria 2011). Similar win-win outcomes can be conceived and planned for other wetland tourism activities.

Key Messages for Sustainable Wetland Tourism

In order to achieve these goals, wetland management planning has a crucial role to play to resolve the multiple potential conflicts entailed in wetland tourism. Development plans for tourism should therefore be integrated within wetland management plans addressing biodiversity conservation, ensuring that objectives for wise use and conservation are not compromised by proposed tourism activities. This necessarily entails the involvement of local communities in decision-making, also ensuring that benefits accrue to local people and contribute to their sustainable livelihoods. The needs of local people and the demands of tourism enterprises should be resolved consensually within this planning framework if future conflicts are to be avoided, including ensuring that wetland visitation rates are ecologically and socially acceptable. Communication and education of tourists about the values of wetlands and their biodiversity can also strengthen support for wetland conservation.

The culture of the tourism sector and of wetland managers is often entirely different. Close dialogue is therefore required to determine how best to integrate the financial viability of tourism with the needs of wetland conservation. Ideally, overlapping areas of mutual benefit will be identified through dialogue and consequent forward planning. Involvement of local business interests can be helpful in ensuring long-term conservation and social benefits. This can include high-quality local guiding and interpretation, improving the visitor experience and contributing further to mutually beneficial goals. The siting of facilities for tourists is an important consideration in minimization of adverse impacts, as is marketing to set realistic expectation of what the wetland can or cannot offer.

Of course, these planning considerations must fit with regional and national planning policies, and compliance with environmental and other regulations is also essential to avert unintended damage.

Conclusions

Wetlands offer a diversity of opportunities for the burgeoning global tourism market. There are opportunities to link tourism activities with wetland conservation goals and benefits to local people. However, there are also many potential conflicts that need to be carefully managed. Dialogue between tourism interests and wetland management, also involving the participation of local stakeholders, is essential to avert harm to wetland character and ecology. Involving local people can also enhance the tourism experience, reinforcing mutual benefits for tourists, wetland conservation, and local livelihoods.

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Religious and Spiritual Aspects of Wetland Management

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Bas Verschuuren

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Abstract

The spiritual significance of wetlands is closely related to the religious, cultural and historic importance wetlands play in human well-being. Spirituality contributes significantly to wetland services and values but often remains overlooked and undervalued. Indigenous peoples' spirituality is often directly related to wetlands being imbued by spirits while mainstream religions construct places of worship in wetlands. Pilgrimages the world over follow rivers and wetlands and in cases these can have a profound impact. Religious leaders can help protect wetlands and some incentives in international policy exist to assist policy makers and decision makers with this.

Keywords

Amenity · Religion · Sacred natural sites · Spirituality · Wetlands

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Introduction

Since the beginning of civilization, the world's societies have all been dependent on water and many of them directly on wetlands. Significant human uses of wetlands have included fishing, farming, drinking water, transportation, and energy production. Besides these tangible benefits, increasingly expressed in terms of provisioning services, wetlands are also significant for their important intangible, nonmaterial, and cultural services. For example, wetlands are frequently sites of pilgrimage and spiritual fulfillment, and their waters are often used in rituals and for healing purposes. Wetlands are life, and today the blessing of their waters is part of nearly every mainstream faith (Papayannis and Pritchard 2010). These intangible and other examples of spiritual services and religious aspects have implications for the management and conservation of wetlands.

To Tibetan Buddhists, for example, high mountain lakes are sacred pilgrimage destinations known as "the mirrors of the soul". The lake of Lhamo La-tso, also known as "Oracle Lake", was a site of visions for the second Dalai Lama, and has since been visited by all eleven successive Dalai Lamas to receive visions. Buddhist Monks go to receive visions that aid the search of incarnations of the Dalai Lama. Tibet's lakes are also believed to be protected by the fish that live in it, which are regarded to be so sacred that hardly any Buddhist in Tibet would catch or eat fish today (Jacobsen et al. 2013).

This Article introduces the reader to the spiritual services and religious aspects of wetland ecosystems as well as the current classifications used to enable their wise use and management. A nonexhaustive overview of wetlands with spiritual and religious significance is provided, drawing on examples from wetlands from around the world. The religious aspects of wetlands are then explained for indigenous and local communities as well as for mainstream faiths. Examples are given that provide insights into the complexity of spiritual services and religious aspects of wetlands, illustrating the aforementioned classification. Special attention is given to the phenomena of pilgrimage in relation to environmental impact and the role of religious and spiritual leaders in wetland conservation.

Spiritual Services of Wetlands

The world's wetlands provide 15 trillion US$ a year in supporting, provisioning, regulating, and cultural services, the later being defined as "...nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience, including, e.g., knowledge systems, social relations, and aesthetic values" (Millenium Ecosystem Assessment 2005). The Ramsar Convention Parties have adopted the same classification, and further have recognized the importance of cultural aspects of wetlands in the original Convention text (1971) and subsequently in more detail, for example through Resolutions VIII.19 and IX.21 (Ramsar 2002, 2005). The first Resolution provides recognition and a classification of nonmaterial values. The Convention also has a Working Group on Culture and Wetlands which developed a guidance document, taking a broader approach to culture and wetlands that is inclusive of their spiritual and religious importance (Papayannis and Pritchard 2008). In addition, Verschuuren (2007) also provides a nonexhaustive classification of sociocultural values describing spiritual services as “The qualities of wetlands that inspire humans to relate with reverence to the sacredness of wetlands” and suggests the following indicators:

– Presence of sacred sites or features of wetlands
– Role of wetlands in religious ceremonies and sacred texts
– Oral tradition, song, chant, and stories based on wetlands
– Totemic wetland related species
– Religious use of wetland flora and fauna
– Presence of wetland-related traditional healing systems

Similar to most nonmaterial values of wetlands, the spiritual services and religious aspects are often difficult to measure and may therefore be overlooked in decision-making processes.

The Spiritual and Religious Significance of Wetlands

Around the world, many types of wetlands are recognized for their spiritual, religious, and sacred significance (see Fig. 1). The spiritual and religious significance of wetlands is mostly associated with either crosscultural mainstream religions and localized indigenous spiritualties or the intersection of the two.

Spiritual values are defined as “...relating to or affecting the human spirit or soul as opposed to material or physical things. . .” and at a deeper level as “. . .having a relationship based on a profound level of mental or emotional communion” (Oxford 2013). Following this definition, wetlands may be seen as a source of spiritual inspiration and experience. It becomes clear that a range of new, revived, and socio-cultural types of religious and spiritual values can also be recognized. These values can be experienced and expressed by local inhabitants, conservationists, tourists, fishers, and other people who deeply relate to wetlands.

Spiritual values are understood as a particular subcategory of sociocultural values. They embody the intangible and sometimes supernatural dimensions related to wetlands. The later is particularly the case when the spiritual experience is codified by myth and allegory, in the personification of deities, by magical properties, and is often explained and expressed through religious life. Religion or “re ligare” in Latin literally means “re-linking” and refers to the different dimensions of reality from the material to the spiritual. In the west, during the enlightenment period, science largely displaced religion in the intellectual firmament and subsequently formed the basis for most modern conservation approaches (Palmer and Finlay 2003). Of late, the importance of spiritual and religious values to biodiversity

Fig. 1 Map and table presenting an overview of wetlands across the world that are of sacred, religious, and spiritual significance. *1*. **Africa, Egypt: River Nile and inundation plains**. Various designations of protection cover the cyclical ebb and flow of the river waters and riverside floodplains. The Nile is of historic sacred importance where the gods controlled flooding and the river was seen as a causeway from life to the after-life. *2*. **Africa, Niger Delta, Niger: Adigbe and Esiribi Lake in Osiama and Biseni**. These coastal wetlands are part of an Indigenous and Community Conserved Area covering tidal freshwater, marsh, and flood forest zone dissected by freshwater creeks. Sacred crocodiles and lakes are managed through a system of sacred natural sites, festivals, and taboos that help fishing and local management regulations. *3*. **Americas South, Bolivia and Peru: Lake Titicaca**. A Ramsar Wetland of International Significance, Latin America's largest brackish water lake is known as the cradle of the Tiwanaku, Aymaras, and Incas civilization. Throughout colonization, some of the most relevant sacred natural sites were transformed into Christian holy sites where many religious celebrations and pilgrimages take place. *4*. **Americas South, Colombia: Lake Guatavita:** The lake is designated as a Natural Park. This sacred freshwater lake of the Muisca is known for a ritual thought to be the basis for the legend of El Dorado. The Muisca celebrated a ritual in which the Zipa was covered in gold dust to dive off a ceremonial raft washing off the gold. *5*. **Americas Central, Yucatán, Mexico: Anillo de Cenotes (ring of cenotes)**. This Ramsar Wetland of International Significance has parts protected respectively as State, Special, and Biosphere Reserves. A cavernous complex carst network forms the prime source of potable water. Imbued with sacredness by the Mayan people, the cenotes are ritual places seen as the gateways to the afterlife. *6*. **Americas North, Canada: Southern James Bay**. These coastal and estuarine wetlands are recognized as a Ramsar Wetland of International Significance. The traditional territory of the James Bay Cree Nation were religious beliefs are linked to animal spirits. These animals are tied to the wetland ecosystem and, as such, the Cree are part of a scared ecology. *7*. **Asia, Bali, Indonesia: Subak Landscape of Pekerisan Watershed**. The Subak landscape is a World Heritage Site which embodies "Tri Hita Karana", the harmony between man, nature, and God. Ancient self-governing associations of farmers share the use of irrigation water for their rice fields and water temples that dominate the landscape are devoted to the people's spiritual relationship with water. *8*. **Asia, Tibetan Autonomous Region: China**. Lake Manasarovar. The Himalayan high altitude lake lays at 4,590 m above mean sea level. It is a place of pilgrimage, attracting religious people from India, Nepal, Tibet, and neighboring countries. Bathing in the Manasa Sarovar and drinking its water is believed to cleanse all sins. The Kailash Manasa Sarovar Yatra pilgrimage takes place every year. *9*. **Asia, China: Xixi Yangtze Delta Wetland**. A National Wetland Park and Ramsar Wetland of International Significance. Xixi-related culture dates back 5,000 years. During the Dong Han dynasty (AD 223), Buddhists gathered to drink water and temples were built. From the Tang Dynasty (AD 618) onwards, Xixi's was reflected in spiritual writings. In 1465, the Dragon Boat Festival was established. **10. Asia, Philippines: Agusan Marsh**. Agusan Marsh is a Wildlife Sanctuary, Ramsar Wetland of International Significance and is

conservation is being rediscovered (Posey 1999) and consequently these values are increasingly recognized in wetland management.

Indigenous Spiritualties and Wetlands

In some cases, a wetland may have spiritual or religious significance because of some factor that is not particularly related to its functioning as a wetland ecosystem. However, the spiritual and religious significance of wetlands to most indigenous people tends to relate directly to natural elements and species. These are often imbued with spirits that reside in nature and are "numinous" in that they possess agency as sources of wisdom and law (Byrne 2010). Indigenous people often express that these wetlands themselves are alive, and venerate them through reciprocal relationships as they would with social relations, a phenomenon typical of animism.

Fig. 1 (continued) listed as tentative World Heritage Site. It comprises freshwater swamp forest, lakes, and manmade rice paddies and fishponds. It is the ancestral territory of the Manobo people, who live in floating houses and perform rituals to the spirits of their ancestors residing in the lake. *11*. **Asia, India: Triveni Sangam**. The confluence of the Yamuna, Ganges, and the invisible and spiritual Sarasvati River is a sacred place for Hindus. A bath here is said to wash away one's sins and free one from the cycle of rebirth. It is the site of the historic Kumbh Mela, held every 12 years. Over the years, it has also been the site of immersion of ashes of several national leaders, including Mahatma Gandhi in 1948. *12*. **Europe, Spain, La Doñana**. A National Park, IUCN Category II, Natural Park, IUCN Category V, Ramsar Wetland of International Importance and EU NATURA 2000 site, the marshes, beaches, and mobile include 88 archeological, prehistoric, and Roman sites. The annual pilgrimage and ceremonies honor "la Virgen del Rocío" also named "Queen of the Marches" exist throughout the wetlands. *13*. **Europe, United Kingdom: Lindisfarne Holy Island**. This National Nature Reserve, IUCN category IV, and a Ramsar Wetland of International Importance includes marine wetlands, mudflats, and marshes that have been part of the "cradle" of Christianity in northern England and southern Scotland since AD 635. Here nature and spirituality are very much linked here through a line of "Nature Saints". *14*. **Europe, Camargue, France, Saintes Maries de la Mer**. These riverine wetlands in the Rhone Delta form the Natural Regional Park of the Camargue, IUCN Category V, including two public nature reserves and one large private nature reserve, IUCN Category III. They have constituted a sacred site since prehistoric times, venerated by the Celts, Romans, Christians, and thousands of gypsies who come on annual pilgrimage. *15*. **Europe, Montenegro and Albania, Lake Skadar**. Ramsar Wetland of International Significance and National Park on the Montenegrin side. The largest open water surface in the Balkan Peninsula accommodates some 20 Christian orthodox monasteries, several scattered holy monuments, and Mount Rumija, with an annual religious procession. *16*. **Oceania, Northern Territory, Australia: Kakadu**. This National Park, in comanagement with local Aboriginal people, is also a World Heritage Site and Ramsar Wetland of International Significance. Land, wetlands, and culture have become inextricably intertwined through song, dance, and creation stories evidenced in a 50,000 years old living art tradition in rock and bark painting. *17*. **Oceania, Federal States of Micronesia: Nan Madol, Pohnpei**. The national Archeological Site of Pohnpei is inscribed on the tentative World Heritage list. It consists of coastal coral reefs on island atolls with manmade canals on land. Nan Madol architectural ensemble exhibits the most perfectly preserved habitation, ceremonial plan, and Nanmwarki leadership system and of the Pacific region. (Image credit: B Vershuuren © copyright remains with the author)

Many wetlands are sacred to and venerated by local and indigenous people. An example is the lakes of the Niger Delta that are home to the endangered West African Dwarf Crocodile (*Osteolaemus tetraspis*) and are sacred to the Bisieni and the Osiama people (Anwana et al. 2010). The crocodile is believed to be the peoples' brother. If killed accidentally, it is given full funeral rites akin to those afforded to humans. When killed intentionally, it is replaced with a live crocodile.

The beliefs of the Bisieni and Osiama have not only led to the conservation of a top predator in the wetland ecosystem but they are also at the basis of customary fishing regulations which secure healthy fish populations (Anwana et al. 2010). The sacred status of the lakes means that fishing occurs only when permitted, in groups, within specific dates and seasons and using selective fishing methods. Common ancestry and social connectedness among neighboring communities link the management of some of these lakes together, forming a network of lakes reserved for periodic fishing based on communal regulations.

Sacred lakes, as with their terrestrial counterparts the sacred groves (trees and woodlands), tend to be part of fragmented ecosystems as they are usually small remnants of habitat preserved by indigenous people (Bhagwat and Rutte 2006). However, the social cohesion amongst these groups secures their management as connected ecosystems throughout the regions concerned, thereby reducing fragmentation of the habitat (Anwana et al. 2010). The example of these culturally protected freshwater lakes therefore provides an insight into how wetland management in the Niger Delta can be strengthened through recognition, respect and support of indigenous people's spiritual and religious practices and beliefs.

In the hamlet of Westerhoven in the Southern Netherlands, a pre-Christian sacred well is known for its healing properties which have kept it safe from agricultural and religious reformations. Christians started using it for baptism and later it was devoted to St. Valentine (14 February) by appointment of St. Willebrord which turned it into a pilgrimage designation. Next to the well, a chapel arose, and recently a nature reserve was created on its adjacent agricultural lands to preserve the typical lowland freshwater creek that feeds it (Fig. 2).

Mainstream Faiths and Wetlands

With the exception of several mainstream faiths that can be considered to be rooted in indigenous spiritualities such as Daoism, Shinto, Hinduism, and Jainism (Palmer 2003), the religious values of other mainstream faiths (Buddhism, Judaism, Christianity, and Islam) are likely to view wetlands as part of a Godly creation. The opening passage of the Holy Bible's Genesis in this respect forms a captivating introduction to the religious importance of the world's first wetlands according to the Christian tradition (The Holy Bible undated).

> In the beginning God created heaven and earth. And the Earth was without form and void; and darkness was upon the face of the deep. And the Spirit of God moved upon the face of the waters.

Fig. 2 Pre-Christian sacred well with Christian Chapel, Westerhoven, Netherlands (Photo credit: B Vershuuren © copyright remains with the author)

This passage also illustrates how, within this second group of mainstream faiths, the spiritual importance of the environment is related to the creation as a work of God. This very act of creation presumes an omnipresence of God manifest in the natural elements. Rather than seeing natural elements imbued with spirits that require to be venerated and treated with respect, the environment is believed to have been created by God for man to subdue.

Religious associations with wetlands are also made with places of significance to religious persons, holy men, or saints that play or have played important roles in those faiths. Some of the religious or spiritual importance of these wetlands may arise out of the specific ecological properties of the area's being a wetland such as the use of water in baptism (see Fig. 2), the water as a source for isolation required for hermits or the rising of a saint from the marshlands such as in La Doñana, Spain (see Fig. 1). By contrast, other religious and spiritual importance can be *independent* or less directly relating to the wetland's nature, including as examples wetlands as locations for man-made structures such as mosques, chapels or temples. In some cases, expansion of such manmade constructions goes at the expense of wetland integrity, especially where large-scale pilgrimages or mass gatherings are being accommodated. Examples of these are the pilgrimages in La Doñana in Spain, Saintes-Maries-de-la-Mer in the Camargue in France (Mallarach 2011), and the Kumbh Mela at Triveni Sangam in India.

Pilgrimage Across Sacred Rivers and Wetlands

Pilgrimage – travel with a spiritual purpose – is often linked with wetlands, rivers, or their sources, sometimes in far-flung high altitude regions. Sacred wetlands, and more so the rivers that feed them, are found on every continent (see Fig. 1). Many are only known to local religious and spiritual practitioners while others have earned world fame through their importance in mainstream faiths.

In India, for example, it is estimated that around 250 million people perform pilgrimages every year (Singh 2013) and many of these pilgrimages lead through and by wetlands and waters (Fallon and Jaiswal 2012). Pilgrims and their spiritual practices also impact on the environments they intersect. This requires organizational responses from wetland management and planning authorities. The Kumbh Mela in India, for example, is arguably the world's largest human gathering, with approximately 333 million people participating in 2013, and it is so large that is can be observed from space. Many pilgrims taking part in the Kumbh Mela also bathe in the river Ganges, for example at Triveni Sangam (see Figs. 1 and 3). The river is sacred to some 200 million Hindus, and many of them undertake the pilgrimage to bathe in the river at least once in their lifetime.

At Triveni Sangam, the confluence of the sacred Ganges, the Yumuna, and the invisible and mythological Sarasvati Rivers, pilgrims of the Kumbh Mella use boats to escape the masses at the river banks in order to perform rituals and bathe in the holy river to wash away their sins.

The Role of Spiritual and Religious Leaders

Despite the religious importance of wetlands, many of their guardians and managers are required to deal with the impacts of pilgrimage. With some 80% of the world's population adhering to one or other of the mainstream faiths – of which a large part is drawn from traditional or folk religions – the impact of religious leaders can be significant (O'Brien and Palmer 1997). Religious and spiritual leaders often have moral, ethical, and theological obligations of environmental care (Palmer 2003), which is why they should be at the forefront of conservation (Abraham 2013). In many indigenous cultures too, spiritual leaders and custodians of specific manmade and natural sacred sites in wetlands play a key role in environmental conservation and wise use of wetlands (Verschuuren et al. 2010). They are often guardians of biological diversity and ensure that related, and sometimes sacred, knowledge, and rituals are looked after respectfully while they can also educate and mobilize communities. It is important in the context of wetland management that spiritual

Fig. 3 Ritual bathing at Triveni Sangam, Allahabad (Photo credit: Puffino © copyright remains with the author)

and religious leaders of all traditions are respected, and that their support for securing the natural and cultural elements of wetlands is maintained.

Spiritual and Religious Values in Conservation Policy

Religious and spiritual practices and beliefs are recognized as rights of all humans under international declarations such as ILO 169 (1989) and UNDRIP (2007). Their importance is also recognized under the UNESCO Conventions on Diversity of Cultural Expressions (2005) and Safeguarding of Intangible Cultural Heritage (2003). The Ramsar Convention offers Resolutions VIII.19 (2002), which elaborates on the recognition of culture and local communities participation in wetland management, and Resolution IX.21 (2005), which strengthens attention to cultural values in policy and management. The Convention on Biological Diversity's (CBD) article 8j provides specific guidelines on indigenous knowledge and cultural practices, including a code of ethical conduct (SCBD 2011) and Guidelines for cultural, environmental, and social impact assessments related to sacred sites (SCBD 2004). Following the IUCN recommendations on Cultural and Spiritual Values of Protected Areas (WPC 2003: Recommendation V.13), IUCN and UNESCO have developed guidelines for protected area managers on Sacred Natural Sites (Wild and McLeod 2008).

Upcoming rights-based conservation approaches have given rise to the rapidly growing concept of Indigenous and Community Conserved Areas (ICCAs), and Sacred Natural Sites (GEO5 2011). Many ICCAs and SNSs contain wetlands of spiritual and religious significance that may not yet be designated with protective status but are thought to make a significant contribution to target 11 in the Aichi Biodiversity Targets. Target 11 seeks to expand the world's protected area coverage of terrestrial and inland water systems from 12.1% to 17% and for marine systems from 2.2% to 10% by 2020. This projected 27% will bring protected areas to the world's single largest form of land use. Comprising some 205 million hectares, Wetlands of International Importance listed under the Ramsar Convention contribute substantially to that global figure. It has to be noted though that all wetlands located within the borders of the countries that are signatories to the Ramsar Convention come under its official protection. Furthermore, it would be important to the wise use of all wetlands, regardless of their conservation status, to fully recognize their spiritual services and religious aspects in management practices and related decision-making processes.

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Landscape Aesthetics and Wetlands

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Mark Everard and Robert J. McInnes

Contents

Abstract

Wetlands span a diversity of wet or periodically inundated habitats, with an equally diverse set of associated cultural interpretations and values. For this reason, wetland landscapes are ascribed a wide range of inspirational qualities, but also some negative associations.

Understanding how the public and different decision-makers interpret wetlands, both natural systems and proposals for constructed or restored wetlands, is important in gaining support for their design, implementation, and management. Landscape architects may have an important role within multidisciplinary teams in translating technical knowledge into forms more readily understood and supported by the public and other nontechnical audiences.

Public understanding of wetlands, both promotion of their inspiration qualities and reinterpretation of potential negative associations, is vital for the success of wetland conservation, recreation, and construction. Framing this in ways that are accepted and welcomed by the public and decision-makers remains a key challenge.

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Cultural · Values · Inspirational · Holy wells · Sacred groves · Funery · Recreation · Tourism · Aesthetic · Green infrastructure · Landscape architects

Introduction

Wetlands embrace a diversity of wet or periodically inundated habitats with equally diverse cultural interpretations and values. For this reason, wetland landscapes are ascribed a wide range of inspirational qualities, but also some negative associations.

Understanding how the public and different decision-makers interpret wetlands, both natural systems and proposals for constructed or restored wetlands, is important in gaining support for their design, implementation, and management. Landscape architects may have an important role within multidisciplinary teams in translating technical knowledge into forms more readily understood and supported by the public and other nontechnical audiences.

Cultural Values Associated with Wetlands

A wide diversity of cultural interpretations and values are associated with wetlands, a definition that spans a range of wet or periodically inundated habitats. Inspirational qualities attributed to wetlands are also diverse. These include, for example, holy wells, sacred groves and islands, baptism and funerary sites, the holy rivers of India, large open skies over saltmarshes and coastal zones offering inspiration to landscape and wildlife artists, places for holidaying or other forms of recreation, and green/blue spaces in congested urban environments. Classic landscape paintings, for example John Constable's *Hay Wain*, also commonly feature water and wetlands as central features. Beaches and coasts, coral reefs, riverscapes, peat uplands, and other wetland landscapes are also major features of tourism destinations, whilst river systems often define towns and cities, and artificial wetlands can provide functional benefits but also aesthetic focal points in development projects.

Aesthetic appreciation of wetlands in the built environment makes a potentially significant contribution to their overall societal value. There is rising awareness of the value of wetland landscapes and services in these urban settings. Sustainable urban drainage systems (SuDS which are also known as 'source control' in the US), other forms of "green infrastructure" and WSUD (water-sensitive urban design) are examples of planning and engineering design approaches that integrate water management with wide sets of benefits, including landscape aesthetics and recreational appeal. Therefore, improving public support for wetlands and their services in both urban and other landscapes is becoming increasingly important. Aesthetic appreciation can be a major contributor to the value and public acceptability of wetland sites and landscapes. One indicator of how aesthetic appreciation manifests in urban wetlands is reflected in how they can enhance values in urban areas. Hedonic pricing methods indicate that the value of real estate closer to water and green spaces may be around 8% higher than equivalent properties in dense developments (CABE 2005).

However, there are conflicts between inspirational qualities ascribed to wetlands and the negative connotations that people may place upon them. For example, some wetland landscapes are regarded by lay observers as "wastelands" or places where diseases can proliferate (Purseglove 1988). Yet, even where these negative connotations persist, wetland landscapes can still serve as inspiration to art and literature, as for example in the wetland-dwelling character Gollum in J.R.R. Tolkein's *Lord of the Rings* (Tolkein 1954) and the threatening swamplands that played host to Sir Arthur Conan-Doyle's *Hound of the Baskervilles* (Conan-Doyle 1902). Equally, American blues music, emanating from the wetlands of the Mississippi Delta, have been attributed affiliations to the devil, part of which may be attributed to the isolation of the wetland landscape of the Delta.

Public interpretation of natural wetlands should not therefore be regarded as uniform or insignificant in garnering support for their conservation and positive management. Consideration of public understanding and valuation is also important in engendering support for wetland construction and restoration schemes, including agreement to necessary investment for their design, implementation and ongoing presence, and maintenance in the built environment. Landscape architects can play an important role in multidisciplinary wetland projects, synthesizing technical information derived from professionals such as engineers, geologists, and wetland scientists into forms more readily understood and appreciated by the public (Minich 2011). Aesthetic appreciation of wetlands is important for public acceptance, irrespective of technical performance, and can therefore play a key role in project approval and success.

Future Challenges

Public understanding of wetlands, both promotion of their inspiration qualities and reinterpretation of potential negative associations, is vital for the success of wetland conservation, recreation, and construction. Framing this in ways that are accepted and welcomed by the public and decision-makers remains a key challenge.

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The Arts and Wetlands

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Peter Howard

Contents

Abstract

Wetlands play significant roles in the arts, both in formal and informal senses, throughout the world. This brief chapter explores their many, longstanding roles within British culture. Despite the UK focus, many of the principles discussed have wider geographical and cultural relevance.

Keywords

Formal · Informal · British culture · Cultural relevance · Landscape pictures · Artistic relationships · Dreary landscapes · Somerset levels · Lincolnshire

Introduction

Wetlands play significant roles in the arts, both in formal and informal senses, throughout the world. This brief article explores their many, long-standing roles within British culture. Despite the UK focus, many of the principles discussed have wider geographical and cultural relevance.

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Shifting Artistic Portrayals of Wetland Landscapes

The history of landscape pictures, and the way in which different landscapes show different historical characteristics and highlight different artistic needs, reveals significant insights into our artistic relationships with wetlands and other landscapes. Much of the content in this analysis derives from the book *Landscapes: The Artists' Vision* (Howard 1991).

By classifying and mapping all the landscape pictures exhibited at the RA Summer Exhibition from 1759 to 1980, one could recognize the wetlands (including most obviously of the Somerset Levels of western England and the Fens of eastern England) as part of the movement towards "dreary landscapes" starting in the 1850s, but gathering pace dramatically after 1870. Millais "Chill October" is a good example, although B.W. Leader's "February Fill Dyke" of 1881 from Lincolnshire is probably the type example. In the later nineteenth century, there was an increasing recognition of the importance to the rural laborer, following the French Barbizon school, investing him with "the dignity of labor." So the poor workers were cast as heroes, most obviously as fishermen at Newlyn and other locations.

For these purposes, the constable image of happy healthy workers downing jugs of ale at lunchtime in the "Hay Wain" was not appropriate. New places had to be found. Apart from the coast, these new places included the moorlands, the open fells, the heaths, the marshes, fens, and bogs. Only against such dreary backgrounds could the "dignity of labor" be appreciated. One of the most obvious cases was the photographer P.H. Emerson in the fenland, with his posed photographs of local workers at some highly specialized trades. Notably these new places were depicted largely in winter or autumn. This type of dreary marsh, typified by the works of E.M. Wimperis, remained a regular stereotype well into the 1960s. However, this was slowly overtaken by a new style of picture in such places, much more based on wildlife rather than people. The dreary marsh became the precious wetland.

Before 1870, such wetlands were not much depicted, but they certainly occur in literature. The inhabitants of remote, damp places had a long history of being outside normal society, and even strangely different: almost web-footed. Here, the French stilt walkers of the Aquitaine marshes come to mind. They were certainly a "breed apart" and often the seat of antisocial and antigovernment ideas. The marshland people of *Great Expectations* (Dickens 1861) are a particular case, with the criminal Magwitch fitting perfectly into such a scene.

This fits a long tradition of the outlaw who may become a hero. The classic English cases are three-fold, the Arthurian legend based around Glastonbury as the island in the marsh, the much better attested account of King Alfred also in the Somerset marshes at Athelney, burning the cakes before emerging from the bog to raise his standard. Not so much later, the only effective resistance to William's invasion came from Hereward in the marshes of the Fens. The marshes are a superb defense, as the Dutch know very well, as indeed do the French in Vendee and Poitou during the Revolutionary Wars. So the draining of the marshes, best exemplified by Mussolini and the Pontine marshes, is much more than a practical, even heroic, operation but the putting down of potential insurrection. Saddam Hussein also knew that the marshes were dangerous, draining them as a weapon of war against the rebel Marsh Arabs.

The islands in the marsh thus have a cultural significance vastly greater than their area and were particularly important as holy places of retreat for monasteries. Glastonbury, Ely, and Crowland are the most obvious cases in southern England, but even little Burrow Mump is crowned with its ruined chapel (of St Michael of course). Another was Muchelney Abbey in Somerset, the village of which featured in the floods of late winter 2014, with film of the flooded road rapidly becoming symbolic, once again of a race apart. Not surprisingly, in such places verticals become of great significance and here are some of the finest church towers in the country, and other vertical features also acquire symbolic status, including, for example, windmills. Other archaeology includes the lake villages and causeways largely from the later Bronze Age.

In literature, our greatest novelist to equate with the 1870s search for the heroic worker was obviously Thomas Hardy, but unfortunately Dorset is noteworthy for its heaths rather than its bogs. Nevertheless, Hardy immortalized the ‘silver gridiron’ landscape of managed floodplain water meadows in *The Return of the Native* (Hardy 1929). In recent years, Graham Swift’s *Waterland* (Swift 1983) has attempted to do for the Fens what Hardy did for Egdon Heath, but many authors home in on the unique characteristics. Various other works touch upon facets of the wetland landscapes, including, for example, Dorothy Sayers’ book *The Nine Tailors* about the church and the floods of the Fens, and the mist, the will-o’the-wisp, and the church bells through the fog.

Wetland environments also shape sporting activities. For example, the Dutch do extraordinary things pole vaulting their canals. The people of Wells (Somerset, southern England) always tell you that Mary Rand, who in 1964 became the first-ever British female to win an Olympic gold medal in a track and field event, developed her long-jumping prowess by leaping over the Rhynes (networks of small drainage and irrigation channels) in the Somerset Levels. Nowadays outsiders come in and take over this sodden landscape for the emerging sport of Bog Snorkelling. Bog Snorkelling is often practiced most observantly on International Bog Day (which fell on 27th July in 2014).

These diverse examples illustrate that that wetlands require the same treatment and increasing respect as was given to the coast, for example, in Alain Corbin and Jocelyn Phelps’ book *The Lure of the Sea* (Corbin and Phelps 1995), therefore warranting considerably more research and public appreciation.

Conclusions

British wetlands have inspired painters and writers throughout history, variously perceived as “gloomy” or threatening places but changing in appreciation and interpretation over time to include, for example, “the dignity of labor” or the “precious wetland.” Representation of wetlands in the arts tells us much about both how they have enriched culture and how cultural attitudes to wetland have shifted over time.

Although the observations in this chapter relate to British, and mainly English, wetlands, generic principles can be observed internationally, for example, in the representation of wetland animals in Australian Aboriginal painting, wall painting of the San people of southern Africa, and diverse other artistic interpretations throughout history and all continents.

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Section XIII

Importance of Managing Wetland Supporting Services

Mark Everard

Supporting Services for Wetlands: An Overview

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Mark Everard

Contents

Abstract

Supporting services comprise ecosystem services necessary for maintenance of ecosystem integrity, functioning, and resilience and for the production of all other ecosystem services. They differ from provisioning, regulating, and cultural services in that their impacts on people are often indirect or occur over a very long time. Although not directly exploited by people, supporting services nevertheless have vital direct and indirect impacts on human wellbeing. However, because this category of services is currently largely excluded from markets, supporting services and the ecosystems that provide them are vulnerable to degradation in favor of other, more narrowly framed, services such as production of food and fiber.

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Degradation of ecosystem resilience and functioning inevitably degrades not only the wetland resource but also the many services that wetlands provide and the interests of their many beneficiaries. It is then essential that supporting services are fully recognized and included into the positive management of wetlands and other habitats, consistent with the Ramsar Convention's "wise use" concept recognizing the needs to balance the supporting benefits provided by wetland systems with the production of provisioning, regulatory, and supporting services.

Keywords
Support services · Legislation · Common law · Subsidies · Benefits · Conservation · Natural capital

Introduction

Supporting services (Supporting Services) – those ecosystem services necessary for maintenance of ecosystem integrity, functioning, and resilience and for the production of all other ecosystem services – differ from provisioning, regulating, and cultural services in that their impacts on people are often indirect or occur over a very long time (Millennium Ecosystem Assessment 2005a).

Although not directly exploited by people, supporting services nevertheless have vital direct and indirect impacts on human wellbeing. However, because this category of services is currently largely excluded from markets, supporting services and the ecosystems that provide them are vulnerable to degradation in favor of other, more narrowly framed, services such as production of food and fiber. Degradation of ecosystem resilience and functioning inevitably degrades not only the wetland resource but also the many services that wetlands provide and the interests of their many beneficiaries.

It is then essential that supporting services are fully recognized and included into the positive management of wetlands and other habitats. This is consistent with the Ramsar Convention's "wise use" concept (Wise Use Concept), recognizing the needs to balance the supporting benefits provided by wetland systems with the production of provisioning, regulatory, and supporting services, cumulatively contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation.

Why Managed Wetlands for Supporting Services is Important

Owing to their general omission from markets, ecosystem management has tended to overlook the value of supporting services provided by wetlands and other ecosystems. This is notwithstanding their substantial value to society. The Millennium Ecosystem Assessment synthesis specifically considering global Wetlands and

Water (Millennium Ecosystem Assessment 2005b) summarized many of the ways in which wetlands provide invaluable supporting services. The MA Wetlands and Water synthesis also found that different types of wetlands produce a different balance of supporting and other services, emphasizing the importance of both the diversity and location of wetland types across landscapes. Nevertheless, there has been a substantial erosion of the wetland resource in many parts of the developed and developing world.

A number of instruments have been put in pace to manage and protect different facets of wetlands and of other perceived priority habitats including: statutory legislation, spatial planning controls, common law, operational tools, international protocols, markets, and market-based instruments.

Statutory Legislation

Some of the values of wetlands which we might now term "ecosystem services" are safeguarded in statute legislation. The body of statutory instruments in which protection of wetlands feature includes, as examples, the EU Habitats Directive and the EU Birds Directive, both of which require designation of habitats important for a variety of taxa or because the habitat type has been recognized as of high conservation importance. The EU Water Framework Directive too requires water bodies and water-dependent habitats (implicitly including wetlands) to be managed to achieve Good Ecological Status (▶ Chap. 71, "European Union Water Framework Directive and Wetlands"), recognizing both their inherent value as well as the importance of the heath and functioning of ecosystems for supporting multiple dimensions of human wellbeing.

The Ramsar Convention on Wetlands (2012) endorses the importance of the "avoid-mitigate-compensate" sequence as an important tool for maintaining the ecological character of wetlands (though its benefits are not limited just to wetlands). This is taken up in Africa, for example in Burkina Faso where the EIA decree (Le President du Faso 2001) reflects the precautionary principle to prevent damage to the environment as a consequence of human activities, as well as in China where there is a range of statutory requirements to avoid and minimize impacts to forest areas in development projects (Bennett 2009). In the American neotropics, El Salvador's Law of the Environment (Asamblea Legislativa de la República de El Salvador 1998) requires an EIA following a similar sequence or hierarchy of "prevent, attenuate, compensate" for proposed projects on fragile or protected areas and in wetlands. The Australian State of Queenland's Policy for Vegetation Management Offsets (Queensland Government Department of Environment and Resource Management 2011) emphasizes that land-based offset may only be proposed where an applicant has demonstrated that the development has first avoided and minimized the impacts of the development on vegetation prior to proposing an offset.

Additional statutory safeguards apply at national level, many of them in Europe implementing EU and other legislation and protocols. However, a great deal of legacy regulation is less than explicit about precisely what features and services are being protected. Furthermore, the protection of habitat that may be functionally important yet which lies outside of protected zones may escape these conservation measures. This is significant given the importance of the diversity and location, and often the temporarily wet nature, of wetlands across landscapes.

Spatial Planning Controls

The spatial planning system is designed to address land uses, and may therefore be influential in the production of supporting and other ecosystem services across the broader landscape. However, as yet, the spatial planning system is slow to catch up with the evolving ecosystem services agenda, and in many localities (including for example the UK) has little impact on agricultural land uses which may have significant impacts on the water cycle and wetland ecosystems. Recent developments, such as the UK's 2012 National Policy Planning Framework (Department of Communities and Local Government 2012), require planners to have regard to ecosystem services in planning determinations, though this guidance is far from explicit about how this should be implemented.

Nevertheless, screening frameworks applied in planning tools such as Strategic Environmental Assessment (▶ Chap. 113, "Strategic Environmental Assessments") and Environmental Impact Assessment (▶ Chap. 112, "Environmental Impact Assessments") can, with little adjustment, be expanded to address the full set of ecosystem services categories defined by the Millennium Ecosystem Assessment as a means to make more inclusive and comprehensive the consideration of the full implications of development proposals.

Common Law

The basis of the common law is, as for statutory legislation, the protection of rights. However, it has been developed as a less formalized body of case law evolving over centuries to uphold the rights of individuals or communities potentially infringed by the actions of others. Case law relating to rights to air, water, soil, and views of undiminished quality as well as sporting or access rights demonstrable common law protection extended to what we now term ecosystem services. Many of the services provided by wetlands have been addressed under the common law, albeit generally not explicitly in those terms.

This is exemplified by the substantial body of case law protecting the quality and enjoyment of recreational fisheries (for example Carty and Payne 1998). Case law may evolve from local disputes, but may then have wider applicability across national or other jurisdictions. It has been argued that ecosystem services represent a wider framework of rights to which common law protection may be extended to address rights of broader constituencies of society, many of which may historically have been omitted from planning considerations (Everard and Appleby 2008). Taking a wider systemic perspective of wellbeing end-points may then serve to increase quanta of damages potentially claimed by broader constituencies of defendants arising from harm to ecosystems, or further justify injunctions to prohibit damaging activities.

Operational Tools

While legal and planning frameworks are necessary enablers to the making of more integrated decisions, they need in practice to be supported by pragmatic operational tools supporting day-to-day decision-making. There is at present something of a shortfall in pragmatic tools, many pre-existing tools reflecting the narrower perspectives (generally one or just a few services rather than considering ecosystem services as an interdependent system) of legacy regulations and guidance.

A notable exception and exemplar here is *eThekwini Catchments: A Strategic Tool for Management* (Diederichs et al. 2002), which uses an ecosystem services evaluation of the major river systems serving the municipality of eThekwini, on the coast of KwaZulu-Natal in South Africa, as a basis for supporting transparent decisions about urban development planning. City planners traditionally consider limiting factors to economic growth including industrial capacity and inward investment, housing provision, amenity areas to enhance quality of life, and so forth. However, this analysis all too often overlooks the more fundamental ecosystem resources – access to water, green space, air quality, and general "liveability" features – that might ultimately constrain the value of urban expansion. The services provided by rivers and other aquatic systems serving cities fall into this latter category, in terms not only of the provision of water in adequate quantity and quality but also the capacity of river corridors to regulate flooding, dilute waste, produce food, purify air, and provide amenity and other services besides. Overuse of river corridors converging on the city of Durban was recognized as potentially limiting further development, the *eThekwini Catchments: A Strategic Tool for Management* coding ecosystem services capacity under a "traffic light" color system: red for over-exploited; orange for at capacity; and green for excess capacity. This provides an intuitive and transparent basis for decision support, making it clear, for example, that development proposals likely to increase impermeable surfaces in a sub-catchment already "red" with respect to flood management will be refused due to the imposition of greater flood risk upon resident communities. The transparency of this approach is also a stimulus for innovation in design by development proponents. This eThekwini planning tool remains very much in a minority, though it does illustrate the relative simplicity of incorporating a multiple ecosystem services perspective into practical decision-support tools.

International Protocols

In addition to statutory protection, a number of international protocols also extend protection to aspects of the wetland resource and other habitats and biota. These are diverse but significantly include the Ramsar Convention and the Convention on

Biological Diversity, with additional instruments such as CITES and the BERN Convention and Bonn Convention also adding protection for scheduled wetland taxa.

Some of these multinational protocols are statutory, creating obligations on signatory territories which generally results in modification of national legislation, while others are less formal though scrutiny and potential disapprobation by the international community gives them near-statutory weight.

Markets

Markets too exert a significant, indeed sometimes an overwhelming influence over the management of wetlands and other natural resources. However, markets have routinely failed to address the systemic nature of ecosystems and their interactions with people.

The Millennium Ecosystem Assessment Wetlands and Water synthesis (2005b) concluded that agriculture was the foremost pressure leading to the degradation of wetlands worldwide, and this is driven by narrow market forces that reward maximization of a few selected provisioning services. An unintended consequence of this is oversight of implications for nontargeted ecosystem services and the vitality of the ecosystems that produce them. This has led in turn to substantial degradation of wetland quality and quantity, in particular the supporting and other nontraded services.

This is a classic illustration of the "tragedy of the commons" (Hardin, 1968) in operation, wherein private benefits from ecosystem degradation are reaped at net cost to benefits that are shared with a broad public.

Market-Based Instruments

Acknowledging distortions in the market, a range of market-based instruments have been developed to provide protection for valued ecological and social facets of the environment.

A range of market-based instruments, including for example taxation or incentive payments, respectively dissuade or encourage measures affecting wetlands and other resources. For example, many nations have well-developed agrienvironment schemes, often put in place to favor positive management as a means to implement statutory protection. Wetland protection may be enhanced under these measures, though very often protection is narrowly bounded to protected areas only, effectively sanctioning erosion of the nonprotected wetland resource. Even within protected areas, payments may be targeted on an area basis that may overlook localized wetland characteristics within landscapes disproportionately important for particular ecosystem functions and services such as nutrient cycling, soil formation, and provision of microhabitats essential for particular species or their life stages. Thus, many of these dissuading and incentive measures are poorly targeted both at the diversity and precise location of wetlands across landscapes, and also the precise mix of ecosystem services that they provide.

Novel market mechanisms including "payments for ecosystem services" (PES ► Chap. 131, "Payments for Ecosystem Services") are seeking to bring more ecosystem services into the market. This builds upon a range of other market-based instruments, including, for example, agrienvironment subsidies that reward extensification and a range of other land use practices protective of desirable biodiversity and other beneficial features of the environment. However, in practice, most agrienvironment subsidies are only poorly targeted at specific ecosystem services, and supporting services other than some localized provision for "habitat for wildlife" are largely overlooked.

Fragmented Protection

This broad range of statutory and common law, market instrument, pan-national protocol, and pragmatic tools that implement them represents a range of measures potentially protective of different facets of wetlands and the services they provide. However, the piecemeal nature of their evolution, often to address narrow acute problems as they manifested including, for example, rare species and habitat protection or specific forms of pollution, mean that the protective measures are fragmented. Furthermore, many measures, such as various statutory protection measures affecting wetlands, are only poorly targeted.

Many also embody a historic perspective of nature conservation which, in a narrow market that values disproportionately only a few selected provisioning services, tends to posit conservation of natural assets as a constraint on economic and other forms of development. Modern conceptions of ecosystem services recognize the functions of wetlands and other habitats as important "natural capital" which, far from competing with economic progress and societal wellbeing, is the fundamental resource underpinning it.

Therefore, the fragmented legacy of potentially protective measures is today no longer fit for the purpose of protecting the wetlands resource and recognizing its multiple values to society in an integrated way. Supporting services in particular, furthest from immediate exploitation by people, tend to become net victims of the short-sighted management and practice permitted by this fragmented policy environment.

Expanding Protection of Wetland Supporting Services

The integrity, functioning, and resilience of wetlands and other habitats, represented by their supporting services, are a particular priority for protection if these ecosystems are to provide habitat for nature and maintain their many contributions to human wellbeing. The existing spectrum of statutory and common law, spatial planning, international protocol, market-based and other measures potentially protective of wetlands and their values to society is broadly compatible with this aim, albeit at present structured in too fragmented a way.

It is therefore a priority to expand the systemic reach of all instruments better to take an integrated approach protective of wetlands and their many beneficial services. Undoubtedly, some new legislation and economic instruments may be required, framed on a systemic rather than, as at present, largely addressing single or a few benefits in isolation. However, considerable progress can be made by requiring, through guidance supporting application of the legislation, that implementation should be framed in a systemic context. For example, successive revisions of the EU Common Agricultural Policy (CAP) have progressively shifted subsidies from outputs of farmed produce towards different facets of environmental protection. This can be extended ideally to address the full gamut of ecosystem services, critically including the supporting services underpinning ecosystem functioning and resilience. However, national-scale guidelines on the targeting of support payments under existing CAP rules could be broadened to better target specific wetland types and wetland processes within mixed landscapes that are important for the production of supporting and other desirable services, in preference to the current poorly targeted area-based payments that dilute the benefits of this public investment. Thereby, the link from public taxation to clearly-articulated public benefits, and hence value for money, is thus made more explicit. Similar arguments could be made for the inclusion of ecosystem service-based screening in the implementation of planning measures such as Strategic Environmental Assessment (SEA) and Environmental Impact Assessment (EIA).

As noted above, research is already asking questions about the potential extension of the common law to address a broader range of potential beneficiaries of ecosystem services, for example strengthening the case for injunctions averting harm to wetlands or alternative expanding the quantum of damages where harm has occurred. Apart from adding protection to broader facets of the natural environment and its beneficial services, this also represents a more equitable approach reflecting the interests of more ecosystem service beneficiaries, many of whom have been formerly excluded from consideration. Novel market-based instruments such as PES are also bringing more ecosystem services into the mainstream of benefit protection. However, difficulties arise for both common law remedies and marketing of new services in that supporting services are rarely directly exploited, hence it is neither easy to make a case for which beneficiaries are potentially compromised or who is willing to pay for service protection. It is here that the state generally intervenes, where the case for habitat and service protection is overwhelming, whether in the form of direct legislation (as in the case of designation of threatened habitat or access to urban green spaces) or redirection of public funds into targeted investments (as in the case of agrienvironment subsidies).

Future Challenges

It is essential to protect the diversity of wetlands occurring across landscapes, rather than merely isolated designated wetland systems, as well as wetland processes across the landscape that are essential for ensuring supporting services continue to be produced for the benefit of all in society, including future generations.

However, this does represent a significant culture change. Part of this entails redefinition in the public mind as well as that of regulators and other governance bodies that wetland conservation and the protection of supporting and other services is not a "strait jacket" on development but an investment in resources essential to secure human wellbeing including further economic opportunity.

The inherited "siloing" of policy areas also presents an obstacle, to be overcome through recognition and internalization of the contribution of wetland supporting services to all policy areas. Taking a systemic view of the value of wetlands and other habitats entails recognition of the benefits of their supporting and other services to air quality, flood management, fishery enhancement, wildlife protection, accessible low-carbon transport routes through urban environments, education, health and other facets of government interest, internalizing those benefits not merely within nature conservation policy but across all beneficiary policy areas. The policy and economic environment is as yet immature with respect to this systemic connection from supporting natural resources to human wellbeing end-points, illustrating the magnitude of this challenge for public education, economic tools, and the vision of government.

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Nutrient Cycling in Wetlands

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Mark Everard

Contents

Abstract

Wetlands play important roles in nutrient cycling, transforming and changing the mobility and biological availability of growth-promoting and, when in deficit, growth-limiting chemical substances. Nutrient cycling is a supporting ecosystem services. The supporting services are perhaps the most overlooked of all ecosystem services, relating largely to processes within ecosystems – also including soil formation, habitat for wildlife, photosynthetic productivity and oxygen generation, and water recycling – that though not directly consumed are fundamental for the resilience of ecosystems and their capacity to produce other, more directly consumed services.

Enrichment of ecosystems by mined phosphorus, remobilizing phosphorus sequestered by long-term processes, is a pernicious problem resulting in eutrophication of water and soils. Agriculture is a major source of diffuse nutrient pollution globally, representing a particular hazard to wetlands and their ecosystem services. However, wetland systems are effective at attenuating phosphorus, and this supporting service may potentially be exploited not merely to reduce nutrient loads from wastewater and diffuse run-off but also for nutrient recovery.

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Keywords
Nutrient · Cycling · Phosphorus · Nitrogen · Silicon · Supporting service · Eutrophication · Nitrification · Denitrification

Introduction

Supporting services are perhaps the most overlooked of all ecosystem services. Supporting services relate largely to processes within ecosystems – including soil formation, habitat for wildlife, water and nutrient recycling – that though not directly consumed are fundamental for the resilience of ecosystems and their capacity to produce other, more directly consumed services. Wetlands play important roles in nutrient cycling.

Wetlands and the Phosphorus Cycle

The natural phosphorus cycle comprises in essence two interlocking phosphorus cycles: one operating at geological scale and the other at ecosystem scale (Everard 2001). Over geological time-scales (millions of years), there is a net movement of phosphorus from land to the seas and back again. Weathering and erosion from rocks results in the presence of phosphorus in terrestrial and aquatic systems, often in limiting concentrations. Once passed through ecosystem-scale cycles, phosphorus entering the oceans eventually accumulates in sinks on continental shelves and in inland basins and profundal depths in the form of insoluble deposits, gradually becoming locked away into rocks (i.e., partitioning from the *biosphere* into the *lithosphere*). Tectonic movements operating over geological time-scales raise crustal plates from the sea floor and expose phosphates and other phosphorus-rich deposits to weathering on land, completing the immensely long-term cycle involving release from the *lithosphere* to the *biosphere*. Wetland systems are involved in the initial sequestration of phosphorus, though geological processes are the predominant forces affecting this cycle. Water systems then have roles in remobilization of exposed phosphorus deposits.

Ecosystem time-scales operate extremely fast compared to geological time-scales, in reality operating at a range of temporal and spatial scales (seconds to decades or centuries). All of the diverse and adaptive ecosystems of this planet operate through the cyclic reuse of substances, powered by net inputs of energy from the sun (Odum 1982, 1983). All organisms within ecosystems require phosphorus, which comprises significant proportions of molecules in living cells (nucleic acids, NADPH, ATP, phospholipids, etc.) as well as the bones and teeth of vertebrates. Plant uptake of phosphorus from soils and the water column is in turn utilized by herbivores, carnivores, shredders, decomposers, and other functional components of the ecosystem. This results in phosphorus movement between water column, sediment, and biotic compartments (Mainstone et al. 2000) and produces complex patterns of phosphorus residence times, storage, transport, export, and concentrations within each compartment of the ecosystem (Hoffmann et al. 1996). Wetland systems play important roles in these ecosystem-scale transformations, manifesting primarily as supporting services.

Enrichment of ecosystems by mined phosphorus, remobilizing phosphorus sequestered by long-term processes, is a pernicious problem resulting in eutrophication of water and soils. Agriculture is a major source of diffuse nutrient pollution globally (Millennium Ecosystem Assessment 2005a), representing a particular hazard to wetlands and their ecosystem services (Millennium Ecosystem Assessment 2005b). However, wetland systems are effective at attenuating phosphorus, and this supporting service may potentially be exploited not merely to reduce nutrient loads from wastewater and diffuse run-off but also for nutrient recovery (Everard et al. 2012).

Wetlands and the Nitrogen Cycle

Whereas environmental sinks of phosphorus are in the lithosphere, nitrogen comprises some 79% of the content of the air. However, this atmospheric nitrogen is overwhelmingly in the molecular form N_2, which is unavailable to most living organisms. However, a variety of microorganisms have the capacity to “fix” atmospheric nitrogen into biologically available forms (primarily ammonia NH_3 and nitrate NO_3^-), making it available to natural cycles. Planktonic algae in both freshwater and marine wetlands, particularly various species of blue-green algae (cyanobacteria), are particularly important as nitrogen fixers. So too are a range of soil-based microorganisms particularly those living symbiotically in the roots of leguminous plants of the family *Papilionaceae* and some other plant families. Wetland locations may therefore be particularly important locations for nitrogen fixation.

Wetlands are also important for nitrification and denitrification processes, which collectively return nitrogen to the atmosphere. Nitrification is the biological oxidation of ammonia NH_3, first forming the ammonium ion NH_4^+ and then converting this into nitrite NO_2^- and then nitrate NO_3^-. Nitrification is conducted by a range of microorganisms in aerobic conditions. Denitrification is also a microbial process in which nitrate is reduced by a large group of heterotrophic facultative anaerobic bacteria to produce molecular nitrogen N_2 which is then released into the atmosphere. Nitrification and denitrification combine to recirculate biologically available forms of nitrogen back into the atmosphere. Wetlands are important locations for this coupled process, as closely juxtaposed aerobic and anaerobic microhabitats are common.

Humanity is also now a major source of nitrogen fixation, using a variety of industrial processes to “mine” nitrogen from the air for use primarily as agricultural fertilizer. Indeed, 50% of the nitrogen fixed on Earth now stems from human origins (May 2000). Fertilizer use increased fivefold between the 1960s and 1990s (WWF 1999), compounding eutrophication problems. The supporting services of wetlands may then have a key role to play in removing excess nitrogen contamination from point source and diffuse wastewater, averting some problems associated with eutrophication.

Wetlands and Other Nutrient Cycles

Other nutrients too may be important in promoting or limiting biological activity. For example, silicon is commonly limiting to the formation of diatom blooms in freshwater and some marine ecosystems during annual cycles. The supporting services of wetland ecosystems play important roles in cycling silicon as well as iron, manganese, and a range of other elements. However, the primary focus of this brief chapter is on the two nutrients of primary importance – phosphorus and nitrogen – which have generic similarities with other cycles mediated by supporting services.

Conclusions

The supporting services of wetlands play important roles in the cycling of nutrients, as sinks, sources and locations important for nutrient transformations. Wetlands thereby play important roles in wider connected ecosystems.

Wetland supporting services may also be exploited as a low-input means of reducing the impacts of anthropogenic eutrophication, and possibly also for recovery of some nutrients (particularly phosphorus) for beneficial reuse.

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Biodiversity in Wetlands

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Mark Everard

Contents

Abstract

Ecosystem services, spanning a diversity of benefits accruing to humanity, result from the functions of wetland ecosystems. These functions in turn are generated by the interactions of both living and nonliving components of ecosystems, including interactions with adjacent terrestrial, aquatic, and atmospheric ecosystems. The role of wetland biodiversity is crucial for the production of a wide range of ecosystem services and associated socio-economic benefits. In turn, wetland biodiversity is dependent on maintenance of wetlands in good biophysical condition.

Keywords

Supporting services · Biodiversity · Geodiversity · Cycling · Degradation · Functioning

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Introduction

Ecosystem services, spanning a diversity of benefits accruing to humanity, result from the functions of wetland ecosystems. These functions in turn are generated by the interactions of both living and nonliving components of ecosystems, including interactions with adjacent terrestrial, aquatic, and atmospheric ecosystems.

Defining Biodiversity

The living elements of ecosystems are referred to as biodiversity, a contraction of the term "biological diversity." Biological diversity refers not merely to species, but is defined by the Convention on Biological Diversity (1992) as "…the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems." This and related definitions of biodiversity include genetic, species, habitat, and geographic scales thereby encompassing all living things and associated systems.

The origins of the term "biological diversity" in its current sense date back to the early 1980s, with interest in the concept elevated by publications such as "Limits to Growth" (Meadows et al. 1972) which discussed the implications of unrestricted population and economic growth on the environment. Use of the term has ranged from a focus on species richness (number of different species in a location/sample) (Lovejoy 1980) progressively to encompass increasing emphasis on ecological and genetic diversity (Norse and McManus 1980). The specific origin of the truncated term "biodiversity" is most often attributed to W.G. Rosen in 1985 (Hawksworth 1995), with more widespread use of the term promoted by the proceedings of the forum at which Rosen had presented which were published by E.O. Wilson in his book *Biodiversity* (Hamilton 2005).

The Millennium Ecosystem Assessment (2005a) collated information reflecting that, across all of the world's major habitat types, biodiversity is being lost at an alarming rate which poses a risk for the provision of ecosystem services. While the loss of many provisioning services can be felt relatively directly in terms of limitations on the supply of resources, a lesser proportion of regulatory and cultural services have such immediate economic recognition. However, the contribution of biodiversity to supporting services is perhaps the most overlooked, as supporting services refer largely to processes within ecosystems – such as soil formation, habitat for wildlife, water and nutrient recycling – that are not directly consumed, yet which are fundamental to system resilience and capacity to produce other more directly consumed services.

Biodiversity is not itself an ecosystem service. Rather, services are the beneficial human outcomes generated by ecosystems including both their biological diversity and geodiversity (nonliving elements). However, the quality of wetland biodiversity is a key indicator of wetland condition and functioning and so can be used cautiously as a surrogate for the vitality of supporting services.

Functional Wetland Biodiversity

Wetlands are among the most diverse and productive of the Earth's ecosystems, providing all forms of ecosystem services. They not only directly support many aquatic taxa but also provide food resources, a locus for cycling of energy, nutrients, carbon, and other substances, and serve as transmission pathways for species and life stages of organisms primarily adapted to other ecosystems. As no life can exist without water, the supporting services of wetlands are irreplaceable.

Not all ecosystem services depend on biological diversity. For example, the abiotic structure and geomorphological processes of river valleys and coasts are a dominant factor in natural flood attenuation and wider hydrological benefits. However, biodiversity plays important roles in most ecosystem services. This includes the more readily evident roles of large and visible flora and fauna in natural cycles, provision of shade and storm regulation, and the production of food, fiber, and natural medicines. However, the roles of the microscopic and other poorly understood biodiversity of water and soils may be of greater significance, including nutrient and other chemical transformations and cycling by soil meiofauna and microorganisms, photosynthetic production of biomass and oxygen and nitrogen fixing by planktonic algae, natural detoxification processes, and soil-forming processes. The pathways by which biodiversity produces the wide array of supporting services essential for the continuity of life and human wellbeing is poorly understood, but is of vital importance.

Concerns About the Loss of Wetland Supporting Services

The observed loss of wetlands at human hand then becomes problematic. As one example, in southern Ontario alone, 80–90% of the wetland base existing before human settlement has been lost due to drainage, filling, or habitat alteration (Ambrose 1998). This situation is replicated to varying degrees globally, with wetlands one of the planet's most vulnerable habitats. The loss of wetland on land masses is substantially driven by agricultural activities (Millennium Ecosystem Assessment 2005b).

The extent of wetland loss is at least proportional to the loss of important supporting services such as nutrient transformations, water and carbon recycling, and soil formation and fertilization, without which ecosystem integrity and resilience is lost. However, the damage inflicted by wetland degradation may be greater than loss of wetland means. Many human activities result in impacts – eutrophication and other forms of pollution, disconnection of flyways, noise and light pollution, habitat fragmentation, disruption of natural hydrology, and many more – that compromise wetland biodiversity, function, and service production.

Conclusions

Biodiversity is not itself an ecosystem service but, as an indicator of ecosystem health, can be used cautiously as a surrogate for ecosystem functions and resilience and hence capacity to produce supporting ecosystem services.

The pathways by which biodiversity produces the wide array of supporting services essential for the continuity of life and human wellbeing is poorly understood, but is of vital importance. In this regard, the microscopic and otherwise underappreciated biodiversity of wetlands and other habitats may be of disproportionately great importance.

Protection and restoration of the representative biodiversity of wetlands – in terms of species, year classes, genetic diversity, and trophic relationships – is then a priority to ensure overall system resilience and capacity to provide fundamental supporting ecosystem services.

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Soils of Wetlands and Their Ecosystem Services

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David Hogan

Contents

Abstract

Soils are an important component of wetland ecosystems and having an understanding their characteristics and properties are crucial in best wetland management. A summary is given of important soils found in wetlands, including their characteristics and classification. Understanding their hydrology and maintaining optimal water regimes are key factors in wetland management. Important ecosystem services provided by wetlands depend on their soils and highlight the importance of appropriate management for their optimization. Protection of soil quality remains a challenge for the future and there remains a pressing need for improved education about soils.

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Wetland soils · Gley soils · Peat soils · Organic matter · Soil classification · Soil wetness · Ecosystem services · Hydrological regime

What Are Soils?

Soil is the thin crust covering the earth's surface, between the bedrock (lithosphere) and the air (atmosphere), and on which all terrestrial living things (the biosphere) develop and depend. At the simplest level, it forms by weathering processes breaking down mineral material, with additions of organic matter from dead plants and animals, together with the effects of a developing ecosystem. The eventual soil type depends also on landform, climate, and time – some soils are evidently much older than others – together with the impacts of human use and management. Soil provides a home for living things and supports the growth of both natural vegetation and specific crops, thereby delivering many environmental benefits (ecosystem services). A comprehensive introduction to soil science is given by White (2006).

What Are Wetland Soils?

The main characteristic that wetlands share is their prolonged saturation with water, and soils influenced by these conditions are termed *hydromorphic soils*. Depending on features of the local geology, topography, and climate, the source of this water will vary. Additionally, soils themselves have, or develop through management, properties which control the permeability, storage, and run-off of water. Wetness may result from rising groundwater, especially in low-lying positions such as basins or river valleys, from rainfall in humid areas, from rain falling on soils of low permeability (as surface water developing a perched water table), or from overbank flooding of the river channel. Sometimes these water sources may be combined, as in the case of accumulated run-off from springs. Soils which are or have been perennially waterlogged demonstrate features which enable their identification and separation within schemes of classification. The general term used to describe a wet mineral soil is *gley*, characteristically grayish in color or mottled, while organic soils are generally referred to as peat. In tidal swamps, sulfidic sediments can form which, if subsequently drained, form acid sulfate soils with pH as low as 2 (Dent and Pons 1995). Wet soils are predominantly anaerobic, and as such have processes which deliver a range of important ecosystem services (see below).

Soil Material

Soil material may be described as being either organic or inorganic. Organic soil (or peat) will usually contain at least 35% organic matter; in the range 35–50%, this can be qualified as either sandy or loamy peat, depending on sand content, while more than 50% is simply termed peat. With increasing mineral content, categories become in turn loamy peat (down to 20–25% organic matter, depending on clay content), humose mineral soil (down to 6–10% organic matter, again depending on clay content). Soils of lower organic matter content are termed mineral (see below).

Peat soil material is described according to the type of plant remains found and their degree of decomposition, often using a modified scheme of von Post (1924). In the classification of peat soils, the degree of ripening is also being taken into account. This refers to the processes of soil formation including physical (irreversible changes caused by water loss and structure development), chemical (oxidation of organic matter, iron and sulfur), and biological (the activity or soil organisms).

Inorganic (mineral) soils are ascribed particle-size (texture) classes such as sandy loam or silty clay, as shown in familiar triangular diagrams, which express the proportions of sand, silt, and clay-sized particles in a particular soil, and have important controls on properties such as permeability and water-holding capacity. Soil materials are also described according to color and stoniness, together with other features described below in more detail.

Soil Formation

Soil-forming processes begin to take place as soon as suitable materials become available on the surface of the Earth. The rate and degree of development depends on the properties of that substrate (known as parent material) and environmental conditions operating, including temperature regime, availability of moisture, and consequent biological activity. The effects of temperature can be seen in the higher rates of soil formation in the tropics compared with temperate or colder zones (Thornthwaite 1931). The age of soils also varies considerably; those found in temperate regions, particularly in the Northern Hemisphere, are predominantly post-glacial, while many ancient land surfaces in the tropics have deeply weathered soils initiated as early as the tertiary geological period (40–60 million years ago). In the case of wetlands, soil formation operates under predominantly saturated conditions. Peat soils form in organic material accumulated where the rate of decomposition is slower than that of accumulation. Where conditions remain permanently waterlogged, decomposition is largely precluded by anoxic/reduced conditions limiting the activity of organisms responsible for decay. Peat can accumulate at varying rates: Walker (1970) gives a range 21–60 cm per 1,000 years for the British Isles. Deposition of alluvial material on floodplains also varies considerably depending on catchment factors, and soil formation may not have become very well developed if new material is frequently being added. Like wetlands themselves, soils can take a long time to form, though almost invariably their damage or destruction can be much more rapid, particularly in the case of peat soils. Consequently protection of the soil should be an integral part of wetland conservation and play a key role in wetland management.

Soil Classification

Soils are described by a sequence of layers (horizons) arranged in a vertical sequence called a profile. Soil classification depends on the recognition of conceptual classes with diagnostic horizons identified by key properties. Many systems of classification have been developed for a variety of uses, from local to global scales. Schemes differ in the emphasis given to certain profile features. The term 'gley' is often applied in the classification of anaerobic mineral soils by using categories and descriptors such as gleysol, gleyed, or gleyic.

Most soil classifications are hierarchical systems in which classes are progressively subdivided according to presence or absence of specific identifiable soil profile characteristics: high-level broad classes are defined by important soil properties such as organic matter content or shallowness; low-level classes such as the soil series, used for many surveying purposes, often use narrow classes in keeping with local features such as parent material type or texture.

Two of the most important schemes, used on a global basis, are those developed by the FAO (World Reference Base for Soil Resources – FAO 1998) and in the USA (Soil Taxonomy – Soil Survey Staff 1999). The FAO system subdivides 30 groups successively down to over 200 subunits, while soil taxonomy has 10 orders and over 10,000 series. Small countries have appropriately fewer categories (e.g., England and Wales (Avery 1980) with 10 major groups and 1,080 soil series).

In the FAO system, these are the main soil classes associated with waterlogged mineral materials*: fluvisols* are formed in alluvial deposits; *gleysols* are wet mineral soils characterized by grayish and/or mottled colors; histosols are organic soils in which at least half of the upper 80 cm qualifies as peat. In addition, some other categories are likely to be found in wetlands, in particular *planosols* which are subject to seasonal waterlogging resulting from a heavier-textured impermeable subsoil.

In Soil Taxonomy, young soils in unconsolidated materials with poorly developed profiles comprising just a topsoil are in the order of Entisols, with wet soils in the suborders of *fluvents* (permeable soils in river alluvium but susceptible to flooding) and *aquents* (wet floodplain soils receiving constant additions of alluvium), and Inceptisols, suborder *aquepts* (wet or artificially drained gleys). Mollisols (characterized by dark topsoils) include the suborder *aquolls*, which have a high groundwater table. Other gleys are included in base-rich (high fertility) Alfisols, the suborder *aqualfs* found bordering river systems and base-poor (low fertility) Ultisols, the suborder *aquults*, occur in wet areas on coastal plains.

Describing Soils

Soils can be described in-situ in a section or a purpose-dug pit, or from a log sample taken by a hand auger or mechanized corer. Horizons are coded sequentially from the surface using letters A (and/or F, H, O for organic), E, B, C, and R, each with one or more lower-case suffixes (Ah, Btg etc). A variety of handbooks are available to guide the description of key properties, in particular:

texture: the relative proportions of sand, silt and clay giving the particle-size class
color: Munsell color charts are used to describe the main and subsidiary colors including mottles, mineral deposits, infilling of pores and channels
stoniness: content, size, shape, lithology of stones: many wetlands have low stone content due to their development from organic or alluvial materials
structure: the shape and size of structures (peds) and the degree to which they have developed. Soil structure develops from the action of weather (wetting and drying, shrinking and swelling) and of soil biota (roots, earthworm activity, microbes). Many wetlands have not experienced sufficient drying phases to enable structure to develop.
organic matter: plays an important role in nutrient cycling and fertility, and in structure development

In addition, laboratory analysis may be required to confirm properties described in the field (such as texture and organic matter content), or to determine aspects of soil chemistry (e.g., lime or fertilizer requirements for agriculture or detection of heavy metal pollution); soil physical measurements can also be undertaken to determine properties such as water retention and available water capacity. Clay mineralogy can indicate potential fertility, shrink-swell capacity, or the source of parent material.

Reasons for Soil Wetness

Soil wetness can be the result of various factors. It can be due to entirely high rainfall, as in peat bogs, or some combination of groundwater, run-off, and flooding, providing a source of minerals such as in fens and marshes. Understanding the mechanisms of hydrological support for a wetland is crucial in providing effective management. In coastal areas, freshwater marshes give way to brackish and marine systems and the degree of soil ripening declines as opportunities for desiccation of the soil become more limited. Soil maps give an indication of where wetlands are or more often used to exist. Since prehistory, wetlands have been reclaimed by human endeavor and today may not resemble wetland ecosystems at all. But the soil in its appearance will often retain some record of previous conditions, especially in the case of mineral soils, and which may offer the opportunity for reinstatement given the right circumstances. Equivalent conditions for organic soils are less likely as organic material disappears through the processes of wastage following drainage.

Soil Ecosystem Services

The framework for ecosystem services, derived under the Millenium Ecosystem Assessment (MEA 2005), indicates the wide range of benefits available for human well-being from natural ecosystems. Soils are a key component in the functioning of these systems to deliver goods and services of environmental, social, and economic benefit, all of which are provided by soils of wetland systems, their delivery optimized by appropriate management.

Providing food, fuel and fiber: soil provides both anchorage and nutrients for plants found in ecosystems of wetlands; vegetation can also provide a useful means of removing pollutants.

Storing carbon and controlling climate change interactions: saturated soils store carbon from undecomposed plant material precluding its release to form carbon dioxide.

Buffering pollution: in the presence of carbon, anaerobic soils perform denitrification converting nitrate, often derived from fertilizers in run-off from the wider catchment, into nitrous oxide and nitrogen gas, which are given off to the atmosphere.

Regulating water storage and flows, and consequent flood risk: soils are a key component of the wetland ecosystem, supporting plant communities; the capacity for water storage within wetland soils is limited to those periods when soil pores may not be fully charged with water, though rough soil surfaces can provide a temporary store for standing water.

Supporting biodiversity: soils provide the habitat for a host of important fauna responsible for nutrient and organic matter cycling and processing.

Supporting cultural heritage: wetland soils and sediments, particularly peat, preserve important information for the paleoenvironmental and archeological record.

Wetland Soil Management

A key factor in managing wetlands is the protection of their soils and the ecosystem services they are able to provide. In some cases, wetlands can be regarded as natural pristine systems, largely unaffected by human impact, but in most cases some form of impact has or continues to occur. Indeed there are many examples of wetlands created and maintained by traditional forms of management, which includes drainage regulation. Water levels can be adjusted on a seasonal basis, perhaps allowing them to lower during the summer months to enable grazing and mowing to take place. In the case of peat soils, provided this is restricted to shallow depth and for a limited period, then minimal loss through shrinkage, compression, and oxidation is likely to take place, though more intensive management and especially drainage brings about steady peat loss. Similarly but more rapidly, peat is lost directly through cutting for fuel or horticultural products; subsequent restoration of peat depends upon the establishment of appropriate vegetation (usually bog moss) and the maintenance of suitable water regime and quality. The wastage of peat containing organic archeology means that the remains together with any additional paleoenvironmental evidence is lost forever, even if restoration of the wetland subsequently takes place.

Acid sulfate soils commonly occupying coastal plains have been seen as well suited to irrigated agriculture. However drainage produces acidification, affecting yields, and frequent abandonment of fields, as experienced in South East Asia. There are many examples of where leaving this kind of land as undrained wetland is a more sustainable option.

Future Challenges

Global climate change offers major challenges to wetlands and their soils as to other ecosystems. Risks of changes in hydrological regime and water quality are major issues. The future existence of peat soils depends on maintaining a permanently wet anaerobic environment; desiccation leads to peat loss (wastage) through shrinkage, compression, and decomposition by oxidation. A shrinkage rate of 18 cm per year was recorded at Holme Fen, Cambridgeshire, UK, between 1850 and 1860, as a result of land drainage in the area (Hutchinson 1980). Changes in river flow dynamics alter the delivery of sediment, the parent material of soils occurring in riverine, estuarine, and coastal wetlands. And that sediment can contain pollutants, the source of which can be extremely distant locations within a catchment.

In common with dryland areas, protecting the quality of the soil resource is a major global issue. With regard to soil, the maxim *out of sight, out of mind* rings true, and it is a major challenge to raise awareness about the importance of soil and the ways in which it functions to deliver important ecosystem services and to promote understanding of appropriate management.

While the environmental, economic, and social importance of other environmental components (air, water, ecosystems) are generally recognized, that of soil remains largely neglected. There remains a pressing need for the development of soil protection policies at both national and international levels to provide the necessary framework for effective protection and management.

For policy development in these areas to be effectively implemented, soil scientists will be required to play a more prominent role in developing soil information systems, interpreting data, advising decision-makers, and providing training and advice about soil issues at a range of scales. Education in soil science, as part of a wider understanding of wetlands, will need to expand substantially in order to provide the necessary expertise required to replace that lost through natural wastage.

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Supporting Services: A Summary

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Mark Everard

Contents

Abstract

Supporting services comprise services internal to ecosystems, maintaining their integrity, functioning, and resilience including their capacity to produce other more directly exploited ecosystem services (provisioning, regulatory, and cultural services). They relate to processes and functions within ecosystems which, though of fundamental importance, generally have no associated market values and have historically been overlooked in economic analyses and management. Recognizing, valuing (generally in nonmonetary terms), and incorporating these vital services into wetland management are vital for system integrity and health.

Keywords

Supporting services · Photosynthesis · Soil formation · Habitat · Resilience · Functioning · Cycling · Integrity

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Introduction

A more extensive definition of "supporting services" is provided in a linked chapter in this Wetland Encyclopedia. In summary, supporting services comprise those ecosystem services that ". . .are necessary for the production of all other ecosystem services" (provisioning, regulatory, and cultural services) from which they differ ". . .in that their impacts on people are either indirect or occur over a very long time. . ." (Millennium Ecosystem Assessment 2005a). As such, they relate to processes and functions within ecosystems that maintain their integrity, functioning, and resilience and which, though of fundamental importance, are most readily overlooked in economic analyses.

Wetland systems provide a wealth of ecosystem services, including many supporting services. The Millennium Ecosystem Assessment (2005b) states that "Supporting and regulating services (such as nutrient cycling) are critical to sustaining vital ecosystem functions that deliver many benefits to people. The delivery of fresh water is a particularly important service both directly and indirectly," emphasizing that different types of wetlands produce different balances of supporting services.

Production of Supporting Services by Wetlands

The Millennium Ecosystem Assessment (2005a) classification of supporting ecosystem includes: soil formation, primary production, nutrient cycling, water recycling, photosynthesis (production of atmospheric oxygen), and provision of habitat. The contributions of wetlands to these services are outlined below.

Soil Formation

Sediment accretion processes and storage of organic matter in wetland systems can make them important sites for soil formation. Peatland and wetland forest systems in particular can accrete substantial volumes of partially decayed organic matter, while mudflats and floodplains can store substantial volumes. There are clear interactions here with other services, for example, with peatland ecosystems also comprising the planet's most efficient carbon sink (Hugron et al. 2013). Also, as soil loss is one of the most pressing challenges facing a growing global population for which food security and climate stability are creating major pressures (FAO 2011), the capacity of ecosystems to regenerate soil quantity and quality may be of disproportionately high importance.

Primary Production

Wetlands are among the world's most productive broad habitat types (Millennium Ecosystem Assessment 2005b), providing primary productivity supporting marine, freshwater, and connected terrestrial and atmospheric habitats. Retaining this productivity is of significance, including both overall capacity as well as different types of products supporting a multiplicity of types of organic compounds. Substantial global appropriation of productivity for human uses – about 36% of the Earth's bioproductive surface is "…entirely dominated by man" (Hannah et al. 1994) – may be efficient in terms of providing commodities for humanity. However, maintenance of natural primary production supporting resilient food webs remains of high importance, as does the potential of productive ecosystems to provide a diversity of linked supporting and other services that are not the priority focus of contemporary intensive agricultural, aquaculture, and other human uses. The strategic roles that wetland systems play in landscapes and waterscapes may be of substantial importance in maintaining primary productivity as well as a range of linked ecosystem services.

Nutrient Cycling

Wetland systems play significant roles in nutrient transformations, as reviewed in far greater detail in the Wetland Encyclopedia ▶ Chap. 200, "Nutrient Cycling in Wetlands." Retention, transformation, and cycling of nutrient substances is important for the functioning of wider landscapes and waterscapes, including the attenuation of pollutants from agricultural (see ▶ Chap. 180, "Wetlands in the Management of Diffuse Agricultural Run-Off") and other sources.

Water Recycling

Evaporation from wetlands and the recapture of aerial moisture by complex vegetation in wetland can play a major role in the retention of water in tight cycles within landscapes. Evidence of this can be observed visually in the recapture of moisture as mists in swamps and forests, and in green valleys in otherwise semiarid and arid landscapes where complex vegetation intercepts and retains water. These processes may be local but are also substantial at wider scales up to the continental, as for example, in the importance of forest systems in global water cycles. Examples include the extensive rainforests of the Congo basin in which as much as 90% of the rainfall is generated by the forest itself (Everard 2015) and the Amazonian rainforest which recycles up to a third of its moisture content mitigating the effects of drought and also affecting adjacent ecosystems (Harper et al. 2014). Water cycling in urban settings by wetlands and other habitat types can also play significant roles in microclimate regulation (a contribution of this supporting service to a regulatory service) that may have benefits for urban health and comfort.

Photosynthesis (Production of Atmospheric Oxygen)

Photosynthetic processes responsible for primary productivity also generate oxygen, the replenishment of which is clearly important for human and ecosystem wellbeing.

Although oxygen availability is not (at least yet) limiting in open Earth systems, the significance of this supporting service in regenerating the resource should not be overlooked. As sites of high primary productivity, wetlands may be of disproportionately high significance in this regard. While the role of large wetland vegetation in photosynthesis may be better appreciated, the role of microscopic biota may be appreciable, if commonly overlooked; between 50% and 85% of the oxygen content in the air we breathe is produced by oceanic phytoplankton (EarthSky 2014).

Provision of Habitat

Wetlands are important for their role in supporting a wide diversity of plants, animals, and microorganisms, and this biodiversity in turn is important for the multiple ecosystem functions it performs including production of diverse ecosystem services. If habitat degrades, so too does associated biota, functions, services, and benefits to humanity, emphasizing the importance of wetland conservation. Wetland biodiversity is also, of course, the primary indicator of the vitality of wetland habitat and the processes and benefits it supports. The provision of habitat for wildlife and role of wetland biodiversity are considered in far greater detail in the Wetland Encyclopedia ► Chap. 199, "Supporting Services for Wetlands: An Overview."

Interdependencies

It is clear from the above review that there are close interdependencies not only between all supporting services but also with all other ecosystem services. A useful analogy for the supporting services is that of an iceberg, in which the more obviously visible and exploited services (provisioning, cultural, and many regulatory) represent only a small proportion (around 10%) of the iceberg visible from the surface. However, this exploitable fraction rests upon an often invisible 90% of supporting surface below the water, maintaining the integrity, functioning, and resilience of the whole interconnected ecosystem. For this reason, retaining wetlands and their supporting services is of substantial importance in maintaining the viability and durability of wider landscapes and waterscapes, and the benefits they provide to humanity.

Challenges

Recognition of supporting services remains a primary challenge, as they often elude broader recognition and valuation due to their role in underpinning ecosystem functioning and service production rather than being directly consumed. However, loss of system integrity, resilience, and productive capacity has substantial consequences for the viability of all other services and the diverse socioeconomic interests dependent upon them.

Retaining a heterogeneity of wetland types across landscapes is also important, recognizing that different wetlands provide different spectrums of ecosystem services, including supporting services.

The Ramsar Convention's "wise use" concept (see the Wetland Encyclopedia ► Chap. 55, "Wise Use Concept of the Ramsar Convention") recognizes the needs to balance the supporting benefits provided by wetland systems with the production of provisioning, regulatory, and supporting services, cumulatively contributing to multiple dimensions of human wellbeing and ongoing resilience, including their contribution to poverty alleviation.

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Section XIV

Wetland Delineation and Classification

Philippe Gerbeaux

Wetland Classification: Overview

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Philippe Gerbeaux, C. Max Finlayson, and Anne A. van Dam

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Abstract

The term "wetland" groups together a range of largely aquatic habitats that usually have a number of common features, such as the presence of specific vegetation, soils, and water regimes, including the occurrence of continuous, seasonal, or periodic standing water or saturated soils. Most approaches used around the world to classify wetlands are referred to as "classification systems". The wetland definition and typology used by the Ramsar Convention on Wetlands

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is an example which includes a wider range of wetland types than included by many others. The Ramsar definition was given in the text of the Convention in 1971 and is one of several classification approaches used around the world. The evolution of wetland classification systems is outlined and a number of other classification systems currently in use around the world are briefly reviewed and discussed in this chapter.

Introduction

The term "wetland" groups together a range of largely aquatic habitats that usually have a number of common features, such as the presence of specific vegetation, soils, and water regimes, including the occurrence of continuous, seasonal, or periodic standing water or saturated soils. There is often confusion and sometimes controversy over whether a specific habitat is considered a wetland. The wetland definition and typology used by the Ramsar Convention on Wetlands is an example which includes a wider range of wetland types than included by many others. The Ramsar definition was given in the text of the Convention in 1971 and has not been changed since – areas of marsh, fen, peatland, or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt, including areas of marine water, the depth of which at low tide does not exceed 6 m. While the definition has remained unchanged the number of habitats included in the accompanying typology has been extended and now contains 42 wetland types – 12 marine/coastal, 20 inland, and 10 human-made (Ramsar Convention Secretariat 2010).

The wetland literature contains a large number of terms that have been used in many languages to describe wetlands. This is not unexpected given that wetlands have been important to people for millennia and often occupy an intermediate position between terrestrial and aquatic ecosystems and include a diverse range of habitats. The multiplicity of wetland types and terms used to describe them can make classification a difficult task. The purpose of wetland classification, as given by Finlayson and van der Valk (1995), is to standardize and define the terms being used to describe wetlands. Given the range of terms, there is a strong case for a uniform set of terms at an international level, although at a regional or local level this may not be as necessary.

An international system that avoided the use of geographically specific terms, but which enabled the inclusion of regional or national modifiers, could supplant fears about the loss of local relevance or the diversity of terms that exists (Finlayson and van der Valk 1995). The well-known Ramsar typology of wetland types (Finlayson this volume) is an example of a standardized but simple listing of wetland types that has been used to describe a range of wetlands for international purposes. The main criticisms of the typology are that it was not systematic (Semeniuk and Semeniuk 1997) and included habitats not normally considered as wetlands (e.g., caves). Attempts to replace the Ramsar typology with a more systematic classification have failed given a reluctance to move away from the current listing of wetland types and the protracted nature of discussions about a replacement. In order to overcome the commonly encountered difficulties with agreeing on a universally acceptable classification, Finlayson et al. (2002) proposed the collection of core data sets covering, for example, landform and hydrologic, ecological, and chemical parameters that could be combined as required to describe wetlands in line with local needs and linguistic preferences.

Why Classify Wetlands?

The classification of ecosystems is often presented as a necessary step for their systematic management and conservation. However, due to the complexity of gradients present (especially where land meets streams, rivers, lakes, and estuaries but also within "wet" lands), classifying wetlands, perhaps more so than with any other ecosystems, is fraught with as many difficulties as obtaining agreement on a standardized definition of wetlands (Scott and Jones 1995; Finlayson and van der Valk 1995). As such, the need for and acceptance of a particular classification system is often challenged.

Nevertheless, in order to meet management needs, organize knowledge, and reconcile the dynamic and complex realities of wetlands, it is widely accepted that wetland classification can enable a systematic grouping of sites into categories on the basis of evolutionary or structural relationships which exist between and among them (Simpson 2002). Generally, classifying wetlands is found useful for a range of reasons including supporting the following activities:

- Inventory, mapping, evaluating, and ranking sites in a consistent way (whether at national, regional, or local scale)
- Conservation and/or restoration planning (e.g., producing broad-/fine-scale sets of representative area upon which to focus protection or restoration efforts)
- Providing a framework that can easily describe the natural values, functions, and ecosystem services attached to the different categories and tailor management needs and practices
- Assessing and monitoring environmental trends (health) with indicators adapted to each (and/or all) type
- Fulfilling National and International State of the Environment reporting requirements in a consistent way
- Enhancing and simplifying information contained in geospatial databases and frameworks used in water resource planning and management, thus aiding decisions about resource/catchment management
- Raising public awareness of the diversity, values, uses, and anthropogenic effects on different wetland types
- Providing uniformity in concepts and terminology

Further, the role of classification as a basis for the language of communication between scientists and policy-makers or other interest groups should not be underestimated. A classification may permit disparate disciplines and nonspecialists to communicate more effectively and more reliably (Newson et al. 1998).

What Is a Classification System?

Kondolf (1995) describes *classification* as "the ordering of objects into groups based on common characteristics and attaching labels to the groups." Other terms are also used for describing the process of classifying. According to Bailey (1994) *classification* is the "general process of grouping entities by similarity," *taxonomy* is an empirical, mainly objective procedure of allocating cases on the basis of their measured attributes, while *typology* is conceptual, based upon a priori, subjective judgments of class definitions and boundaries (often based on expert opinion). With the increasing acquisition of detailed field data (e.g., hydrological or species data) in a number of countries, environmental classifications resembling taxonomic systems can now be generated at local and/or regional scale. They are often referred to as "*bottom-up*" approaches, while conceptual ones, based on expert opinion, are labeled "*top-down*" approaches.

Most approaches used around the world for classifying wetlands are referred to as "classification systems" (see Finlayson and van der Valk 1995) even though many would be probably better defined as *typologies* (sensu Bailey 1994). A detailed terminology and an explanation of characteristics associated with the classification system are essential to describe and define the resulting types. It guarantees a level of consistent understanding among users.

Evolution of Wetland Classification Systems

Some of the earliest efforts for classifying wetlands were for the purpose of finding wetlands that could be drained for human use. Using soil information and the potential for land utilization, many surveys were carried out worldwide, and wetlands were classified according to their potential for farming as drainable swamps (farmable), difficult swamps or wet bush flats (marginal), undrainable swamps, or difficult wetlands (unfarmable). This led to rapid wetland drainage or conversion (MEA 2005; Davidson 2014). The first environmental wetland classifications were largely introduced to bring together information in wetland inventories or directories. They often emphasized a morphological topology (a description of the situation, shape, size, and depth of the wetlands) or described the vegetation (land cover), water source, or nutrient status. They were also often centered on the desire to compare different types of wetlands in a given region, often for their value to specific taxa, such as waterfowl or fish. It is still common to find different classification systems in use for specific regions within the same country (e.g., Finlayson and van der Valk 1995; Finlayson et al. 1999).

The botanical interest in wetlands has seen the development of many classifications based on the structural and compositional features of wetland plant communities. Mires/peatlands have often been classified on the basis of their vegetation, often combined with information on their nutrient status and/or acidity. An example is the multilevel classification of mires of the former Soviet Union based on physiognomic and phytogeographical features (Masing et al. 2010). While these approaches have their place in specific contexts, increasingly, managers and scientists have expressed the need to identify and group wetland types primarily according to the abiotic features that drive their functionality (such as their hydrology and geomorphic or landform features) and secondarily according to their biotic features (such as vegetation). Hierarchical or semi-hierarchical systems have been developed to incorporate multiple datasets with the possibility that these could be adapted for specific purposes and without necessitating the collection of new data (Finlayson et al. 2002). Naiman et al. (1992) point out the following features of an enduring classification system for streams that could equally apply to further wetland types: it should encompass broad spatial and temporal scales, integrate structural and functional characteristics under various disturbance regimes, convey information about the underlying mechanisms controlling the main features of the wetland, be low-cost, and reach a high level of consistent understanding among managers.

Current Classification Systems

Many local and national wetland classifications exist and more are being proposed, often incorporating local terms or definitions that are not necessarily known or accepted elsewhere. As noted above, this may not be a major problem when used locally or for specific purposes and has definite advantages for communicating with national stakeholders and managers. The example of the many approaches used to classify peatlands clearly shows the diversity and disparity that has occurred between classification approaches (Lindsay this volume). However, for comparisons and management at an international level these differences may present difficulties. At the national and international levels, it can be extremely difficult to develop a classification that is acceptable to all wetland scientists and experts even when the advantages of reducing the confusion that exists with commonly used terms, such as swamps, marshes, or lagoons.

Possibly one of the most widely applauded national classifications is that developed by Cowardin et al. (1979) for the USA which divided wetlands into systems, subsystems, classes, and subclasses, along with a set of modifiers for water regime, chemistry, and soil types. Before developing and approving this national system, wetland classification in the USA had suffered from a lack of consistency, e.g., by using wetland terms that did not have the same meaning in different states. Another change was that in the new system hydrology was the primary determinant for classifying an area as a wetland, which meant that areas without vegetation (e.g., beaches and mudflats) could be included (see Wilen and Golet this volume). More recent developments have added hydrogeomorphic descriptors (landscape position, landform, water flow path, waterbody types) to the categories of the Cowardin system, resulting in better description, categorization, and mapping of US wetlands and their functions in the landscape (see Tiner this volume). In Brazil, a national classification system was developed with the main objective of providing a scientific basis for national decision-making and policy development (see Junk et al. this volume). A key feature of wetlands in Brazil is the strong periodicity of rainfall which leaves a major part of the wetlands (often river floodplains) dry for most of the year (and therefore vulnerable to encroachment and conversion). It is hierarchical, with a first level of classification based on broad wetland system type (coastal, inland, or artificial), followed by lower levels of classification using hydrology and vegetation. The Brazilian system needs to deal with very large wetlands (like the Amazon region and the Pantanal) which enclose areas of dry land that are essential parts of these wetlands but would not be classified as wetlands under other classification systems. Another national wetland classification system is that of South Africa (Ollis et al. this volume) which is primarily a system for aquatic ecosystems of which wetlands are an integral part. The South African system first distinguishes between marine, coastal, and inland aquatic systems, followed by a more refined classification based on depth and slope, substrate type, connectivity, and biogeography. Within the inland systems, a classification based on hydrogeomorphic units is used (see Ollis et al. this volume). In the first classification system in India in the early 1990s, wetlands were grouped into saline and freshwater types and then further distinguished based on hydrological regime (permanent or seasonal flooding) and vegetation types (herbaceous or woody vegetation, dominant species). Later on, this system was modified to classify wetlands in a limited number of predefined classes (see Gopal, this volume).

Other approaches have been developed based on an initial description of the hydrological and geomorphic settings of wetlands. Brinson's (1993, 1996) hydrogeomorphic system (see Semeniuk and Semeniuk's entry on the hydrogeomorphic classification system, this volume) is based on geomorphology (topographical location), water source (precipitation, surface water, or groundwater), and water dynamics (direction and strength of flow). The principle of the system is that the geomorphic, physical, and chemical characteristics (and not the species distribution itself) provide an explanation of the ecological character and species distribution of the area. Brinson's system resulted in seven main classes of wetlands: riverine, depression, slope, mineral soil flats, organic soil flats, estuarine fringe, and lacustrine fringe wetlands.

Further development along these lines resulted in the geomorphic-hydrologic classification system of Semeniuk and Semeniuk (1987, 1995; see also Semeniuk and Semeniuk's ▶ Chap. 207, "Wetland Classification: Geomorphic-Hydrologic

System," this volume). This system aims at a global classification of wetlands based on the two essential features of wetlands: land (landform or geomorphology) and water (hydrology). Landform is the result of regional and local processes like, e.g., weathering, erosion, barring, and deepening while water processes include, e.g., inundation, water logging, and drying. The geomorphic-hydrologic system starts by distinguishing between non-emergent (on land surfaces or in hollows/ channels) and self-emergent wetlands (rising above the land surface through deposit accretion) and then further refines the classification at the site level by looking at landform types (hills, cliffs, slopes, flats, vales, channels, and basins) and water regime (permanently inundated, seasonally inundated, intermittently inundated, seasonally waterlogged, and permanently waterlogged). The combination of landform and water determines the conditions for biota to develop. In this classification system, wetlands in different parts of the world can be classified as the same type despite differences in vegetation or other biota which may be due to biogeographic region or climate. Separate classifications and typologies for coastal systems and estuaries were developed based on geomorphological characteristics (see Semeniuk and Semeniuk's entry on coastal systems and Davidson's entry on typology of estuaries, both this volume).

Many other classification systems have drawn on the principles outlined in these classifications. In particular the Cowardin et al. (1979) classification provided a base for others to use as a model. Further interest has been expressed in using hydrological and geomorphic settings and ensuring that the resultant classification is systematic and with clear criteria to separate wetland types. A key issue when choosing or developing a classification is to clearly identify the purpose for classifying the wetlands and to ensure the chosen approach can be readily used with available information. It is likely also that future classifications will be influenced by the technology available, such as ready access to remotely sensed images, which may affect the manner in which the classification is constructed. The number of classification schemes has increased over the past decades with the hierarchical and hydrological and geomorphic approaches being widely used, often linked with information about the land cover or vegetation. At the same time, simpler typologies, such as that used by the Ramsar Convention, serve a useful and ongoing purpose.

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Wetland Delineation: Overview

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Ralph W. Tiner

Contents

Abstract

While wetlands had been drained for agricultural purposes in many regions for hundreds of years or longer, once wetlands became appreciated for the environmental services they naturally provide society (e.g., floodwater storage, water quality renovation, bank and shoreline stabilization, and provision of essential habitat for fish, aquatic invertebrates, and other animals dependent on wetlands),

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people became concerned about wetland losses. Filling and the combination of dredging and filling were particularly destructive to wetland functions and altered the picturesque view that many wetlands offered, especially wetlands along coasts and large water bodies. Public concern sparked efforts to protect wetlands through three chief means: (1) acquisition of wetlands for the establishment of wildlife refuges, management areas, or nature preserves; (2) purchase of easements on private property to set aside wetlands for conservation purposes; and (3) passage of laws to directly or indirectly protect wetlands. For these purposes, it became important to identify wetlands on the broader landscape and to be able to delineate their boundaries on the ground. Such efforts often involved the production of wetland maps and required development of field-based procedures to delineate wetlands on the ground. The former is usually done for natural resource planning and land acquisition for conservation, while the latter are prepared to identify limits of "regulated" wetlands when designing construction in or near them. Before any of these activities can commence, wetlands need to be defined in such a way that they can be mapped through remote sensing techniques and by ground surveys. This contribution provides an introduction to wetland delineation with a focus on US practices. *Wetland Indicators* offers a comprehensive examination of the topic including the rationale behind many of the properties used for delineation.

Keywords
Wetland delineation · Wetland mapping · Wetland detection · Wetland identification · Wetland classification · Wetland indicators · Hydrophytic vegetation · Hydric soils · Wetland hydrology

Introduction

While wetlands had been drained for agricultural purposes in many regions for hundreds of years or longer, once wetlands became appreciated for the environmental services they naturally provide society (e.g., floodwater storage, water quality renovation, bank and shoreline stabilization, and provision of essential habitat for fish, aquatic invertebrates, and other animals dependent on wetlands), people became concerned about wetland losses. Filling and the combination of dredging and filling were particularly destructive to wetland functions and altered the picturesque view that many wetlands offered, especially wetlands along coasts and large water bodies. Public concern sparked efforts to protect wetlands through three chief means: (1) acquisition of wetlands for the establishment of wildlife refuges, management areas, or nature preserves; (2) purchase of easements on private property to set aside wetlands for conservation purposes; and (3) passage of laws to directly or indirectly protect wetlands. For these purposes, it became important to identify wetlands on the broader landscape and to be able to delineate their boundaries on the ground. Such efforts often involved the production of wetland maps and required development of field-based procedures to delineate wetlands on the ground. The former is usually done for natural resource planning and land acquisition for conservation, while the latter are prepared to identify limits of "regulated" wetlands when designing construction in or near them. Before any of these activities can commence, wetlands need to be defined in such a way that they can be mapped through remote sensing techniques and by ground surveys. This contribution provides an introduction to wetland delineation with a focus on USA practices. "Wetland Indicators" (Tiner 2016) offers a comprehensive examination of the topic including the rationale behind many of the properties used for delineation.

Wetland Mapping

With a wetland definition in place, the next step for mapping is to develop a classification system that can be used to identify the variety of wetlands in a given region. There are two basic approaches to classification: horizontal and hierarchical. A horizontal classification would identify a certain number of types and the mapper would place wetlands into one of the specified categories. The Ramsar wetland classification essentially follows this model, with the world's wetlands placed into roughly 40 types (Ramsar Convention Bureau 1998). In contrast, the hierarchical approach is more complex in form, resembling a decision-tree where one starts with a few basic types based on general characteristics, then each main category is divided into another set of branches based on certain properties with additional branches added as different features are examined and used to separate wetlands into more unique types. The US Fish and Wildlife Service's wetland classification is an example of this approach (Cowardin et al. 1979; FGDC Wetlands Subcommittee 2013). When considering dominance type as the final level of this type of classification, thousands of types can be recognized globally. The hierarchical approach is a flexible method that allows for a more complete description of a particular wetland and permits better representation of the diversity of wetlands.

Once the classification is in place, mappers can use remote sensing techniques (e.g., aerial image interpretation or satellite image processing, or a combination of these and other analytical methods) to produce wetland maps (Tiner et al. 2015). While wetland maps were hard-copy products in the past, the advent of computers, geographic information systems (GIS), the Internet, and web-based mapping tools have made geospatial wetland data the primary product of many wetland mapping projects. These data are posted online for users to view onscreen and print custom maps through the latter tools or to download for use in a GIS environment, such as the "wetlands mapper" (U.S. Fish and Wildlife Service) and the "NWI+ web mapper" (Association of State Wetland Managers 2014).

On-the-Ground Delineation

While it might be considered a form of mapping, field-based techniques are used to survey wetlands on a particular property or to verify the results of remotely sensed wetland mapping. For this work, onsite characteristics are typically used to identify both the presence and limits of wetlands. To verify "wetland maps" derived through remote sensing, the emphasis of the field work would be to identify various unique indicators that can be used to readily separate wetlands from nonwetlands, e.g., the primary indicators method (Tiner 1993). When evaluating a parcel of land for regulatory purposes (i.e., to establish the limits of government jurisdiction) in the USA, a more in-depth examination of site characteristics is required as such action may restrict the use of private property. For this purpose, a three-factor method (analyzing vegetation and soils, and recording observations of various signs of hydrology) is frequently used to document the presence of wetland and delineate its boundaries. Elsewhere less than three factors are used for identification. For example, in temporarily and seasonally flooded wetlands, such as the turloughs common in the karst limestone dominated landscape of western Ireland, wetland boundaries can be determined by known maximum flooded area, or presence of wetland vegetation (Sheehy Skeffington et al. 2006). Turloughs are fed by rising groundwater, typical of higher winter rainfall, with water receding in the drier summer periods. Many of these sites have the characteristics of shallow lakes during the flood period and are used as pasture for low intensity grazing in the summer.

Wetland Indicators

Aerial Imagery

Aerial photographs have been traditionally used to produce wetland maps for geographic areas of varying sizes. The traditional approach involved stereoscopic interpretation where pairs of aerial photos were viewed via a stereoscope. This allowed the interpreter to see relief (three-dimensions) which aided in detecting wetlands in depressions and broad flats and also for identifying different types based on life form (e.g., separating forests from shrub thickets). Some of the indicators that interpreters look for included the following: depressions, floodplains, broad flats (coastal plains and glaciolacustrine plains) often with creeks, drainage patterns, or pockmarked with basins, flooded areas, saturated soils, and unique plant communities that display a characteristic "photo-signature" (e.g., ericaceous shrub bogs, Atlantic white cedar swamps).

Black and white panchromatic film was the first type used for wetland mapping, but was replaced by color infrared film which became widely available and permitted better discrimination of vegetation types, especially evergreen from deciduous forests. Interpretation was done largely by delineating wetlands on acetate photo-overlays with pen and ink. Cartographers would then convert the interpretations to hardcopy maps. With the development of computers and geographic information systems, and collection of digital imagery, this traditional method has been replaced by onscreen delineation where the image analyst delineates wetlands on a computer screen to create a digital geospatial data layer that is then displayed and accessible via an online GIS data viewer or "online mapper" (e.g., ESRI Inc. 2014; US Fish and Wildlife Service 2014; Association of State Wetland Managers 2014). The available digital imagery is mostly true color (e.g., Google Earth and Bing Maps). Satellites and other sensors have been used since the 1970s to collect remotely sensed data for mapping land use and land cover including wetlands (e.g., Carter et al. 1974). Recently much progress has been made and many researchers are now using remotely sensed data to map the world's wetlands given their significance in the global carbon cycle (e.g., see Tiner et al. 2015).

Field Indicators

Since the 1970s, the USA has been actively involved in regulating uses of wetlands on private and public lands through the Federal Clean Water Act. Wetlands are considered one of the nation's waters. Consequently, the US government has probably spent more time contemplating how best to identify wetlands on the ground than any other nation. The practice of wetland delineation has evolved from the early days of wetland regulation and now there is considerable agreement amongst various Federal agencies on what is and what is not a wetland. The development of an interagency manual in 1989 was instrumental in bringing agencies together to produce a unified approach to delineation (Federal Interagency Committee for Wetland Delineation 1989). Since it was the first wetland manual adopted for mandatory use across the country, it met with strong resistance from certain interest groups as the method identified more wetland than had been regulated beforehand in some regions of the country. Nonetheless it set a standard for using a more science-based approach for wetland delineation. Today regional supplements to the Corps of Engineers 1987 wetland delineation manual are the standards used to identify wetlands (e.g., Environmental Laboratory 1987; US Army Corps of Engineers 2012, 2014). The supplements contain the list of updated wetland indicators while the manual still serves as a guide to how to evaluate plant communities on a parcel of land to make wetland determinations. The following sections summarize the common indicators of wetlands used to identify wetlands. It is important to emphasize that the US. federal delineation methods focus on the identification of vegetated wetlands, as nonvegetated wetlands such as tidal mudflats and periodically exposed shorelines of inland water bodies are considered "other waters of the United States" for regulatory purposes. The methods and field indicators are intended for use throughout the year (during wet and dry seasons, and even provide guidance for extreme droughts) and that only deep snow and frozen soils should be problematic. At the latter time, only an approximation of the location of wetlands can be made by consulting aerial imagery, existing maps, and other sources or by site characteristics. Subsequent field assessment should be conducted once snow melts and soils thaw for an accurate delineation.

Indicators of Hydrophytic Vegetation

While many plant species are unique to wetlands and can be used to positively identify wetlands, there are many others that grow in both wetlands and nonwetlands

(Tiner 1991). This makes it impossible to use vegetation alone to identify all wetlands. Nonetheless many wetlands – usually the wetter ones – can be easily recognized by their vegetation, e.g., aquatic beds, marshes, swamps, fens, and bogs. Yet other wetlands including some wet meadows, bottomland forests, flatwoods, floodplains, and disturbed wetlands typically require examination of other factors, namely soil and hydrology to verify the presence of wetland.

There are many ways to interpret vegetation with respect to its "wetlandness." One could simply look at the dominant species and determine if they are species that have a wetland preference. Alternatively one could look for the most sensitive species that are always or mostly found in wetlands or plants with morphological adaptations for life in wetlands that can be readily identified in the field. Finally one could include all species in the analysis and give individual species weight based on their affinity for wetlands and their abundance in the community. In the USA, all four approaches are used to some degree for determining if a given plant community is a "positive" indicator for hydrophytic vegetation when evaluating a site for regulation under the Federal Clean Water Act.

Hydrophytic Vegetation Indicators: US Army Corps of Engineers

Hydrophytic vegetation is just one of three factors used to verify the presence of a wetland for regulatory purposes by the US Army Corps of Engineers. Their procedure for interpreting the "wetlandness" of a plant community contains four indicators. Application of the indicators is stepwise as the amount of analysis required increases as one goes down the list. The USA has reviewed the ecological distribution of plants and assigned "wetland indicator status" to all its plants: obligate (OBL) for plant species that nearly always occur in wetlands, facultative wetland (FACW) for those that usually occur in wetlands, facultative (FAC) for those that occur in both wetlands and nonwetlands, facultative upland (FACU) for plants that usually occur in nonwetlands but may occur in wetlands, and upland (UPL) for species that almost never occur in wetlands (Reed 1988; Lichvar et al. 2014). The method also involves evaluating dominant species in different strata and in usually circular plots of variable size depending on life form (see U.S. Army Corps of Engineers 2012 for details).

The first indicator is called the "rapid test" and is based on a visual assessment of the community. If all the dominant species are rated as OBL or FACW, or a combination of these two categories, the plant community is determined to be hydrophytic (i.e., the area contains plants common to wetlands). The second indicator is the "dominance test" which involves the identification of dominant species in each of typically four strata (based on life form; i.e., tree, sapling/shrub, herb, and woody vine). In this case, when >50% of the dominant species are OBL, FACW, and/or FAC species, the community is a positive indicator for hydrophytic vegetation. The remaining indicators can only be applied in cases where hydric soil indicators and sufficient wetland hydrology indicators were found, while the plant community did not pass the first two tests for hydrophytic vegetation. The third indicator utilizes a "prevalence index test" which considers all species in the community, assigns weights based on their "wetland indicator status" (i.e., OBL = 1, FACW = 2, FAC = 3, FACU = 4, and UPL = 5) and their areal cover. The following equation is used to calculate the index:

$$PI = \frac{A_{OBL} + 2A_{FACW} + 3A_{FAC} + 4A_{FACU} + 5A_{UPL}}{A_{OBL} + A_{FACW} + A_{FAC} + A_{FACU} + A_{UPL}}$$

where

PI = Prevalence index
A_{OBL} = Summed percent cover values of obligate (OBL) plant species;
A_{FACW} = Summed percent cover values of facultative wetland (FACW) plant species;
A_{FAC} = Summed percent cover values of facultative (FAC) plant species;
A_{FACU} = Summed percent cover values of facultative upland (FACU) plant species;
A_{UPL} = Summed percent cover values of upland (UPL) plant species.

At least 80% of the total vegetation cover on the plot must be of plants that have been identified to species and have an assigned wetland indicator status (including UPL). For this indicator, a plant community with a prevalence index of 3.0 or less is a positive indicator of hydrophytic vegetation. The fourth indicator – "morphological adaptations" – allows certain FACU species to be treated as FAC species provided they possess one or more of the characteristic morphological adaptations of wetland plants (e.g., shallow roots, hypertrophied lenticels, hypertrophied stems, adventitious roots, and multistemmed trunks) and that such features are present in more than 50% of the individuals in the hydric soil area and that such adaptations do not occur in the same species growing outside the hydric soil area. When this situation is true, the FACU species is treated as FAC and the dominance test or prevalence index test are recalculated. If either test is passed, the vegetation is hydrophytic for this purpose.

Indicators of Hydric Soil

Wet soils develop certain properties that make it relatively easy to separate most wetlands from nonwetlands, although soils along low gradients (e.g., areas of low topographic relief) and disturbed soils remain somewhat problematic (Tiner 2016). Prolonged flooding and/or saturation typically leads to the accumulation of organic matter. Nearly all organic soils (Histosols; i.e., muck, peat, and mucky peat), with one exception (Folists – organic soil formed under nonsaturated conditions) are associated with wetlands. These soils are comprised of the remains of plants and characterized by thick organic deposits at least 40 cm thick but may be shallower if on bedrock, for example. In their undrained condition, they are without question a highly reliable indicator of wetland. Other soils are mineral soils characterized by some combination of sand, silt, and clay. Alternate wetting for long periods and drying of these soils creates other unique properties that can also be used to separate wetlands from nonwetlands. The degree of soil saturation influences the soil color through reduction-oxidation and precipitation of various elements. Redoximorphic features (e.g., depletions and concentrations of iron) reflect varying degrees of wetness in mineral soils (Vepraskas and Craft 2015). “Hydric mineral soil” properties include but are not limited to: (1) a gleyed matrix below the A-horizon (top mineral soil layer), (2) shallow organic soil layers (roughly 20–40 cm thick; histic epipedon), (3) soils with thick dark surfaces (black to very dark brown) underlain by low chroma subsoils possessing redoximorphic features within 30 cm of the surface, and (4) sandy soils with organic coatings or redox concentrations or depletions near the surface.

The term “hydric soil” was first used in the US Fish and Wildlife Service’s wetland classification (Cowardin et al. 1979) as the presence of undrained hydric soil was recognized as an indicator of wetland. The US Department of Agriculture Natural Resources Conservation Service (NRCS) has published various documents describing field indicators of hydric soils (e.g., Vasilas et al. 2010). The indicators are periodically reviewed and updated. The regional supplements have adopted and in some cases refined these indicators for use in identifying jurisdictional wetlands. The following is a list of hydric soil indicators used in the Northcentral and Northeast Region (see the regional supplement for details; US Army Corps of Engineers 2012). The indicators are placed in four groupings: (1) all soils (regardless of texture), (2) sandy soils, (3) fine-textured soils, and (4) problematic soils. For all soils, the following are recognized as hydric soil indicators: histosol, histic epipedon, black histic, hydrogen sulfide (odor within 30 cm), stratified layers, depleted below dark surface, and thick dark surface. For sandy soils: sandy mucky mineral, sandy gleyed matrix, sandy redox, stripped matrix, dark surface, polyvalue below surface, and thin dark surface. For fine-textured soils: loamy mucky mineral, loamy gleyed matrix, depleted matrix, redox dark surface, depleted dark surface, and redox depressions. Indicators for problematic soils include 2 cm muck, coastal prairie redox, 5 cm mucky peat or peat, iron-manganese masses, Piedmont floodplain soils, mesic spodic, red parent material, and very shallow dark surface. The use of some indicators is restricted to certain portions of the region as noted in the regional supplement. The presence of hydrophytic vegetation and sufficient wetland hydrology indicators must be found before using the problematic soil indicators.

Indicators of Wetland Hydrology

In their review of federal wetland delineation methods, the National Academy of Sciences’ Committee for Wetland Characterization stated that hydrophytic vegetation and hydric soils are the “common diagnostic features of wetlands” and that they should be found in all wetlands except where “specific physiochemical, biotic, or anthropogenic factors have removed them or prevented their development” (National Research Council 1995). Consequently, the presence of these two factors should be sufficient to identify most wetlands with the noted exceptions. The US federal wetland delineation approach, however, typically requires further verification of wetland through the use of “wetland hydrology indicators.” Since the regulatory delineation method is establishing legal boundaries to regulate land use activities on

Table 1 List of wetland hydrology indicators (US Army Corps of Engineers 2014)

Indicator group	Indicators
Direct indicators of inundation or saturation	Water on the surface during the growing season
	Water within 30 cm of the surface during the growing season
Primary indirect indicators of inundation	Water marks
	Algal mats or crusts
	Water-stained leaves
	Water-carried debris (drift lines)
	Sediment deposits
	Sparsely vegetated concave surface
	Inundation on aerial imagery
	Iron deposits
	Aquatic fauna
	True aquatic plants (e.g., floating leaved aquatics)
	Marl deposits
Primary indirect indicators of saturation	Oxidized rhizospheres (2% or more) around living roots within 30 cm of the surface
	Odor of hydrogen sulfide within 30 cm
	Presence of reduced iron (colorimetric chemical test)
	Sign of recent iron reduction in tilled soils (2% or more redox concentrations within 30 cm of the soil surface)
	Thin muck surface (2.5 cm or less)
Secondary indicators of surface water	Surface soil cracks
	Drainage patterns (sloughs, drainageways)
	Moss trim lines
Secondary indicators of saturation	Dry season water table from 30–60 cm below the surface
	Crayfish burrows
	Saturation visible on aerial imagery
Other secondary indicators	Stunted or stressed plants
	Geomorphic position (e.g., depression, toe-of-slope, fringe of water body, or groundwater discharge site)
	Shallow aquitard (capable of producing a shallow water table within 30 cm of surface)
	Microtopographic relief (pit and low-mound topography)
	FAC neutral test (vegetation is mostly OBL and FACW species)

private property, seeking other "wetland hydrology indicators" is an additional checkpoint to help ensure that what one is seeing in terms of vegetation and soils is not a relict condition but a reflection of current hydrology. Wetland hydrology indicators can be categorized into four groups: (1) direct indicators of inundation and/or saturation (the observed presence of water on, at, or near the soil surface), (2) indirect indicators of surface water, (3) indirect indicators of soil saturation, and (4) other indicators (see Table 1). For details, consult one or more of the Corps regional supplements for wetland delineation (US Army Corps of Engineers 2014).

Delineation Methods

Primary Indicators

The primary indicators method is a rapid assessment method for wetland identification and delineation in areas without significant hydrologic modification (Tiner 1993). This approach recognizes that unique plant communities and/or soil types have formed in wetlands due to varied hydrologic regimes, climatic conditions, soil formation processes, and geomorphologic settings across the country. Within similar geographic areas, wetlands have developed characteristics different than adjacent uplands (nonwetlands) due to the presence of water in or on top of the soil for prolonged periods during most years. The visible expression of this wetness may be reflected by the plant community or in the underlying soil properties. Consequently, every wetland in its natural undrained condition should possess at least one distinctive feature that distinguishes it from the adjacent upland. This approach is not really new, but is an outgrowth of traditional methods used to recognize wetlands, including the Fish and Wildlife Service's wetland classification system (Cowardin et al. 1979). It provides a quick means of assessing the presence or absence of wetlands by looking for unique wetland characteristics. Table 2 provides a list of the primary indicators.

Three-Factor Approach

The US federal government uses three factors to identify and delineate wetlands: hydrophytic vegetation, hydric soils, and wetland hydrology. Indicators for these factors are listed above. For each plant community on a parcel of land, the three factor test is applied. To be classified as wetland, the plant community typically must exhibit a positive indicator for each factor (with few exceptions – problem situations). Hydrology indicators are separated into two groups: primary indicators (where only one is needed to verify wetland hydrology) and secondary indicators (where two or more are needed in the absence of any primary indicator). Wetland communities are separated from nonwetland communities and boundaries are marked with flagging and later surveyed, as needed, to prepare a site map showing wetlands that accompanies a permit application for construction in a wetland.

Tiered Approach

The tiered approach has been developed by some U.S. states (e.g., Massachusetts and New York). These states had a tradition of using vegetation to identify wetlands for state laws passed in the 1960s and 1970s. With subsequent knowledge gained in the use of soils for wetland identification in the 1980s, these states incorporated soils and other indicators into a tiered approach for wetland identification. The first step in this approach is to identify wetlands that can be clearly recognized by their

Table 2 List of primary indicators (Modified from Tiner 1993)

Indicator type	Indicators
Vegetation indicators	V1. OBL species comprise more than 50% of the abundant species of the plant community. (An abundant species is a plant species with 20% or more areal cover in the plant community)
	V2. OBL and FACW species comprise more than 50% of the abundant species of the plant community
	V3. OBL perennial species collectively represent at least 10% areal cover in the plant community and are evenly distributed throughout the community and not restricted to depressional microsites
	V4. One abundant plant species in the community has one or more of the following morphological adaptations: pneumatophores (knees), prop roots, hypertrophied lenticels, buttressed stems or trunks, and floating leaves. (Note: Some of these features may be of limited value in tropical USA, e.g., Hawaii)
	V5. Surface encrustations of algae, usually blue-green algae, are materially present. (Note: This is particularly useful indicator of drier wetlands in arid and semiarid regions)
	V6. The presence of significant patches of peat mosses (*Sphagnum* spp.) along the Gulf and Atlantic Coastal Plain. (Note: This may be useful elsewhere in the temperate zone)
	V7. The presence of a dominant groundcover of peat mosses (*Sphagnum* spp.) in boreal and subarctic regions. (Note: Some species may not be wetland indicators; check local authorities)
Soil indicators	S1. Organic soils (Histosols, excluding Folists) present
	S2. The presence of mineral soils with a histic epipedon
	S3. Sulfidic material (hydrogen sulfide, odor of "rotten eggs") present within 30 cm of the soil surface
	S4. Gleyed (low chroma) horizon or dominant ped faces (chroma 2 or less with mottles or chroma 1 or less with or without mottles) present immediately below the surface layer (A- or E-horizons) and within 45 cm of the soil surface
	S5. Nonsandy soils with a low chroma matrix (chroma of 2 or less) within 45 cm of the soil surface and one of the following present within 30 cm of the surface: iron and manganese concretions or nodules; distinct or prominent oxidized rhizospheres along several living roots; low chroma mottles
	S6. Sandy soils with one of the following present: thin surface layer (2.5 cm or greater) of peat or muck where a leaf litter surface mat is present; surface layer of peat or muck of any thickness where a leaf litter surface mat is absent; a surface layer (A-horizon) having a low chroma matrix (chroma 1 or less and value of 3 or less) greater than 10 cm thick; vertical organic streaking or blotchiness within 30 cm of the surface; easily recognized (distinct or prominent) high chroma mottles occupy at least 2% of the low chroma subsoil matrix within 30 cm of the surface; organic concretions within 30 cm of the surface; easily recognized (distinct or prominent) oxidized rhizospheres along living roots within 30 cm of the surface; a cemented layer (ortstein) within 30 cm of the soil surface
	S7. Native prairie soils with a low chroma matrix (chroma of 2 or less) within 45 cm of the soil surface *and* one of the following present: thin surface layer (at least 0.625 cm thick) of peat or muck; accumulation of iron (high chroma mottles, especially oxidized rhizospheres) within 30 cm of the surface; iron and manganese concretions within the surface layer (A-horizon, mollic epipedon); low chroma (gray-colored) matrix or mottles present immediately below the surface layer (A-horizon, mollic epipedon) and the crushed color is chroma 2 or less. (Note: The native prairie region extends northward from Texas to the Dakotas and adjacent Canada)
	S8. Remains of aquatic invertebrates present within 30 cm of the soil surface in nontidal pothole-like depressions
	S9. Other regionally applicable, field-verifiable soil properties associated with prolonged seasonal high water tables (e.g., Vasilas et al. 2010)

vegetation alone. For other plant communities, hydric soil or wetland hydrology indicators are used as necessary to identify the presence of wetland.

Problem Situations

Wetland delineation during the dry season may pose problems; for those situations it is vital to consider indicators that reflect wetland boundaries during the wet season. It is important to recognize that wetland delineations performed during either the wet or dry season should produce the same boundary. While the actual boundary should not change with the seasons, the presence of certain indicators definitely varies seasonally (e.g., the presence of water).

Whenever certain rules are established to identify a given feature, there always seems to be exceptions and this is especially true for wetland delineation manuals. Problem areas may be grouped in five general categories: (1) lands used for agriculture and silviculture (problems – planted species and altered hydrology), (2) problematic hydrophytic vegetation (FACU and UPL plants growing in wetlands), (3) problematic hydric soils (hydric soils lacking indicators or nonhydric soils with low chroma subsoils), (4) wetlands that periodically lack wetland hydrology indicators, and (5) wetland-nonwetland mosaics (difficult to separate due to interspersion). Many plants that are more typical of upland (dry land) can also be found in wetlands and sometimes they dominate wetlands, so exceptions have to be made for them if following a three-factor wetland identification procedure. Since many wetlands have been drained to varying degrees, they may possess relict plant communities dominated by hydrophytic species and relict hydric soils. The effectiveness of drainage needs to be determined to see if the site currently is wet enough to be classified as wetland. Overall disturbance of wetlands and the surrounding landscape makes wetland delineation a challenging exercise. Wetland delineation manuals typically address the above situations in a special section (e.g., Difficult Wetland Situations; US Army Corps of Engineers 2012). Consult regional supplements for details as differences occur across the country (US Army Corps of Engineers 2014).

Future Challenges

While much progress has been made in defining wetland and identifying them through remote sensing and on the ground, there remain many challenges. Perhaps the first challenge in wetland delineation is to have wetland scientists reach agreement on how wetland is defined (e.g., the maximum depth of water in which a wetland occurs and the minimum wetness). There are significant differences in how wetlands are defined (e.g., Ramsar Convention and US Fish and Wildlife Service). This will be not be simple as we come from different perspectives, but given the global interest in wetlands it is at least worth consideration. At a minimum, discussions between leaders of the two main definitions (the Ramsar representatives and US Fish and Wildlife Service) should convene a meeting to discuss this topic.

While photointerpretation/image analysis has proven effective in producing relatively high-quality wetland maps, the effort required to do so is very expensive and time consuming. After 35+ years of mapping wetlands, the USA finally has produced wetland maps for the conterminous USA, yet unfortunately most of the data are from the mid-1980s. Satellite-derived data are now widely available from many sources. Moreover, improvements in remote sensing technologies and analytical procedures show promise for producing data of similar quality to aerial image-derived maps (e.g., Tiner et al. 2015), but need to be compared with those derived through conventional (manual) image analysis and ground-truthed sufficiently to determine whether they are more or less equivalent products, or at least, to better understanding their differences. This is especially important to researchers modelling the impact of climate change on wetlands and carbon cycling.

Present field techniques for delineating wetlands in the USA for regulatory processes should be streamlined as considerable effort is spent collecting data that is not really necessary for making a wetland and nonwetland determination. The tiered approach to wetland delineation offers a more efficient procedure than the three-factor method. Overall, more experience in mapping tropical and subtropical wetlands needs to be gathered to help identify the best wetland indicators for ground surveys in those regions.

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Wetland Classification: Hydrogeomorphic System

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Contents

Abstract

The hydrogeomorphic classification of wetlands emphasises wetland hydrological processes and functions and their ecological significance within a generalized landscape context. In the hydrogeomorphic classification wetlands are defined as areas inundated or saturated at a frequency to support, and which normally do support, plants adapted to saturated or inundated conditions. The classification system is based on (1) geomorphic setting (i.e., topographic location), (2) dominant water source and its transport (precipitation, surface flow, subsurface flows, groundwater discharge, and artesian upwelling), and (3) hydrodynamics (e.g., the direction of flow and the strength of water movement within the wetland), and groups wetlands into seven classes: (1) DEPRESSIONAL, (2) RIVERINE, (3) MINERAL SOIL WET FLATS, (4) ORGANIC SOIL WET FLATS, (5) ESTUARINE (also referred to as TIDAL FRINGE), (6) LACUSTRINE (also referred to as LACUSTRINE FRINGE), and (7) SLOPES. The classes are considered to have distinctive "ecological character" as they represent the hydrogeomorphic functions of wetlands relating to plant structures, primary production rates, biogenic accumulation rates, and wetland sedimentary fills.

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Introduction

In hydrogeomorphic classifications of wetlands (also known as the Wetland Classification: Geomorphic-Hydrologic System for Inland Wetlands), such as that of Brinson (1993), the emphasis is on hydrological processes and functions within a generalized landscape context. The objective of such classifications generally is to develop wetland classes that show the functions and ecological significance of the wetlands. Hydrogeomorphic classifications stand in contrast to the geomorphic-hydrologic inland wetland system of Semeniuk and Semeniuk (1995, 2011) in which the emphasis is a classification that addresses and incorporates the basic attributes of specific landforms and hydrology in a nongenetic manner.

Use of hydrological and landscape attributes have been an integral part of wetland classification since earliest times (Davis 1907; Cajander 1909, 1913; Shreve 1910; Horton 1945; Strahler 1957, 1964; Odum et al. 1974; Couwenberg and Joosten 2002) and continue to be included in the most recent designs. In hydrogeomorphic classifications, the term "wetland" encompasses areas that are inundated or saturated at a frequency that will support plants adapted to such conditions. From the various hydrological and geomorphological attributes selected for classification, the hydrologic maintenance of the wetland was simplified into the following categories: precipitation, perched, groundwater, seepage, fluvial inflow and/or lateral surface flow, or by artesian water (O'Brien and Motts 1980; Gilvear et al. 1989; Brinson 1993). These hydrologic types were subsequently qualified in various ways, such as landscape position or discharge- or recharge-dominated (Novitzki 1979; Rosgen 1994; Winter 2001; Gharani 2011). Other wetland hydrologic attributes such as characteristic outflows and inflows, flooding regime, water quality, and surface water dynamics were incorporated into terms such as hydroperiod, hydrologic energy, salinity, and nutrient level.

An emphasis on hydrological processes in wetland classification is founded on the basis that they underpin biological, chemical, and energy functions, such as productivity, nutrient enrichment, and floodwater control, among others. In the various geomorphic settings that wetlands may occur, these functions derive from geomorphic, stratigraphic, hydrological, and physical and chemical conditions.

Given the principles and objectives of any hydrogeomorphic wetland classification, the benefits are its simplicity in reducing a wide range of wetland types and settings to several basic types and its relative independence from the biogeographic distribution of species. As such, the overall objective of hydrogeomorphic wetland classifications is to facilitate in a given region a clearer understanding of the relationships between species that inhabit wetlands and their environments.

Hydrogeomorphic Classification System of Brinson

The example of hydrogeomorphic classification selected here for further description is that of Brinson (1993, 1996, 2009) and Brinson and Malvárez (2002), because this approach incorporates much that is common in other hydrogeomorphic wetland classifications. The definition of wetland used by Brinson (1993) is: "those areas that are inundated or saturated at a frequency to support, and which normally do support, plants adapted to saturated or inundated conditions." Thus the definition and classification are both oriented towards identifying and working in areas colonized by wetland plants. Brinson's specific aims were to explain natural variation in wetlands and to identify ecological functions.

The hydrogeomorphic approach uses wetland hydrologic characteristics and geomorphic position to group wetlands into different classes (Brinson 1993). Initially, Brinson (1993) presented a list (deriving from F. Golet) of various geomorphic settings for wetlands (viz., blanket bog, raised bog, depressional, slope, channel, floodplain, and fringe) that may be specific to or advantageous for use in some geographic regions. The approach of Brinson (1993) was later expanded and formalized by Brinson and Malvárez (2002) to seven classes (Fig. 1): (1) DEPRESSIONAL, (2) RIVERINE, (3) MINERAL SOIL WET FLATS, (4) ORGANIC SOIL WET FLATS, (5) ESTUARINE (also referred to as TIDAL FRINGE), (6) LACUSTRINE (also referred to as LACUSTRINE FRINGE), and (7) SLOPES. (It was recommended by the United States Department of Agriculture, Natural Resources Conservation Service (2008) that the nomenclature of the categories at the level of class be capitalized; nomenclature of subclasses would be lower case).

The classification system of Brinson (1993) and Brinson and Malvárez (2002) is based on (1) geomorphic setting (Fig. 1), (2) dominant water source and its transport (Fig. 2), and (3) hydrodynamics (Fig. 2). Geomorphic setting usually refers only to topographic location (e.g., depression or flat) or to environmental setting (e.g., margins of a lake or margins of an estuary). There are several types of water source: precipitation, surface flow, subsurface flows, groundwater discharge, and artesian upwelling. Hydrodynamics includes the direction of flow and the strength of water movement within the wetland. For each of the core attributes, Brinson gives examples to demonstrate how they may possibly be linked to function and ecological significance (Table 1). To further explain wetland variation, Brinson selected attributes of hydroperiod, hydrologic energy, and nutrient level.

The wetland types distinguished in the hydrogeomorphic classifications noted above have distinctive "ecological character" (defined in Resolution VII.10 of the Ramsar Convention on Wetlands (Ramsar Convention 1999)) as they represent the hydrogeomorphic functions of wetlands relating to the plant structures, primary production rates, biogenic accumulation rates, and wetland sedimentary fills. The attributes of hydrology within the hydrogeomorphic types selected for distinguishing wetland functional types include those which directly affect vegetation and biota, such as source of water, quality of water, mechanisms of flow, and seasonality of flow.

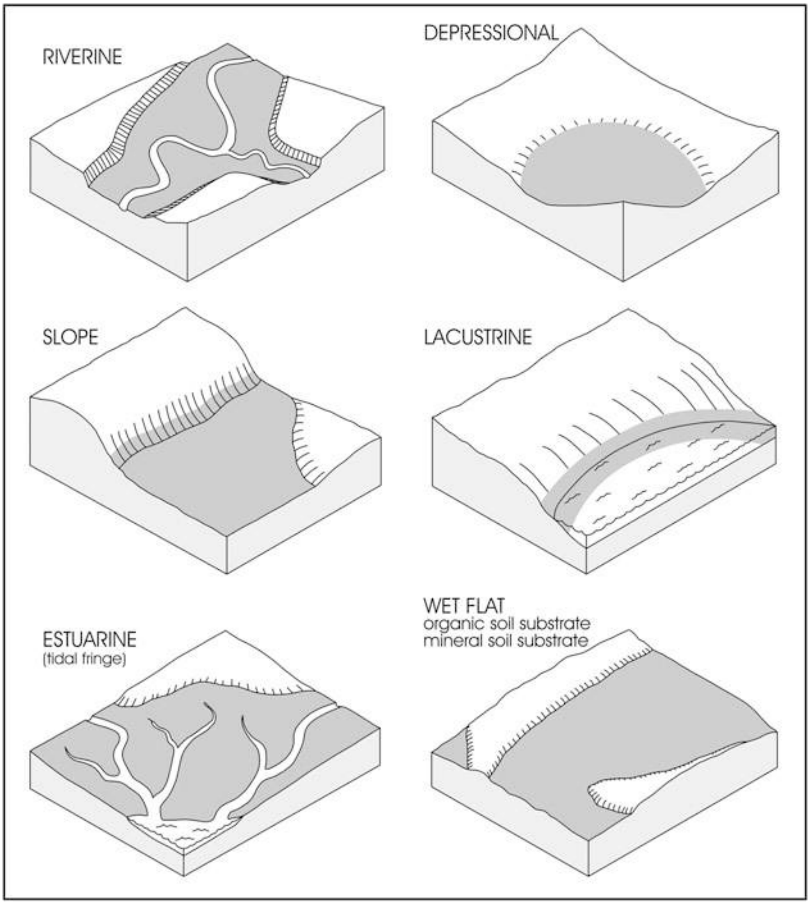

Fig. 1 Geomorphic setting of the seven classes of wetlands in the hydrogeomorphic classification; noting that the class of WET FLAT is subdivided into MINERAL SOIL WET FLAT and ORGANIC SOIL WET FLAT (Diagram adapted after Brinson and Malvárez 2002 and here modified)

There have been modification and formal expansion of the original scheme of Brinson (1993), e.g., United States Department of Agriculture, Natural Resources Conservation Service (2008), where, in addition to the seven major classes, there is use of subclasses. While the allocation of a wetland to a class provides an overview of where it occurs in the landscape, the subclass can provide details of the characteristics of the wetland and insight into major hydrologic inputs and outputs. The United States Department of Agriculture, Natural Resources Conservation Service (2008) stress that the subclass should be reflective of the primary hydrologic influence and, in keeping with the principles of the hydrogeomorphic classification, subclasses must be distinguished on the basis of morphological characteristics, water source, and/or hydrodynamics.

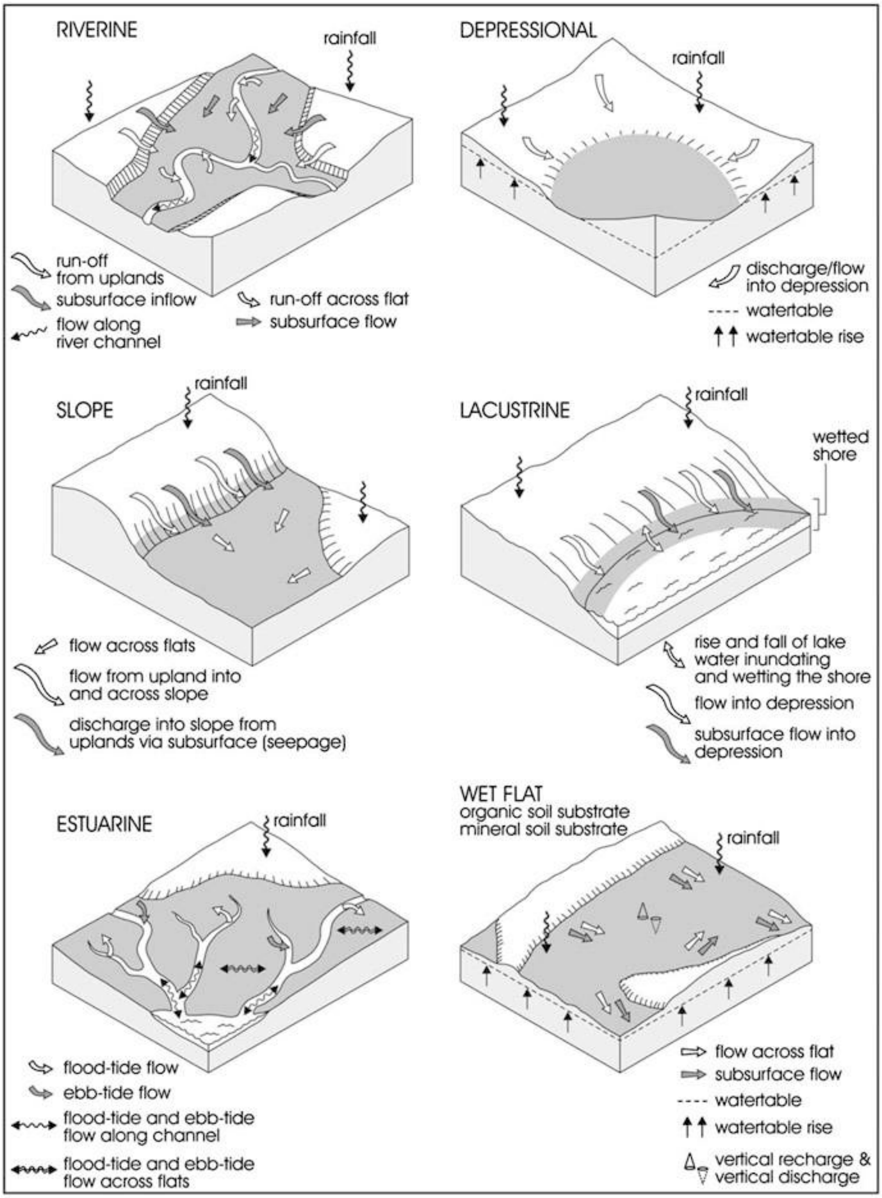

Fig. 2 Some of the hydrologic processes in terms of water sources, water flow, and local dynamics of the classes of wetlands as related to their geomorphic setting (Modified after Brinson and Malvárez 2002 and expanded here to incorporate additional hydrologic processes present in a given geomorphic setting)

Table 1 Examples of (A) geomorphic setting, (B) water source, and (C) hydrodynamic property in relation to function and ecological significance in the hydrogeomorphic classification system of Brinson (1993, 1996)

A: Example of geomorphic setting in relation to function and ecological significance in the hydrogeomorphic classification		
Geomorphic setting	**Functions**	**Ecological significance**
Shoreline of large lake	Lake serves as water supply for wetland and establishes hydroperiod gradient for wetland zonation	Provides shoreline stabilization under moderate wave action; transition habitat utilized by both aquatic and terrestrial organisms
B: Example of water source in relation to function and ecological significance in the hydrogeomorphic classification		
Water source and climatic setting	**Functions**	**Ecological significance**
Groundwater discharge to wetland (mesic climate)	Groundwater supplies nutrients, renews water, and flushes potential plant growth inhibitors	Conditions conducive for stable plant community of high productivity. Peat accumulation possible
C: Example of hydrodynamic property in relation to function and ecological significance in the hydrogeomorphic classification		
Water source	**Functions**	**Ecological significance**
Drawdowns of water table interspersed between frequent rain events that fully saturate sediments	Precipitation and evapotranspiration dominate site water balance. Floodwaters retained by depressions	Fluctuating water table conducive to rapid biogeochemical cycling; strong atmospheric exchanges

Depending on its intended use, the hydrogeomorphic classification subclasses can be expressed as a single-phase nomenclature (such as DEPRESSION) or multiphase (DEPRESSION/flow-through/ground water influenced, i.e., refined by descriptors). By using multiphase nomenclature, the subclass can indicate more detail as to how the wetland functions. Examples of this multiphase nomenclature for subclasses from the United States Department of Agriculture, Natural Resources Conservation Service (2008) include: (1) landscape setting (e.g., alluvial plain, basin, lowland); (2) landforms (e.g., arroyo, barrier flat, bog, fen, flood plain, meander scar, open depression, oxbow lake, slough, terrace); (3) microfeatures (e.g., closed depression, interdune, mound, gilgai, hummocks, mima mounds, pothole, swale, vernal pool); (4) anthropogenic features (e.g., borrow pit, pond, quarry, rice paddy); (5) tidal, nontidal, upland, bottomland; (6) ponded, flooded, saturated, open; (7) groundwater influenced; (8) leveed, incised; and (9) flow-through, recharge, discharge, connected.

The hydrogeomorphic classification is widely used as it provides a method for assessing how alterations to wetlands affect their condition and ability to perform their functions. Indicators observed in the field are often used as surrogates for functions to aid in classifying a wetland; therefore the classification is dependent on the quality of the recording and interpretation of indicators and other surrogates of wetland function.

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Wetland Classification: Geomorphic-Hydrologic System

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Christine A. Semeniuk and Vic Semeniuk

Contents

Abstract

The geomorphic-hydrologic classification treats wetlands as wet landforms, with wetlands being defined as "areas of permanently, seasonally, or intermittently waterlogged to inundated soils, sediments, or land, whether natural or artificial, fresh to saline" without recourse to identifying the vegetation type that may inhabit them. Water, through its geological/geomorphic, hydrological, and biotic interactions also drives biological productivity resulting in these wet landforms being inhabited by mosses, sedges, reeds, rushes, heaths, and forests. In the geomorphic-hydrologic classification, wetlands are separated into two fundamentally different types: 1. terrain-conforming, occurring in hollows and channels, covering plains/flats, and residing on vales, slopes, cliffs, and hill-tops; these may generate wetland sedimentary deposits, or they may simply be wetted land surfaces without any sedimentary deposits; and 2. self-emergent wetlands whose deposits accrete and rise into mounds above the land surface. Based on landform type and water regime 22 non-genetic primary categories of terrain-conforming wetlands are recognised which can be further subdivided by using

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descriptors such as size, shape, water salinity, vegetation cover, amongst others. Self-emergent wetlands can be differentiated into several types based on the composition of their accretionary material. The objective of the geomorphic-hydrologic wetland classification is to capture the global range of wetland types, and to base the classification and its associated descriptors on a hierarchical and systematic treatment of the two fundamental factors which determine the existence of wetlands: land and water.

Keywords

Wetland classification · Geomorphic-hydrologic classification · Terrain-conforming wetlands · Self-emergent wetlands · Definition of wetland

Introduction

The geomorphic-hydrologic classification of inland wetlands was first presented in 1987 to break away from approaches which subdivided wetlands into parts, zones, or vegetation types in which these subdivisions are given full wetland status. Its intention was to treat wetlands as wet landforms; hence, the definition used is that wetlands are "*areas of permanently, seasonally, or intermittently waterlogged to inundated soils, sediments, or land, whether natural or artificial, fresh to saline*" (Semeniuk 1987), without recourse to identifying the vegetation type that may inhabit it. The most obvious difference between wet landforms and all other landforms is the predominance of water in the former as surface water or near-surface groundwater (Warner 2004). Water, through its geological/geomorphic, hydrological, and biotic interactions, also drives biological productivity as a result of which wet landforms are inhabited by mosses, sedges, reeds, rushes, heaths, and forests. Glacial ecosystems are excluded on the basis *a priori* that their fundamental building material and environmental driver is ice, not water, though melted derivatives of ice (i.e., water) can form wetlands downstream of glaciers. Underground aquatic ecosystems, such as caves, are excluded because they do not occur *on* the land.

The Geomorphic-Hydrologic Classification System

The first wetland classification solely to employ geomorphic and hydrologic attributes in its construction (Semeniuk 1987) focused on wetlands in south-western Australia, but was later expanded and refined to be applicable globally (Semeniuk and Semeniuk 1995, 2011). In the geomorphic-hydrologic classification, wetlands are separated into two fundamentally different types (Semeniuk and Semeniuk 1995, 2011):

1. Those developed on and along land surfaces (i.e., they are terrain-conforming) occurring in hollows and channels, covering plains/flats, and residing on vales,

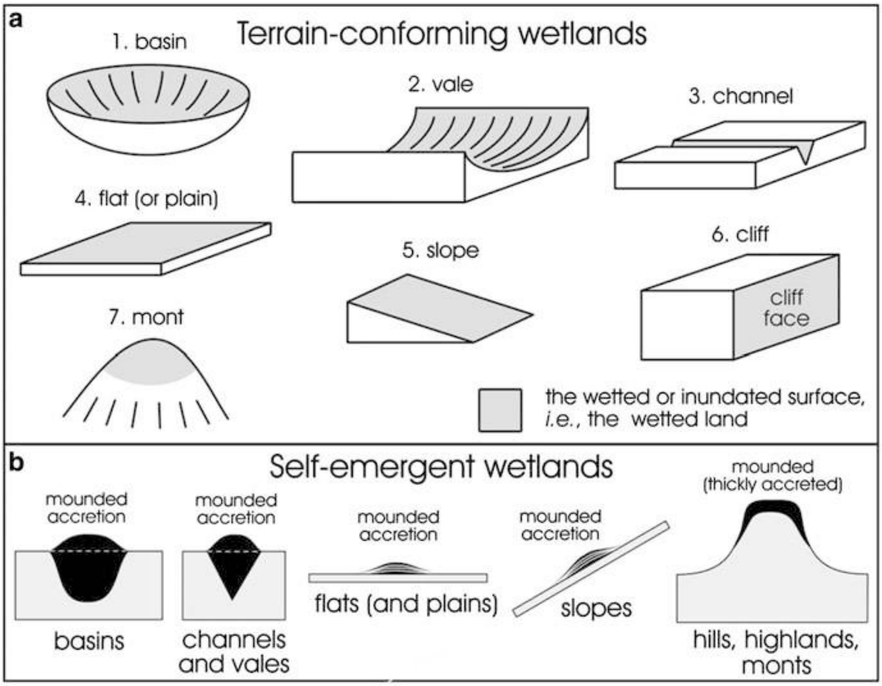

Fig. 1 Idealized diagram showing the types of landforms underpinning inland terrain-conforming wetlands and types of self-emergent wetland mounds. Represented here for terrain-conforming wetlands is a geomorphic series ranging from basin, to channel, to hill-top landscapes

slopes, cliffs, and hill-tops; these wetlands may generate wetland sedimentary deposits (but not to the extent that they have accreted to form positive self-emergent mounds), or they may simply be wetted land surfaces without any sedimentary deposits (Fig. 1a).

2. Self-emergent wetlands whose deposits accrete and rise into mounds above the land surface (Fig. 1b).

Geomorphic-Hydrologic Classification at the Site Level

Wetlands Developed on and Along Land Surfaces

At the site-specific level, the primary attributes of land and water are used to classify inland wetlands (Semeniuk and Semeniuk 1995, 2011). Seven landform types, which host wetland development, were identified: basins, channels and vales, flats (plains) slopes, cliffs, and hills (Fig. 1a). The landform refers to the geometry or structure of the land, not to the combined structure of land and accumulated wetland sedimentary fill. The complete landform should be classified, not compartmentalized,

Table 1 The matrix combining the land and water attributes, the 22 possible primary categories of wetlands, and their nomenclature (Semeniuk and Semeniuk 2011)

	Water regime or hydroperiod				
	Permanent inundation	Seasonal inundation	Intermittent inundation	Permanent waterlogging	Seasonal waterlogging
Landscape					
Basin	LAKE	SUMPLAND	PIRAPI	BASINMIRE	DAMPLAND
Channel	RIVER	CREEK	WADI	CHANNELMIRE	TROUGH
Vale				VALEMIRE	PALUSVALE
Flat or plain		FLOODPLAIN	BARLKARRA	FLATMIRE	PALUSPLAIN
Slope				SLOPEMIRE	PALUSLOPE
Cliff				CLIFFMIRE	PALUSCLIFF
Hill-top				MONTMIRE	PALUSMONT

into its smaller parts; for example, a basin is identified as an entity in preference to subbasin components of central basin (a flat floor) and steeper margin (slope). Where two or more basins are interconnected, the local central depressions still define each basin even though their littoral zones may merge.

Five categories of water regime or hydroperiod are used: permanently inundated, seasonally inundated, intermittently inundated, seasonally waterlogged, and permanently waterlogged.

Combining types of landforms with the various water regimes in a matrix structure results in 22 nongenetic primary categories of wetland (Table 1).

For the nomenclature of the primary categories of wetlands, two criteria have been used to select or design terms: 1. they must be single words rather than multiples of words (e.g., a single term such as "sumpland" is preferable to a string of words such as "seasonally inundated basin"); and 2. the core attributes or contributing attributes of the term should be deducible by its deconstruction (e.g., palusmont, whose etymology is from the Latin – *palus* and *mont*, meaning wet and marshy, and hill-top, respectively). Existing terms have been employed for some wetland categories (e.g., lake, floodplain, river). The choice of "lake" and "river" was made because they have been used relatively consistently in the literature and are in common usage to refer to, respectively, a permanently inundated basin and a permanently inundated channel.

Where hydrological regimes grade into one another, the hydrological condition should be the prevailing one and should be based on the presence of water rather than its absence. *Permanent inundation* refers to water covering the land surface throughout the year in all years, to flooded areas which are very intermittently exposed, and to permanently flooded areas which contract to a central pool, while the remaining wetland area becomes saturated or dry. An area of permanent water greater than 10% should be used as the cutoff point between permanently inundated and seasonally inundated categories during periods of surface-water contraction. Figure 2 shows a

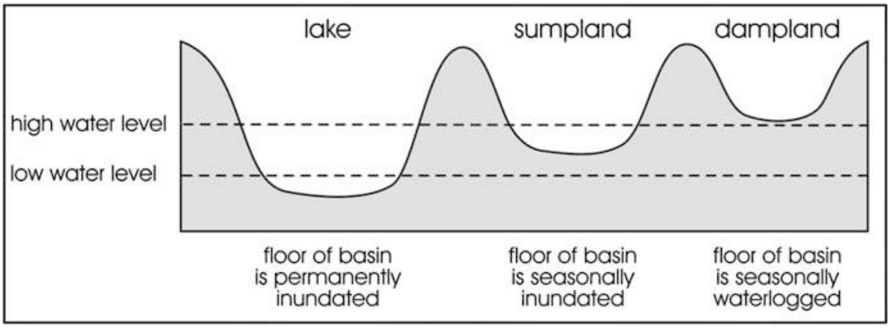

Fig. 2 Idealized diagram showing the types of water regimes or hydroperiods in three wetlands resulting from an undulating landscape intersecting a fluctuating water level

fluctuating water level intersecting an undulating landscape (of basins) with three types of hydroperiod producing three wetland types.

At the primary level of the geomorphic-hydrologic wetland classification, the style or type of hydrologic maintenance is not incorporated (i.e., how the water arrives at or sustains the wetland is not used in the primary classification), thus maintaining nongenetic aspects as it does not rely on identifying the source of the water.

Descriptors to further refine the classification of wetlands are used for both land and water (Table 2). For land, they were based on geometry, size, and vegetation cover (Semeniuk et al. 1990). For water, they were primarily related to water salinity and its consistency. An example of the application of descriptors to a wetland basin (a sumpland) is shown in Fig. 3.

Self-emergent Wetlands

Where the final wetland geomorphology is the result of combined parent (underlying) landform and emerging accumulated wetland sedimentary fill that forms a mound, the wetland is classed as self-emergent (Fig. 1b). Self-emergent wetlands can be differentiated into several types, based on the composition of their accretionary material:

1. Organic mounds, or peat mounds
2. Phytolith mounds
3. Diatomite mounds
4. Carbonate mounds
5. Gypsite mounds
6. Sinter mounds
7. Mud mounds

Table 2 List of descriptors to further refine the classification of wetlands

Descriptors for landforms
For size (see Fig. 3): macroscale, mesoscale, microscale, leptoscale; for plan shape of basins: linear, elongate, irregular, ovoid, round; for plan shape of channels: straight, sinuous, anastomosing, irregular; for substrate/soils: peat, quartz sand, carbonate sand, carbonate mud, diatomite, terrigenous mud, soda mud, gypseous, halite, phytolithic; for stratigraphy (lithology) underlying the wetland: peat-dominate, quartz sand-dominated, carbonate sand dominated, carbonate mud-dominated, diatomite-dominated, terrigenous mud-dominated, soda mud-dominated, gypsite-dominated, halite-dominated, phytolith-dominated, as well as whether it is oligolithic or polylithic; for stratigraphic thickness: thinly filled, or thickly filled; for stratigraphic layering: homogenous, two-layered, heterogeneous
Descriptors for water (in part after Semeniuk 1987; Semeniuk and Semeniuk 1995)
for salinity: fresh, subsaline, hyposaline, mesosaline, hypersaline, brine; for consistence of water salinity: poikilohaline, stasohaline; for opacity/color: clear water, black water, white water; for nutrient enrichment, oligotrophic, mesotrophic, eutrophic; for other specific hydrochemical features: acidic/alkaline, aerobic/anoxic, stratified, carbonate rich, chloride rich, etc.; for water source: ombrotrophic (meteoric), minerotrophic, artesian, magmatic; for water depth: shallow, moderately deep, deep, very deep; for rate of water movement: slow/static (lentic), fast moving (lotic); for annual freezing: cryoperiodic, not frozen (no descriptor)

Comparing the Geomorphic-Hydrologic Classification with Other Systems

The aims of the geomorphic-hydrologic wetland classification are to include the global range of wetland types and to base the classification and its associated descriptors on a hierarchical and systematic treatment of the two fundamental factors which determine the existence of wetlands: land and water. Unlike other classifications, it is not primarily a management tool. It has been designed to show-case the range of extant wetland types. Secondary aims are to facilitate comparisons, to map types of wetlands and show their distribution, and to relate occurrence to landform and climate settings. Vegetation formations are not used as a primary determinant of the classification as these are considered to be a secondary response to the primary determinants of land and water in response to climate setting, geology and soils, and water quality (for discussion see Semeniuk and Semeniuk, 1995, 2011). However, the geomorphic-hydrologic classification approach does provide a framework for relating wetland types to ecosystem structure and function. The wetland types distinguished in the geomorphic-hydrologic classification have distinctive "ecological character" [defined in Resolution VII.10 of the Ramsar Convention on Wetlands (Ramsar Convention 1999)], relating to the hierarchy of small- to large-scale causes and effects on landforms and water regime. Landform is linked to regional and local geology, climate, constructional and destructional geomorphic processes, while water is a major influence on the development of sediment types and on wetland biota and life cycles. The landform that hosts the wetland, and the water regime, which underpins many ecological processes, determine the number and type of wetland functions that occur.

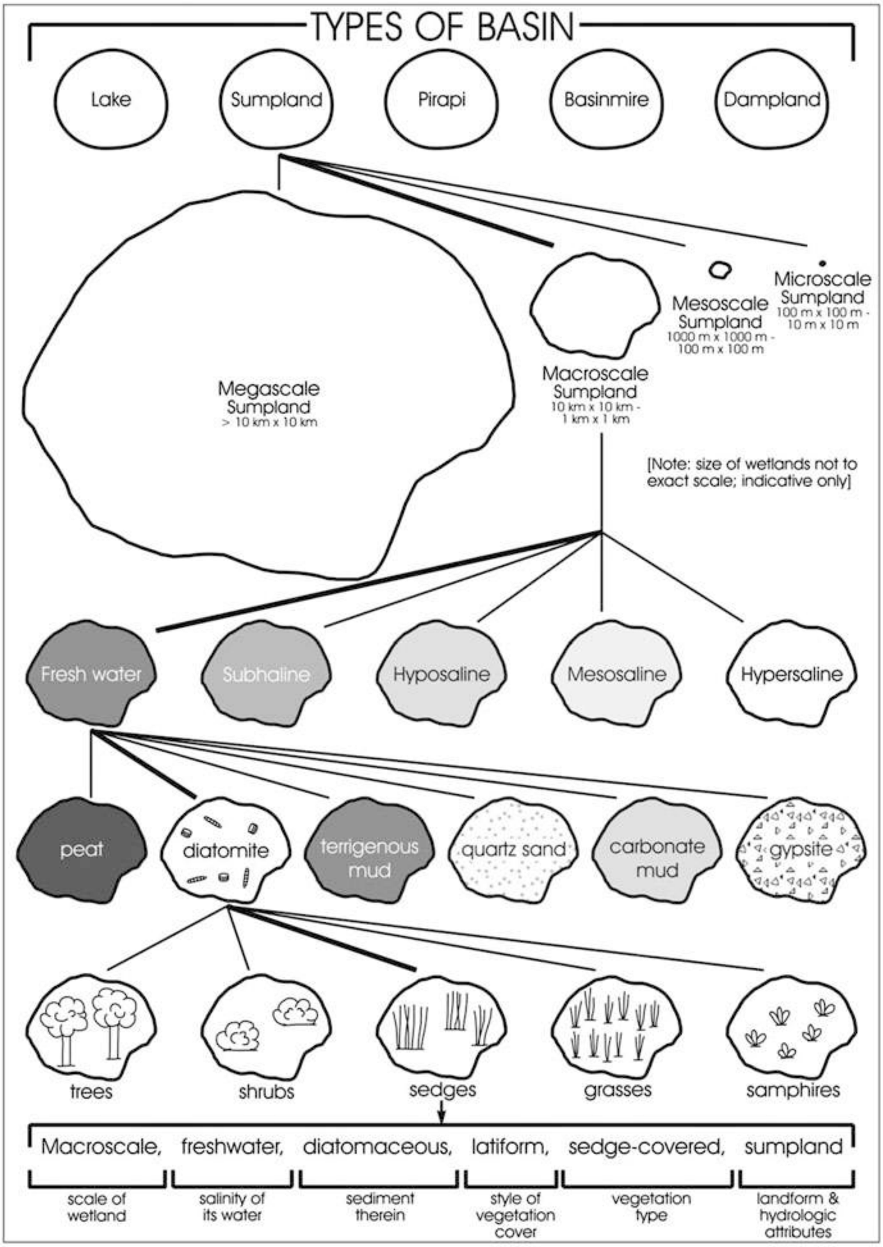

Fig. 3 Systematic application of criteria and descriptors to categorize a wetland basin in detail; in this hypothetical example, the wetland is a sumpland, and its full descriptive classification is "macroscale, freshwater, diatomaceous, latiform sedge-covered sumpland" (From Semeniuk and Semeniuk (2011) with permission from the Journal of the Royal Society of Western Australia)

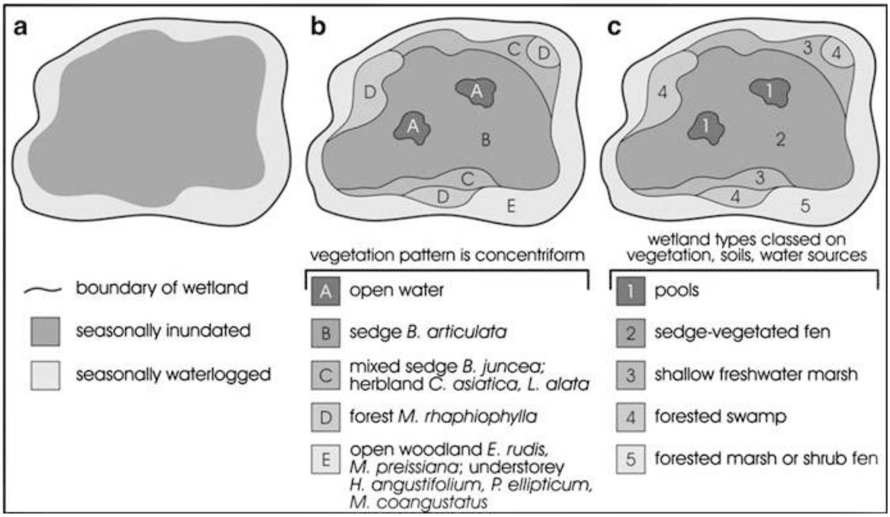

Fig. 4 Contrast in classifications using geomorphic/hydrologic attributes as criteria to classify wetlands and using ecological units. The classification in (**a**) categorizes a basin wetland as a sumpland without referring to vegetation types or ecologic units therein. The classification in (**b**) categorizes the wetland vegetation patterns according to Semeniuk et al. (1990) as concentriform [the full nomenclature of classifying wetland vegetation patterns for this example beyond simply "concentriform" is provided in Semeniuk et al. (1990)]. The species of flora in (**c**) are as follows: *B. articulata* = *Baumea articulata*; *B. juncea* = *Baumea juncea*; *M. rhaphiophylla* = *Melaleuca rhaphiophylla*; *M. preissiana* = *Melaleuca preissiana*; *E. rudis* = *Eucalyptus rudis*; *H. angustifolia* = *Hypocalymma angustifolia*; *P. ellipticum* = *Pericalymma ellipticum*; *M. coangustus* = *Meeboldina coangustus*. The term "concentriform" can be used as an adjectival descriptor to the primary geomorphic-hydrologic category. The classification in (c) categorizes the wetland basin into a plethora of ecologic types based on the variability of soils/sediments and vegetation

Wetland classifications based on geomorphic and hydrological attributes (landscapes and water regimes) are applied to whole and undivided wet geomorphic units with all their components. In contrast, wetland classifications based on ecological units, such as vegetation, often subdivide a single wetland and categorize separate zones and habitats within the wetland. For instance, a bog and a fen are identified as separate wetland types even if occurring in the same wetland basin. The geomorphic-hydrologic system, in using only two components, land and water, makes the classification universally applicable and avoids the same wet geomorphic surface being classified differently in different biogeographic regions or in different climates as happens when biota are used in classification. For example, a wetland basin classified as a "sumpland" using Semeniuk and Semeniuk (1995, 2011) could be classified as a mosaic of swamp, meadow, and bog using criteria of plant formations (Fig. 4).

Classifications that place primary emphasis on geomorphic controls on wetland development may be termed geomorphic classifications and rely on categorizing geomorphic features. Those that place primary emphasis on both the variety of geomorphic *and* variety of hydrologic controls (in combination) may be termed geomorphic-hydrologic classifications. The hydrogeomorphic wetland classification of Brinson (1993) and Brinson and Malvárez (2002) placed wetlands into a framework of hydrologic processes and function and identified classes of wetlands placed in a larger geomorphic setting of flats, depressions (basins), rivers and their valley tracts, slopes, lake edge, and estuarine margins (or tidal fringes) that include inland wetlands and coastal wetlands. The only similarity between the geomorphic-hydrologic wetland classification by Semeniuk and Semeniuk (1995, 2011) and the hydrogeomorphic wetland classification of Brinson (1993) and Brinson and Malvárez (2002) is that both address the physical diversity of landscape or geomorphology, and both are relatively independent of the biogeographic distribution of species. Also, the subdivisions of the classes of Brinson into subclasses (United States Department of Agriculture, Natural Resources Conservation Service 2008) is broadly similar, in principle but not in detail, to the use of descriptors by Semeniuk and Semeniuk (1995, 2011).

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Coastal Wetlands

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Vic Semeniuk and Christine A. Semeniuk

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Abstract

Coastal wetlands are terrains subject to wetting by coastal processes (daily wetting by tides, or wave swash and run-up, and storm surges and atmospheric-depression-induced high-water stands). Coastal wetlands do not include subtidal environments – that is the realm of the marine environment. Classifications of coastal wetlands derive from two approaches: the first is centred on coastal ecosystems, resulting in six major categories based on identifying the overriding source of stress or energy, and often linking classes to species distribution; the second focuses on geomorphic coastal types, which categorizes coastal types on their physical attributes and origin, linking coasts to morphology, evolution, and tectonic, oceanographic, or climatic setting. Geomorphically, coasts in this contribution are classed at three scales: the megascale (*e.g.*, identifying estuaries and deltas), the mesoscale (identifying units such as beaches and tidal flats), and the microscale (identified by features such as tidal level, substrates, biota, and salinity).

Keywords

Coastal classification · Coastal wetlands · Coastal ecosystems · Coastal geomorphology

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Introduction

Coastal wetlands encompass lands bordering the sea that are inundated, wetted, or waterlogged by tides, waves, and splash. Many types of coastal landforms can become wetlands because of the numerous coastal processes of landform building, coastal landform erosion, and wetting, which operate at different scales. In this context, a coastal wetland is defined as *terrain that is subject to wetting by coastal processes*, *viz.*, (1) daily wetting of tidal flats by tides, (2) wave swash and run-up, and (3) storm surges and atmospheric-depression-induced high-water stands causing flooding of lowlands by incursion of estuarine water or marine water.

As the zone of intersection and interaction of land, sea, groundwater, and atmosphere, and the processes therein, the coast is one of the most complex and rapidly evolving environments on the Earth's surface, and one of the most difficult to classify (Brocx and Semeniuk 2009). The coast is comprised of landforms with two distinct origins: those created by marine processes and those which are exposed by marine processes. Note should be taken that coastal wetlands, while encompassing the edge of the land that is wetted by coastal processes, do not include subtidal environments – that is the realm of the marine environment.

Classifications of coastal wetlands have their origin in two diverse approaches. The first is centered upon the range of coastal ecosystems and the second focuses on the range of geomorphic coastal types.

Classification Based on Coastal Ecosystems

For the classification based on coastal ecosystems, Odum et al. (1974) subdivided coastal ecosystems into six major categories based on identifying the overriding source of stress or energy. These were linked to latitudinal position and the associated large-scale climatic and oceanic processes. They ranged from ecosystems with a large latitudinal distribution (high-energy beaches and sedimentary deltas) to tropical, temperate, and arctic ecosystems, to migrating subsystems. From this beginning, wetland scientists derived a variety of ecosystem coastal types, for instance, a mangrove-specific class with different spatial distributions of energy (Lugo and Snedaker 1974; Kangas and Lugo 1990). Unfortunately, the diversity of coastal wetlands was not evident from these approaches, because while coastal wetlands themselves can be extensive, some may not be significant ecosystems, and vegetation such as mangroves can inhabit more than one type of coastal system. More recently, classifications of coastal wetland ecosystems have been based on linking this approach with species distribution (Wells 1983; Bridgewater and Cresswell 2003; Short and Woodroffe 2009).

Classification Based on Geomorphology

The second approach to coastal wetland classification requires, in the first instance, categorization of coastal types largely based on their physical attributes, or on their physical attributes in combination with their physical origin. Coasts have been classified previously using many criteria: morphology, origin, evolution, and tectonic, oceanographic, or climatic setting (Johnson 1919; Valentin 1952; Davies 1980; Semeniuk 1985, 1986; Kelletat 1995; Woodroffe 2002). Categorization of coasts, at the megascale, provides a broad descriptor of the setting of coastal wetlands but this is not a classification of coastal wetlands in itself.

Because of the number and complexity of coastal wetland types, the geomorphic classification of coastal landscapes and wetlands summarized here incorporates three scales of reference (Semeniuk 1986):

- Large-scale or megascale area (e.g., where rivers enter the sea and estuaries or deltas are developed)
- Medium-scale or mesoscale area, incorporating coastal wetland units as beaches, tidal flats, and rocky shores
- Small-scale or microscale area, incorporating the wide diversity of forms, substrates, biota, and salinity, developed in response to coastal gradients and coastal processes

The overall objective of this approach is to capture the scalar variability, complexity, and vastness of coastal wetlands.

Four types of megascale coastal systems are recognized, related to coastal shape and to the interaction between the marine environment and riverine input. These are (Fig. 1):

1. Open coastal
2. Embayed coastal
3. Estuarine
4. Deltaic

Within the megascale coastal units, a diverse range of mesoscale coastal wetlands may be developed. The mesoscale units include the following (Fig. 2):

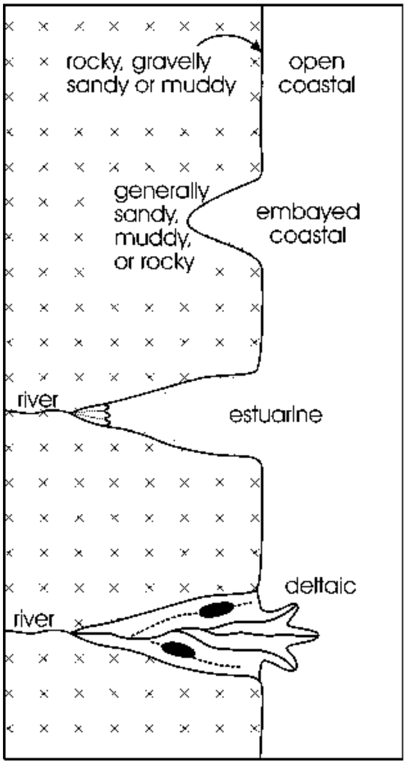
Beach	Rock pavement
Tidal flat	Rocky shore
Strandplain[1]	Bouldery shore
Spit	Alluvial fan
Chenier	Tidal creek
Shoal	Tidal lagoon
Tombolo	Biostrome[2]
Tidal delta	Bioherm[3]

[1]A strandplain is a low-gradient coastal zone of parallel to semiparallel beachridges and cheniers with intervening low swales underlain by mud or sand

[2]A biostrome is a deposit of skeletal material (such as in situ mussels, oysters, or calcareous algae), often forming a rigid frame, that accumulates as an extensive sheet or blanket without topographic relief

[3]A bioherm is a deposit of skeletal material (such as in situ oysters, sabellid worms, calcareous algae, or corals), typically forming a rigid frame, that accretes as a mound with positive topographic relief. Further definitions of the mesoscale coastal units can be found in Bates and Jackson (1987)

Fig. 1 The range of megascale coastal wetlands



These mesoscale units are the basic coastal units which can comprise coastal wetlands. In their specific use as wetland terms only the inundated or waterlogged area is relevant. For instance, traditionally, for coastal geomorphologists and sedimentologists, a beach comprises the shallow subtidal zone, the swash zone, the storm zone, and foredune zone and often is divided into nearshore, foreshore, swash, berm, and foredune, but the term "beach" as a coastal wetland involves only the swash and splash zone. In a second example, traditionally, rocky shores comprise the shallow subtidal, the swash zone, the splash-wetted zone, and any vertical cliff above the wetted zone. The term "rocky shore" as a coastal wetland involves only the tidally inundated and the shore-wetted zone.

Microscale wetlands, in essence, are broadly equivalent to habitat. At the finest scale, the microscale units within the mesoscale coastal wetlands are identified on tidal level or smaller scale morphological subdivisions. The microscale units manifest a wide diversity of form, substrates, biota, and salinity, in response to coastal gradients and coastal process. Examples of microscale units include beach, beach rock, beach berm, high-tidal rock pavement, mid-tidal rock pavement, low-tidal rock pavement, and high-, mid-, and low-tidal rocky shore.

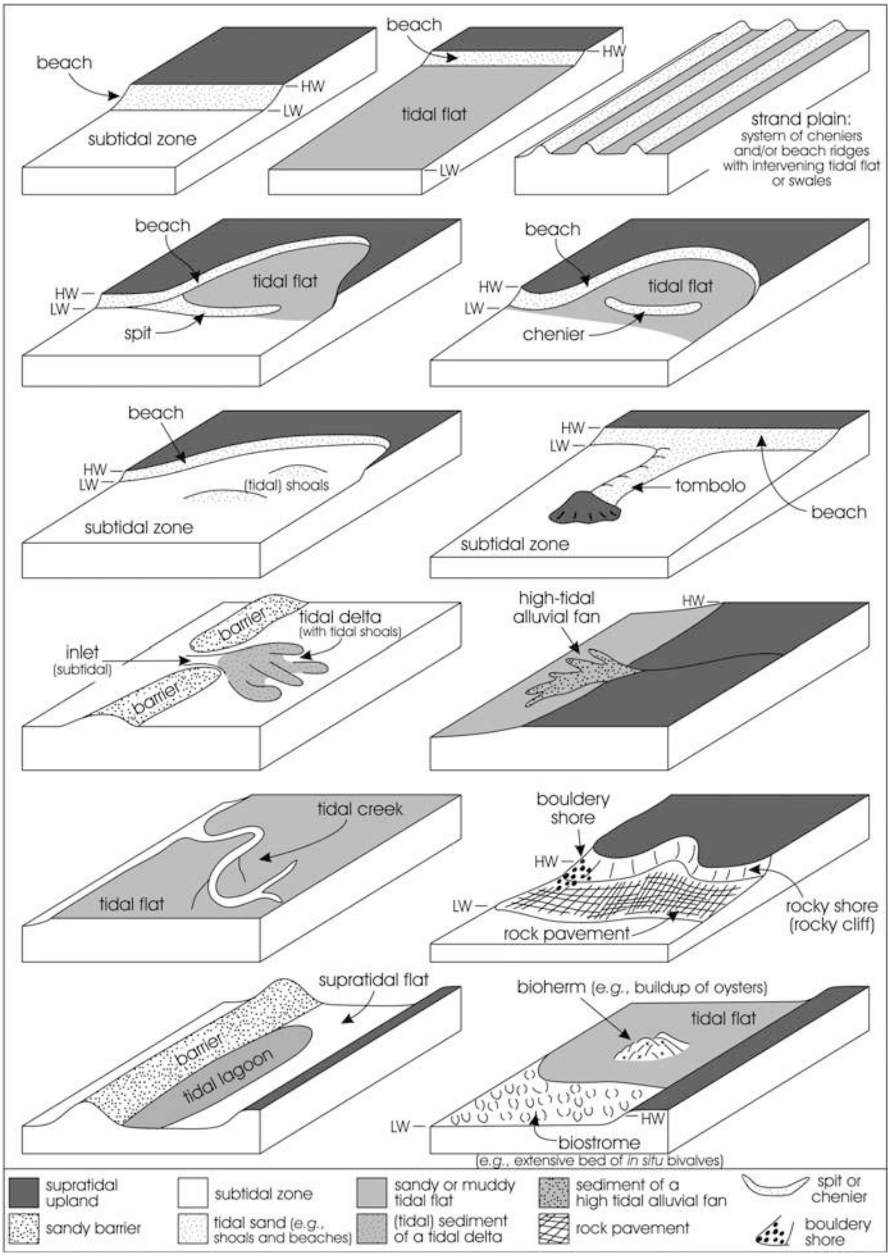


Fig. 2 The range of mesoscale coastal wetlands

Descriptors may be added at each scale to fully describe settings, wetlands, and habitats. Examples of descriptors include oceanic setting, tidal range, climate, substrate, hydrochemical, and resident biota.

The wetland types distinguished in these classifications have distinctive links to ecology (Semeniuk 1985), deriving from the hierarchy of small- to large-scale determinants and patterns of storms, tidal frequency, wave impacts, fetch, and erosional and depositional cycles on coastal landforms. At each scale, the types and effects of coastal processes vary in these dynamic settings; however, coastal wetlands may still be defined by the regime of inundation or waterlogging.

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Estuary Types

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Nick C. Davidson

Contents

Abstract

A geomorphologically-based typology of ten estuary types is described. The estuary types are: fjord, fjard, ria, coastal plain, bar-built, barrier beach, linear shore, embayment and delta.

Keywords

Estuary · Fjord · Fjard · Ria · Delta · Bar-built · Coastal plain · Embayment · Barrier beach

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Introduction

Estuaries occur along all coastlines of all continents of the world (excluding Antarctica). For a global distribution of major estuaries, see Agardy and Alder (2005), who identify about 1200 major estuaries with a total area of approximately 500,000 km^2.

Most typologies of estuaries are geomorphologically based. The descriptions of ten types of estuary below (and illustrated in Fig. 1) are derived largely from those in Davidson et al. (1991), which in turn was partly based on Fairbridge (1980). The range of estuary types are those covered by the definition of an "estuary" as: "a partially enclosed body of water, open to saline water from the sea and receiving fresh water from rivers, land run-off or seepage" (NERC 1975; Davidson et al. 1991). This typology was designed to cover all sizes of estuary, from small estuarine systems with a shoreline or tidal channel of 5 km length or larger (Davidson et al. 1991). Other typologies include those that are salinity and hydrologically based, such as *intermittent, inverse, partially mixed*, *salt wedge,* and *vertically homogenous* (see e.g., Day 1981).

Fjord

Fjords are drowned coastal glacial troughs, often associated with major lines of geological weakness. They are characteristic of areas once covered by Pleistocene ice sheets where glacial erosion has been intense or selective in its operation. Erosion by the ice further deepened existing river valleys, but movement and scouring action of the ice left characteristic shallow rock bars, particularly at fjord mouths. As a result of glacial overdeepening, fjords have a close width-depth ratio, steep sides and an almost rectangular cross-section. Their outline in plan is also typically rectangular, any sharp bends usually reflecting the underlying geological structures. Fjords generally have rocky floors, or a very thin veneer of sediment, and deposition is restricted to the head of the fjord in association with major rivers. River discharge is small compared with the total fjord volume, but as many fjords have restricted tidal ranges, the river flow is often large in relation to the tidal prism.

Fjard

Fjards are typical of glaciated lowland coasts. They are more structurally complex than fjords, with a more open and irregular coastline and often no main channel. Their form also frequently reflects the underlying geological structure, and although ice-scoured rock basins and bars are characteristic features, fjards are relatively shallow. Numerous low islands may provide localized shelter isolated by current-swept narrows. Although fjards are more exposed to wave action than fjords, they are sheltered in their upper reaches.

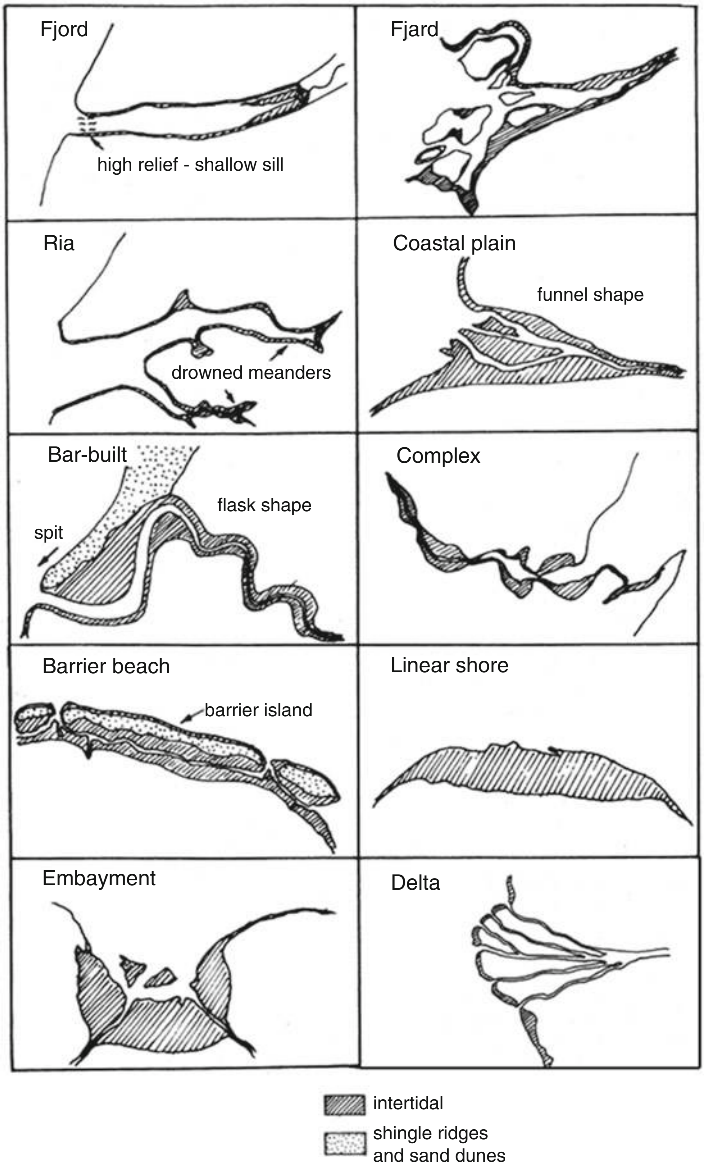

Fig. 1 A typology of estuaries (© Joint Nature Conservation Committee)

Ria

Rias are drowned river valleys formed by tectonic subsidence of the land, a rise in sea level, or a combination of both. Sedimentation has not kept pace with inundation, and the estuarine topography is still much like that of a river valley. Rias are relatively deep, narrow, well-defined channels which are almost completely marine-influenced. They have no entrance sill or ice-scoured rock bars and rock basins and are also shallower than fjords. The predominant substrate of the channel floors of most rias is bedrock, but the sheltered parts of bays and inlets adjacent to the main channels contain soft sediments, and secondary sedimentation elsewhere masks the bedrock. Although superficially rias have some features in common with fjords and fjards, they have not been formed or modified by glacial processes.

Coastal Plain Estuary

Coastal plain estuaries were formed during the Holocene transgression through the flooding of preexisting valleys in both glaciated and unglaciated areas. Maximum depths in these inlets are generally less than 30 m and the central channel is often sinuous. They have the cross-section of normal valleys and deepen and widen towards their mouths, which may be modified by spits. Their outline and cross-section are both often triangular. The width-depth ratio is usually large, although this depends on the type of rock into which the valley was cut. Unlike rias, extensive mudflats and saltmarshes often occur and the estuary is usually floored by varying thicknesses of recent sediment, often mud in the upper reaches, but becoming increasingly sandy towards the mouth. Coastal plain estuaries are generally restricted to temperate latitudes, where the amount of sediment discharged by the rivers is relatively small. River flow in large coastal plain estuaries is also small compared with the volume of the tidal prism, so that salinities in many parts of such estuaries are little reduced from sea water.

Bar-Built Estuary

These are at least partially drowned river valleys, which have been incised during ice ages and subsequently inundated. Unlike rias, recent sedimentation has kept pace with the inundation, and they have a characteristic bar across their mouths. The bar is normally formed where the waves break on the beach, and for it to develop the tidal range must be restricted and large volumes of sediment must be available. Consequently, bar-built estuaries are generally associated with depositional coasts. These estuaries are often only a few meters deep and often have extensive lagoons and shallow waterways just inside their mouth. A dominant feature of bar-built estuaries is the availability of abundant sediments in the coastal system, allowing tidal dynamics to rework and deposit this material along the coast and across river valley mouths. These sand and shingle bars at estuary mouths can develop from a variety of sediment sources: on some the bar develops from material carried down the coast by longshore drift; on others the bars develop as shingle storm beaches that are largely made up from reworked glacial deposits from subtidal offshore areas. These bars are often formed along the line of old glacial boulder moraines that influence the line of the tidal channels and so can be considered as "fossil" compared to the active displacement of estuary mouths by material carried and deposited during longshore drift.

Because of their restricted cross-sectional area, current velocities can be high at the mouth, but in the wider parts further inland they rapidly diminish. The river flow is large and seasonally variable, and large volumes of sediments are transported during floods. The estuary form is governed by the river regime at the flood stage and may show a basin/bar structure caused by meander scouring. During floods the bar may be swept completely away, but will quickly re-establish itself when the river flow diminishes. The mouth may undergo considerable variations in position from year to year. Many naturally formed coastal saline lagoons are essentially an extreme form of a bar-built estuary, in which the bar has entirely closed off the mouth of the inlet or embayment.

Complex Estuary

Complex estuaries are river estuaries of complex origin which have some characteristics of more than one other estuary type. Such estuaries usually result from the influence of a mixture of geological constraints such as hard-rock outcrops, glaciation, river erosion, and sea-level change. Complex estuaries do not, however, show assemblages of features sufficiently diagnostic of any simple estuary type. There are at least two categories of complex estuary: large, deep-water formerly glaciated firths and estuaries which reflect the influence of geological controls, glaciation, and incision into relatively hard-rock types during periods of relative sea-level change during the Quaternary.

Barrier Beach

Barrier beaches are open coast systems which characteristically develop as soft shores in shallow water, where the dissipation of wave and current energy offshore leads to the development of bars and barriers. Abundant sediments are present in these coastal systems. A classic barrier beach system has extensive saltmarshes or mangroves, creeks, and tidal flats developed behind a shingle and/or sand-dune barrier. Others have developed behind a barrier deposited in the lee of a hard-rock outcrop and so have some characteristics of a bar-built estuary.

Linear Shore

Linear shore systems include those where there is little indentation of the coast and the coastal outline is convex, linear, or only slightly concave, where broad areas of soft sediments are deposited in conditions of shallow seas and abundant sediments, and where wave and tidal energy are dissipated out from the shore, as in the development of barrier islands. They are predominately marine saline systems, with local freshwater dilution from run-off and seepage. On a larger geographic scale such linear shores can be considered as component parts of extensive estuarine complexes of a coastal plain estuary including both individual river estuaries and large areas of intervening soft shores.

Embayment

Embayments are formed where the line of the coast follows a concave sweep between rocky headlands, and they are generally infilled with extensive areas of soft sediments. At a more local scale, each of the various rivers that discharge into large embayments may demonstrate characteristics of coastal plain estuaries.

Delta

Deltas form where a river carrying copious sediments discharges into shallow coastal waters with weak tidal currents. As the flow velocity of the river slows, it can no longer carry most or all of its suspended sediment, which deposits to progressively build up the alluvium of the delta out from the coastline into shallow water, forming a deltaic lobe. The gradient of the river channel becomes less steep because of the progressive lengthening of the channel over the same change in its elevation. As the slope of the river channel decreases and sediment levels build up, it becomes easier for the river to break its banks and flow down a shorter and steeper route to the sea. This happening repeatedly over time in a process called avulsion creates a network of “distributaries” and the typical fan-shape of a mature delta. The more frequently the river avulses the closer the delta becomes to a fan-shape: deltas whose river avulses only infrequently, such as the Mississippi Delta, have a “bird’s foot” shape. For the delta to continue to develop, the volume of sediment being deposited must be greater than that eroded by coastal processes. Coastal deltas can be divided into subtypes depending on whether their sediment deposition patterns are dominated by waves or tides.

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Peatland Classification

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Richard Lindsay

Contents

Abstract

Many approaches to the classification of peat-forming (mire) systems have been devised over the years, each with its own particular focus. The present review restricts itself to those classification systems which focus on peatlands as ecosystems. Such systems are in broad agreement that there is a significant ecological

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difference between mires which are waterlogged by groundwater or accumulated surface water, broadly known as ‘fens’, and mires which are waterlogged only through direct precipitation, broadly termed ‘bogs’. Within these broad concepts, however, mires have been variously classified according to their vegetation, their chemistry, their source of water and their hydromorphology. This last system of classification is widely employed, and its hierarchical, or ‘Tope’, approach offers the potential to incorporate other classification systems at various levels within the hierarchy.

Keywords
Mire · Peatland · Bog · Fen · Soil · Wetland · Morphology · Raised bog · Groundwater · Niedermoor · Hochmoor · übergangsmoor · Ombrotrophic · Minerotrophic · Precipitation · Basin · Valley · Hydromorphology · Chemistry · Phytosociology · Synusia · Soligenous · Geogenous · Hydrogenetic · Haplotelmic · Diplotelmic · Acrotelm · Catotelm · Percolation · Tope system · Supertope · Macrotope · Mesotope · Microtope · Nanotope

Introduction

There is no *single* classification which represents the complete or definitive peatland classification system. This review provides an overview of the various ways in which peatlands have been classified while also highlighting logical strengths and degrees of internal consistency displayed by these various systems. Given the huge variety of approaches applied over the years to the description of peatlands, ranging across soil science, exploitation potential, biodiversity value, and carbon storage to name but a few, it is impossible to embrace all of these within this one review. Consequently the present review focuses only on those classification systems created to describe peatlands as *ecosystems*.

Peatlands/Mires

Peat forms when the ground surface becomes waterlogged, generally as a result of the interaction between landform, climate, and groundwater. The nature of this waterlogging is therefore commonly used to classify peat-forming systems. This contrasts with the classification of peat as a soil deposit which focuses on somewhat different factors (e.g., FAO 2009). A widely accepted definition of peat as a soil states it must be at least 30% organic plant matter accumulated in situ (Joosten and Clarke 2002) with a thickness of 30 cm or more.

First proposed by Godwin (1941) the term “mire” is now used as a general term for all natural and seminatural peatland ecosystems (Wheeler and Proctor 2000) and as an all-embracing term for ecosystems associated with peat formation (e.g., Gore 1983a). The definition has been further refined as “. . .*a wetland supporting at least some* *vegetation which is normally peat-forming*" (Löfroth 1994), and for the purposes of the Habitats Directive (European Commission 2007) an indicator of a peatland is it must possess "*a significant area of vegetation that is normally peat forming.*"

Peat accumulation itself generally modifies the landform to a greater or lesser extent and may thereby alter the nature of waterlogging. When this occurs, the morphology of the peat body is then often also used to classify such systems. The *origin* of the water which results in waterlogging, however, is widely accepted as the first key factor in distinguishing differing types of peatland ecosystem and is important because if the source of waterlogging is misunderstood, actions to manage the site may be wholly inappropriate or even damaging to the system.

Classification of Water Source

The terms "bog" and "mire" were explicitly linked in scientific usage by Tansley (1939) and Godwin (1941) to the presence of a peat soil and with peat-forming conditions. Tansley (1939) formally assigned the term "fen" to those peat-forming habitats which ranged from somewhat alkaline to moderately acid conditions as a result of their solute supply and assigned the term "bog" to habitats which were "extremely acid." Tansley (1939) thus made no explicit linkage to the *source* of water supply for these two types of peatland, unlike Weber (1902), who had earlier made a distinction between peatlands fed by mineral groundwater and "raised bogs" which he described as being waterlogged by "meteoric groundwater". Weber (1907) subsequently defined three peatland types for northern Germany: *Niedermoor* [low moor, or "fen"], *Hochmoor* [raised bog] and a transition type between the two which he termed *Übergangsmoor*.

Since then, many scientific mire typologies have recognized the two broad classes of mire system, which are distinguished by the fundamentally differing nature of their water supply. Du Rietz (1954) reviewed various approaches to mire classification, and for the two main classes of mire, he proposed two new formal terms which focused specifically on the source of water supply – namely, "ombrotrophic" for communities fed by rain ("*ombros*" being Greek for a shower of rain and "*trophe*" being Greek for food) and "minerotrophic" for communities supplied by the mineral groundwater table. In recent years, the term "bog" has therefore been applied very specifically to those peat-forming systems which are fed only by direct precipitation, while systems that are waterlogged by an excess of groundwater or surface water are referred to as "fen." The distinction is unfortunately confused by long-established names given to particular sites – for example, in the UK, Farm Bog in London is actually a fen, while Holme Fen in Cambridgeshire is a bog.

Minerotrophic Mires (Fens)

These mires are waterlogged either by groundwater or by surface water, which has gathered into distinct surface-flow channels of water movement. The water chemistry of this type of mire is thus enriched either by solutes from the bedrock and subsoil or through the cumulative effect of focused surface seepage enriched by solutes which have been flushed from adjacent surfaces. The synonym in common language for this mire type is “fen.”

Being largely dependent on the landscape around them for their water and solute supply, fens display an enormous variety of forms and are the most widespread (though not the most extensive) mire type, being found in conditions ranging from desert to the high arctic. Characteristic vegetation types range from low moss carpets to tall sedge stands to tropical wet forest. Meanwhile nutrient and total solute levels may range from extreme nutrient poverty to some of the most enriched wetland systems on Earth.

Ombrotrophic Mires (Bogs)

The second major type of mire arises where waterlogging is caused solely by an excess of direct precipitation over both evapotranspiration and run-off. The mire is entirely rain-fed, receiving no water from groundwater or from the adjacent catchment. Consequently the surface vegetation obtains no solutes apart from those which are supplied by direct precipitation. Precipitation may occur as rain, snow, fog, mist, or even dew. The term for such precipitation-dependent mires in common language is “bog,” although rather confusing terms such as “basin bog” and “valley bog” are often used in scientific literature to describe what should more accurately be termed “basin fen” and “valley fen.”

This reliance on precipitation for supply generally arises because the steady accumulation of peat eventually raises the living surface above the influence of the mineral groundwater table. The living surface remains waterlogged even as it rises above the local water supply because precipitation inputs are sufficiently regular and the peat resists water movement through it to such a degree that although there is both lateral and vertical water movement, this occurs at such a slow rate that any losses are easily replenished by fresh precipitation inputs. Once this precipitation-fed peat has accumulated to a thickness of some 30 cm or so, most plant roots are no longer able to draw on the water and solute supply of the mineral groundwater table and become entirely dependent upon precipitation water which has accumulated in the layer of ombrotrophic peat.

In order to understand the fundamental character of a mire system, it is thus essential first to identify whether the system is minerotrophic fen or ombrotrophic bog. Within a mire-rich landscape, this can require some careful consideration of landform, geology, climate, and pattern of water movement across the landscape throughout the year. Some fen systems may be very small but still make important contributions to the overall mire mosaic. If such systems are merely subsumed into, for example, a larger ombrotrophic mire type during the classification process, this can lead to apparently anomalous behavior and confusion in interpretation which need not have arisen if the small minerotrophic fen within the larger ombrotrophic bog had been correctly identified at the start.

Ecosystem Classification Approaches

Several regional descriptions of mire ecosystems were published in the early or middle decades of the last century, including Weber (1907), Tansley (1939), Sjörs (1948), and Fujiwara (1979) based on a range of differing classification approaches. Moore and Bellamy (1974) reviewed and synthesized a number of these early approaches, providing in particular a hydromorphological typology for the mire complex types found in Europe while also demonstrating the part played by water chemistry in the expression of these types. Interestingly, the two-volume global review of mires coordinated by Gore (1983b) did not attempt any form of global harmonization across differing classification systems in its overview volume, instead restricting itself to a review of the approaches listed by Moore and Bellamy (1974), while the volume devoted to regional studies served more to emphasize the disparate nature of typologies used in different parts of the world rather than explore the similarities and overlaps.

Succow and Jeschke (1990) provided another global overview using typologies which are similar to those proposed by Moore and Bellamy (1974) and made a distinction between "ecological mire types" based on water chemistry and "hydrological mire types" based on hydromorphology. A particularly informative and thought-provoking review of both mire terminology and classification approaches has been provided by Wheeler and Proctor (2000), made all the more so by the response of Økland et al. (2001) who offer a different perspective on several issues concerning the characterization of mire systems.

Joosten and Clarke (2002) have recently attempted to place hydromorphological types within a logical theoretical framework underpinned by a small number of fundamental criteria, and although this framework requires further development and probable modification, it undoubtedly provides a valuable conceptual matrix of actual and potential mire types and is explored further in the course of the present review.

Vegetation Classification

Vegetation composition is the most evident feature of mire ecosystems but has enjoyed relatively limited use as a source of universal classification units because vegetation is strongly influenced by geographical context. It is thus not so amenable to the creation of "universal" classes, although the concept of "vicariant" vegetation stands has been successfully employed (Julve 1999). The most widely applied system for the classification of mire vegetation is undoubtedly the Zürich-Montpellier system of phytosociology (Braun-Blanquet et al. 1932). The principles have been applied to a wide variety of mire systems across the globe to produce a range of regionally based vegetation "Associations."

More recently, the concept of integrated synusial phytosociology (a "synusia" being the smallest homogeneous ecological unit) has linked vegetation units with small-scale ecological structures or niches, thereby producing an approach to the classification of vegetation which is arguably more tightly integrated with the functional elements of various habitats, including mires (Gillet et al. 1991; Gillet and Gallandat 1996). Integrated synusial phytosociology is explored further in the present account when considering a hierarchical approach to mire classification.

Hydromorphological Classification of Mire Systems

Beyond the basic distinction between ombrotrophy and minerotrophy, one of the most widely used approaches to mire classification is that based on hydromorphology of the mire system (e.g., Moore and Bellamy 1974; Lindsay 1995; Steiner 2005, and references therein). Dominant hydrological processes, morphology, and associated landform are perceived as being universally identifiable and have thus tended to form the basis of many national, regional, or "universal" mire classification systems.

As an alternative to using the shape and consequent hydrology of the mire, the varied conditions leading to peat genesis led early researchers such as C.A. Weber to make a distinction between "ombrogenous" peat and "groundwater" peat (Wheeler and Proctor 2000), thereby mirroring the concepts of ombrotrophic and minerotrophic. Groundwater mire systems were then subdivided by Von Post and Granlund (1926) into "topogenous" systems that were waterlogged because they lay in topographic basins and "soligenous" systems that were waterlogged by moving water. The three concepts – ombrogenous, topogenous, and soligenous – are still widely used today in the description and classification of mires.

Joosten and Clarke (2002), in contrast, have focused on the underlying factors which influence peat genesis to construct a mire classification framework based on the source of waterlogging and behavior of the water supply. They use these factors to create a matrix of theoretical hydrological conditions which can give rise to peat formation – see Table 1. This matrix of "hydrogenetic mire types" can largely be populated with actual examples, but there are one or two theoretical "types" within the matrix which currently have no known real-life examples and not all possible methods of waterlogging are addressed. For example, mires formed by condensation linked to avalanche screes are not easily catered for in this matrix.

The two main divisions recognized by the Joosten and Clarke (2002) hydrogenetic typology are "ombro*genous*" and "geo*genous*" mires, reflecting Weber's early distinctions. However, the hydrogenetic approach, in focusing attention on the hydrological processes controlling peat *genesis*, thus deals somewhat ambiguously with the fact that mire systems may change over time from one set of hydrological conditions to another as a result of peat accumulation. Such changes may significantly alter the way in which a mire receives its water and solutes (i.e., its trophic status).

The hydrogenetic definitions of the two main types given by Joosten and Clarke (2002) do recognize the importance of trophic status because their definitions of geogenous and ombrogenous both use the key phrase "fed by." Consequently their

Table 1 Matrix of conditions giving rise to hydrogenetic mire types proposed by Joosten and Clarke (2002) (Reproduced with kind permission of IPS/IMCG)

			Level water level mires			Inclining water level mires			
	Peat formation strategy		Schwingmoor	Immersion	Water rise	Flood	Surface flow	Acrotelm	Percolation
Water supply			Continuous	Mostly continuous	Small	Periodic	Frequent	Frequent	Continuous
Mire slope			None	None	None	None/small	Small/large	Small	Small
Internal water storage			Large	Mostly large	None	Small/large	Very small	Rather large	Large
Effect on landscape water storage			Storage <	Storage <	Storage <	Storage < (>?)	Storage >	Storage >	Storage >
Origin of the water	**Ombrogenous** *bog*		Ombrogenous schwingmoor mire *Schwingmoor in bog*	Ombrogenous immersion mire *terrestrialization in bog*	Ombrogenous water rise mire *water rise in bog complex*	Ombrogenous flood mire *flood mire in bog*	Ombrogenous surface-flow mire *blanket bog*	Ombrogenous acrotelm mire *raised bog*	Ombrogenous percolation mire *percolation bog*
	Geogenous *fen*	**Soligenous**	Soligenous schwingmoor mire *floating mat in moorpool*	Soligenous immersion mire *terrestrialization in moorpool*	Soligenous water rise mire *Kesselmoor*	Soligenous flood mire *Kessel-standmoor*	Soligenous surface-flow mires *sloppy fen, Hangmoor*	Soligenous acrotelm mire	Soligenous percolation mire *some sloping fens*
		Lithogenous	Lithogenous schwingmoor mire *floating mat on lake*	Lithogenous immersion mire *lake terrestrialization mire*	Lithogenous water rise mire *groundwater rise mire*	Lithogenous flood mire *river floodplain mire*	Lithogenous surface-flow mire *most spring mires*	Lithogenous acrotelm mire	Lithogenous percolation mire *typical percolation mire*
		Thalassogenous	Thalassogenous schwingmoor mire	Thalassogenous immersion mire *coastal terrestrialization mire*	Thalassogenous water rise mire *coastal transgression mire, mangrove*	Thalassogenous flood mire	Thalassogenous surface-flow mire	Thalassogenous acrotelm mire	Thalassogenous percolation mires

definitions imply that geogenous is a synonym for minerotrophic, and ombrogenous is a synonym for ombrotrophic. This is not strictly the case. Ombro*genous*, for example, describes the conditions which generate the peat, yet peat formation may occur more than 100 years after the plants of the living surface have died and descended into the lower reaches of the acrotelm, whereas ombro*trophic* describes the conditions currently experienced by the living surface. Nonetheless the terms are treated as synonyms in much existing scientific literature.

Two concepts are however not specifically addressed in their matrix. For the majority of mires possessing a sloping water table, Joosten and Clarke (2002) propose that there are three categories of peat structure. The first category of peat is relatively dense throughout the peat profile apart from an extremely thin surface layer and thus water tends to move as surface flow; and they include "blanket bog" within this category. Damaged blanket bogs that have lost their acrotelm and thus become "haplotelmic" (i.e., single-layered – Ingram and Bragg 1984) certainly fall into this category, but undamaged examples of blanket bog are not of this type.

The second category of peat structure used by Joosten and Clarke (2002) has a relatively conductive thin surface layer (the "acrotelm") sitting on a less conductive lower body of peat (the "catotelm"). This two-layered ("diplotelmic" – Ingram and Bragg 1984) structure is a characteristic feature of both raised and blanket bogs when they are undamaged, whereas Joosten and Clarke (2002) restrict the type to raised bogs only.

The third type of peat structure dispenses with the two-layered structure altogether and simply has a very low density of peat which slowly increases with depth. This type of peat is associated by Joosten and Clarke (2002) with percolation mires. They further propose that some *bogs* are so un-decomposed that they also have this same loose structure and examples are presented from Georgia of "percolation bogs."

While there is little doubt that such percolation bogs exist, it is also fair to say that a continuum exists between the dense peat associated with surface-flow conditions, the diplotelmic conditions of many natural bogs, and the unilayered very soft peat capable of supporting large-scale percolation. The conditions described for "percolation bogs" can be found in a number of undamaged raised bogs and blanket bogs, while other examples have a more clearly defined two-layered structure. Indeed with management intervention, it is possible to move one type into another – thus, drainage can cause loss of the acrotelm and induce subsidence and compression in the catotelm, thereby converting an acrotelm mire into a surface-flow mire.

Recognizing these caveats, the conceptual matrix presented by Joosten and Clarke (2002) is nevertheless valuable in providing a clear theoretical basis for peat formation under differing conditions and helps to underpin classification systems with a logical framework. All these described hydromorphological (or hydrogenetic) types of mire are, however, just one part of the overall process involved in classifying a mire system or indeed a mire landscape. The descriptions of individual hydromorphological *mire types* represent a single level of classification within a broader hierarchy of mire classification which resembles the hierarchy of species taxonomy.

Hierarchical Classification of Mire Systems: Tope System

The complexity of mire ecosystems arises from layers of organization, each having its own distinct characteristic features, coming together to create a distinct entity in which every feature from every level plays an important part. These organizational layers are based on a range of features which are acknowledged to be particularly characteristic of mire systems and which forms the basis of an approach to mire classification first developed in the USSR during the interwar years (Ivanov 1981). These concepts have since been further developed, refined, and adopted by a number of mire-rich countries such as Canada and Norway (Wells and Zoltai 1985; Moen 1985). The approach consists of an integrated hierarchical classification which embraces the categories of mire system and vegetation recognized by more traditional approaches such as those described above, but also makes use of features such as interconnectedness with other mire systems, surface pattern, and microtopography to provide additional levels of classification. Based on the terminology used in Ivanov (1981), this hierarchical system may be referred to as the "Tope" system and is shown as a complete system in Fig. 1.

Vegetation Stand

At the smallest level of organization, there is the individual stand of ecologically uniform vegetation, or "synusia." The synusia may consist of a distinctive but repeated assemblage of species found on the top of hummocks or within a low ridge sward and may be repeated in character if not in detailed species composition in similar synusial settings through the world and described using well-established methods of vegetation classification (e.g., Moore 1968; Gillet et al. 1991; Steiner 1992).

Nanotope

Many mire communities possess a surface which consists of distinct small-scale structures such as hummocks, ridges, hollows, tussocks, and pools. This is particularly the case in those mires with a significant bryophyte component or which are characterized by tussock formation. Within a given setting, each of these structures, or "nanotopes," tends to be associated with a particular range of vegetation stands/synusia. The structures dictate the relationship between their associated vegetation and the water table but may also play an important part in dictating levels of light and shade. For those communities which are permanently or seasonally inundated, the relative depth of water or duration of inundation may act as an equivalent factor.

Microtope

The small-scale nanotope structures are rarely arranged randomly within a mire system. They will normally be arranged in some orderly way which generates a

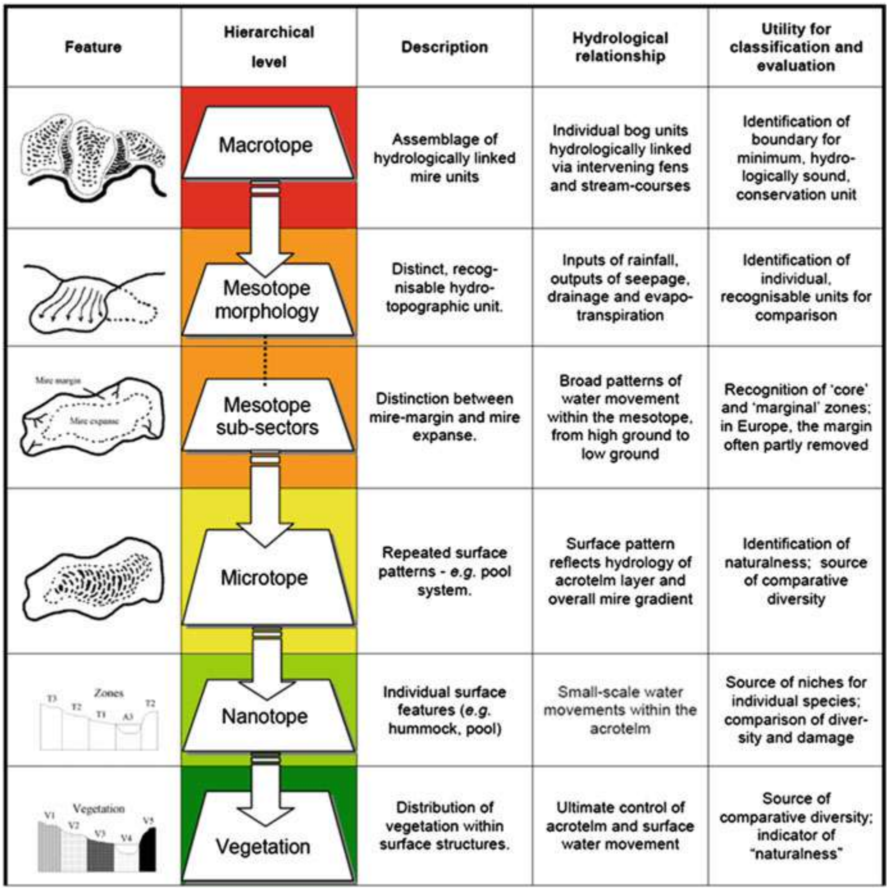

Fig. 1 Hydromorphological hierarchy of peatland ecosystems (Lindsay 2010 – Reproduced with permission of the author)

discernible pattern, termed a "microtope" pattern. This pattern is almost invariably highly informative, providing valuable information about such factors as likely source of water, direction of water flow, height of water table, or likely duration of inundation. It is important to recognize that *even a smooth surface is a pattern* – just a very simple homogeneous pattern. Microtope patterns tend to be at their most pronounced in the bryophyte-dominated mires of boreal regions in both hemispheres, but intensely patterned systems, created in many cases by vascular plants, are also found in subtropical and tropical regions. In tropical peat swamp forest, these patterns can also result from the up-ending of root-mats when trees fall (Dommain et al. 2015).

Mesotope

The mesotope represents the discrete hydromorphological mire system which forms the basis of most descriptive and classification systems for peatlands. It is, for example, quite striking that the global review of mire systems drawn together by Gore (1983a) contains regional accounts which vary greatly in the way that they deal with vegetation and water chemistry, but all provide consistent and harmonious descriptions based on hydromorphology of the mire system. The possible range of mesotopes has been explored, if not comprehensively documented, earlier in the present account. The mesotope represents a completed jigsaw of individual microtopes, each of which provides partial information about the nature and functioning of the mesotope as a whole.

Macrotope

Where a single mire mesotope links with another mesotope, this creates an interlinked peatland complex, termed the "macrotope" in which the long-term functioning of each mesotope depends on the continued functioning of the other. The simplest example of such a complex might be the raised bog and its lagg fen. The raised bog is an ombrotrophic system, while the lagg fen is by definition minerotrophic fen, and neither can continue in a fully functioning state without the other. Within landscapes dominated by peat, on the other hand, it is generally possible to identify a set of interlinked mesotopes which are then separated from other interlinked complexes by major hydrological boundaries such as rivers, rock-faces or mineral-soil ridges, and mounds. The landscape may thus be divided up into a series of largely independent macrotopes in much the same way that the same landscape can be divided into catchments (watersheds in the USA). It will often be found, however, that catchment (watershed) boundaries run through the centers of peatland macrotopes and mesotopes, thereby highlighting the sometimes unsatisfactory nature of using catchment (watershed) boundaries when studying mire systems.

Supertope

A whole mire landscape or mire region, such as the Flow Country of northern Scotland (see "The Peatlands of Caithness and Sutherland") or the West Siberian Plain (Kremenetski et al. 2003), consists of a great many macrotopes, which may be distributed continuously or discontinuously across a region in what is termed a "supertope" (Joosten and Clarke 2002). There may be very clear biogeographical zonation within such a mire region, resulting in distinctive types of mire within differing parts of the region, but broad biogeographical influences also ensure that there are underlying similarities between the various mire types.

The Functional Value of the Tope System

The value of the Tope System is that it provides a logical and coherent means of classification, which works from the smallest level of description to the largest, and highlights the functional links which exist between each level of the classification. Thus, an individual nanotope such as a tussock, present as a repeating feature in a microtope, can reveal that the mesotope has been subjected to repeated fire from which it has not yet fully recovered and therefore its carbon balance is likely to differ from mesotopes which have not suffered fire damage. If all adjacent mesotopes display the same evidence of fire damage, the macrotope as a whole may function less efficiently as a source of clean drinking water.

The Tope System thus encourages the adoption of an approach to land management which recognizes the integrated nature of the peatland system and that parts of the system cannot be managed independently from the remainder. If a corner of meadow is lost, it diminishes the area of the meadow and reduces the living space for meadow species, but the loss does not necessarily result in fundamental changes to the remainder of the meadow. The same is rarely true for a peatland. If a segment of a raised bog is removed by, for example, peat cutting, this will ultimately have an impact across the whole dome of the bog. Hydrological and morphological changes stimulated by removal of this segment will result in a re-shaping of the bog dome over a period of time until a new relatively stable hydromorphology is established, and all components of the Tope System will undergo adjustment.

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Ramsar Convention Typology of Wetlands

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C. Max Finlayson

Contents

Abstract

The Ramsar Convention's typology of wetlands was adopted in 1990 along with an information sheet for describing Ramsar sites. The typology was loosely based on the Classification of Wetlands and Deepwater Habitats of the United States and referred to the definition of wetlands adopted by the Convention in 1972. Given the breadth of the definition the classification covers a wider range of wetland types than many others. The typology comprises three broad groups of wetlands: marine and coastal; inland; and human-made. Within each group, there are a number of types, totalling 42 in all. The purpose of the typology is to provide a broad framework to assist in the rapid identification of the main wetland habitats represented at each Ramsar site.

Keywords

Wetland classification · Wetland typology · Ramsar

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Introduction

Scott and Jones (1995) have described the development of the Ramsar Convention's typology of wetlands. It was formulated by Scott (1989) as one component of an initiative to describe the features of wetlands listed as internationally important (Ramsar sites) under the Convention. It was adopted by the Convention in 1990 along with an information sheet for describing Ramsar sites. The typology was loosely based on the Classification of Wetlands and Deepwater Habitats of the United States (Cowardin et al. 1979) and referred to the definition of wetlands adopted by the Convention in 1972, namely "... wetlands are areas of marsh, fen, peatland or water, whether natural of artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed 6 m." Given the breadth of the definition the classification covers a wider range of wetland types than many others.

The Ramsar definition was purposefully broad in an effort to embrace all the "wetland" habitats of migratory water birds given the emphasis on such species in the negotiations that led to the drafting of the text of the Convention (Matthews 1993). Hence, it includes marine water less than 6 m deep at low tide, which, in northern latitudes, are often important wintering habitats for loons (divers), grebes, and sea ducks. It also includes artificial wetlands, such as reservoirs and seasonally flooded agricultural land, which are often important habitats for ducks, geese, cranes, and shorebirds. Similarly, a large part of the world's coral reefs and seagrass meadows qualify as wetlands. On a number of subsequent occasions, the coverage of wetland types was extended, for example, to incorporate karst wetlands and caves.

The Typology

The current version of the Ramsar typology recognizes three broad groups of wetlands: marine and coastal; inland; and human-made. Within each group, there are a number of types with codes that are used in the Ramsar Information Sheet when describing Ramsar sites. The purpose of the typology is to provide a broad framework to assist in the rapid identification of the main wetland habitats represented at each site. Within each grouping, a number of wetland types have been determined on the basis of settings (e.g., palustrine or riverine), water permanence (e.g., permanent, seasonal or intermittent), soils, substrates and vegetation. The typology contains 12 marine and coastal, 20 inland, and 10 human-made wetland types (Table 1).

Semeniuk and Semeniuk (1997) in a review of the Inland Wetland component of the typology noted that mixed criteria were used to separate the wetlands, and that not all natural inland wetlands had been unambiguously addressed. For instance, there was repetition of types named as "marshes" and some types were ill-defined in that they encompassed a number of types (e.g., Alpine/tundra wetlands encompass bogs, meadows, and other mires). The mixed criteria included some that were generic (such as geothermal); some that were climatic, physiographic, or vegetational; and others that were vegetative in conjunction with hydroperiod and soil

Table 1 Ramsar typology of wetlands

Wetland category	Code	Wetland type
Marine/ coastal	A	Permanent shallow marine waters in most cases less than 6 m deep at low tide; includes sea bays and straits
	B	Marine subtidal aquatic beds; includes kelp beds, seagrass beds, tropical marine meadows
	C	Coral reefs
	D	Rocky marine shores; includes rocky offshore islands, sea cliffs
	E	Sand, shingle, or pebble shores; includes sand bars, spits, and sandy islets; includes dune systems and humid dune slacks
	F	Estuarine waters; permanent water of estuaries and estuarine systems of deltas. Intertidal mud, sand, or salt flats
	G	Intertidal marshes; includes salt marshes, salt meadows, saltings, raised salt marshes; includes tidal brackish and freshwater marshes
	H	Intertidal forested wetlands; includes mangrove swamps, nipah swamps, and tidal freshwater swamp forests
	I	Coastal brackish/saline lagoons; brackish to saline lagoons with at least one relatively narrow connection to the sea
	J	Coastal brackish/saline lagoons; brackish to saline lagoons with at least one relatively narrow connection to the sea
	K	Coastal brackish/saline lagoons; brackish to saline lagoons with at least one relatively narrow connection to the sea
	Zk(a)	Karst and other subterranean hydrological systems, marine/coastal
Inland[a]	L	Permanent inland deltas
	M	Permanent rivers/streams/creeks; includes waterfalls
	N	Seasonal/intermittent/irregular rivers/streams/creeks
	O	Permanent freshwater lakes (over 8 ha); includes large oxbow lakes
	P	Seasonal/intermittent freshwater lakes (over 8 ha); includes floodplain lakes
	Q	Permanent saline/brackish/alkaline lakes
	R	Seasonal/intermittent saline/brackish/alkaline lakes and flats
	Sp	Permanent saline/brackish/alkaline marshes/pools
	Ss	Seasonal/intermittent saline/brackish/alkaline marshes/pools
	Tp	Permanent freshwater marshes/pools; ponds (below 8 ha), marshes, and swamps on inorganic soils; with emergent vegetation water-logged for at least most of the growing season
	Ts	Seasonal/intermittent freshwater marshes/pools on inorganic soils; includes sloughs, potholes, seasonally flooded meadows, sedge marshes
	U	Non-forested peatlands; includes shrub or open bogs, swamps, fens
	Va	Alpine wetlands; includes alpine meadows, temporary waters from snowmelt
	Vt	Tundra wetlands; includes tundra pools, temporary waters from snowmelt
	W	Shrub-dominated wetlands; shrub swamps, shrub-dominated freshwater marshes, shrub carr, alder thicket on inorganic soils

(*continued*)

Table 1 (continued)

Wetland category	Code	Wetland type
	Xf	Freshwater, tree-dominated wetlands; includes freshwater swamp forests, seasonally flooded forests, wooded swamps on inorganic soils
	Xp	Forested peatlands; peatswamp forests
	Y	Freshwater springs; oases
	Zg	Geothermal wetlands
	Zk(b)	Karst and other subterranean hydrological systems, inland
Human-made	1	Aquaculture (e.g., fish/shrimp) ponds
	2	Ponds; includes farm ponds, stock ponds, small tanks; (generally below 8 ha)
	3	Irrigated land; includes irrigation channels and rice fields
	4	Seasonally flooded agricultural land (including intensively managed or grazed wet meadow or pasture)
	5	Salt exploitation sites; salt pans, salines, etc.
	6	Water storage areas; reservoirs/barrages/dams/impoundments (generally over 8 ha)
	7	Excavations; gravel/brick/clay pits; borrow pits, mining pools
	8	Wastewater treatment areas; sewage farms, settling ponds, oxidation basins, etc.
	9	Canals and drainage channels, ditches
	Zk(c)	Karst and other subterranean hydrological systems, human-made

[a]"floodplain" is a broad term used to refer to one or more wetland types, which may include examples from the R, Ss, Ts, W, Xf, Xp, or other wetland types. Some examples of floodplain wetlands are seasonally inundated grassland (including natural wet meadows), shrublands, woodlands, and forests. Floodplain wetlands are not listed as a specific wetland type herein

types (such as the various swamps and marshes). Despite the inconsistencies, the typology has served the purpose it was designed for – to provide a simple listing of the wetland types that were considered by the Convention.

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South African Wetlands: Classification of Ecosystem Types

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Dean J. Ollis, Jennifer A. Day, Namhla Mbona, and John A. Dini

Contents

Abstract

This article describes the classification of marine, estuarine, and inland aquatic ecosystems, including but not restricted to "true" wetlands as per the narrow definition (as in the South African National Water Act) and discusses recent developments in the classification of aquatic ecosystems in South Africa. Marine, estuarine and inland aquatic ecosystems tend to be dealt with separately in South Africa; it is thus not surprising that separate classification systems have been independently developed in the country for these broad groups of aquatic ecosystems. Significant progress has been made in the classification of marine, estuarine and inland aquatic ecosystems in South Africa, especially in the past ten years. Much of this advancement has been spurred on by national biodiversity

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assessments and the rollout of a number of regional systematic conservation planning initiatives across the country. One of the biggest challenges, and opportunities, that has arisen out of the independent development of separate classification systems for marine, estuarine and inland aquatic ecosystems in South Africa over the past few years is the possibility of developing an integrated classification system for all aquatic ecosystems in the country.

Keywords
Classification · Typing · South Africa · Aquatic ecosystems

Introduction

In many countries, the term "wetland" is defined more restrictively than in the Ramsar definition, usually with specific reference to the presence of saturated soils and/or hydrophytic vegetation. In South Africa, as a case in point, the definition of wetlands in the National Water Act is "... land which is transitional between terrestrial and aquatic systems, where the water table is usually at, or near the surface, or the land is periodically covered with shallow water and which land in normal circumstances supports, or would support, vegetation adapted to life in saturated soil" (Republic of South Africa 1998). The prolonged (permanent or periodic) presence of water, on the land surface or in the soil, is a fundamental feature of wetlands, even when narrowly defined as above. An aquatic ecosystem can be defined as "an ecosystem that is permanently or periodically inundated by flowing or standing water, or which has soils that are permanently or periodically saturated within 0.5 m of the soil surface" (after Ollis et al. 2013). Despite being legally or otherwise viewed as transitional systems, wetlands in South Africa are considered to be a type of aquatic ecosystem (where "aquatic" implies relating to, consisting of, or being in, water) and are classified as such. Wetlands, whether defined broadly (as per the Ramsar Convention, for example) or more narrowly (to exclude rivers and permanent standing waterbodies, for example), are generally taken to exclude deep marine waters.

This article describes the classification of marine, estuarine, and inland aquatic ecosystems, including but not restricted to "true" wetlands as per the narrow definition (as in the South African National Water Act), and discusses recent developments in the classification of aquatic ecosystems in South Africa. Some of the future challenges and opportunities relating to the classification of wetlands and other aquatic ecosystems are alluded to.

Classification Systems Currently in Use in South Africa

Marine, estuarine, and inland aquatic ecosystems tend to be dealt with separately in South Africa. For example, there is separate (but overlapping) legislation, different government departments, different tertiary qualifications, and different scientific specialists for each of these. It is thus not surprising that separate classification systems have been independently developed in the country for these broad groups of aquatic ecosystems.

The primary criterion for differentiating between Marine, Estuarine, and Inland Systems is the degree of connectivity with the ocean, as explained in these definitions:

- Marine Systems are aquatic ecosystems that form part of the open ocean, ranging from deep water to the coastline, characterized along their landward edge by exposure to wave action and tidal fluctuations.
- Estuarine Systems are partially enclosed, permanent water bodies that are either continuously or periodically open to the sea on decadal timescales, extending as far as the upper limit of tidal action or penetration of salinity (after Van Niekerk and Turpie 2012).
- Inland Systems are aquatic ecosystems with no existing connection to the ocean, characterized by the complete absence of marine exchange and/or tidal influence (after SANBI 2009; Ollis et al. 2013).

Marine Ecosystems

In 2011, the South African National Biodiversity Institute (SANBI) developed an updated classification system for marine and coastal habitats in South Africa, and this system was subsequently used in the marine and coastal component of the 2011 National Biodiversity Assessment (NBA) (Sink et al. 2012). According to this classification system (Table 1), at a broad level, Marine Systems can be divided into "offshore" areas, "inshore" areas, and "the coast." Offshore areas include the "offshore pelagic" and "offshore benthic" zones, while the inshore zone consists of areas with "rocky" or "unconsolidated" substrate. The coast is divided into areas of "rocky coast," "mixed coast," and "sandy coast."

Wave exposure, grain size, geology, and/or beach state are taken into account to further subdivide the above-mentioned categories into a total of 14 broad marine ecosystem groups. Biogeographical differences (based on the delineation of marine "ecozones" and "ecoregions") can then be used, in combination with these broad ecosystem groups, to regionalize the classes. This results in a total of 136 marine and coastal habitat types for South Africa, of which 41 represent shallower habitats (<5 m deep) where marine and coastal wetlands could occur (as highlighted in Table 1).

Estuarine Ecosystems

One of the most widely used classification systems for estuarine ecosystems in South Africa to date is that of Whitfield (1992). For example, it was used in the estuary component of the 2004 National Spatial Biodiversity Assessment (Turpie 2005). Five types of estuaries are recognized, primarily on the basis of landform and hydrodynamics, namely:

Table 1 Schematic illustrating the hierarchical classification of coastal and marine habitat types in South Africa. Numbers of habitat types in each category are shown in parenthesis, and shallower habitats (<5m deep) where marine/coastal wetlands could occur are highlighted in *green* (Adapted from Sink et al. 2012, with permission South Africa National Biodiversity Institute SANBI)

MARINE AND COASTAL HABITAT TYPES (136)														
Drivers of biodiversity pattern	COASTAL (58) 500 m inland to -30 m including island-associated and lagoon habitat types Benthic & pelagic combined							OFFSHORE (78) 30 m to EEZ boundary Separate benthic & pelagic habitats						
Terrestrial & benthic-pelagic connectivity	Coast type (37) (500 m inland to -5 m) Benthic & pelagic combined			Inshore (17) (-5 m to -30 m)		Island-associated (3)	Lagoon (1)	Offshore Benthic (52)						Offshore pelagic (16)
Substrate	Rocky coast	Mixed coast	Sandy coast	Rocky	Unconsolidated	N/A	N/A	Rocky			Unconsolidated			
Depth and slope	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Shelf	Shelf edge	Deepsea	Shelf	Shelf edge	Deepsea	N/A
BROAD ECOSYSTEM GROUPS	Rocky coast (16)	Mixed coast (5)	Sandy coast (16)	Rocky inshore (8)	Unconsolidated inshore (9)	Island (3)	Lagoon (1)	Rocky shelf (12)	Rocky shelf edge (9)	Seamount (2)	Unconsolidated shelf (19)	Unconsolidated shelf edge (11)	Deepsea sediments (9)	Offshore pelagic (16)
Wave exposure, grain size, geology or beach state	Wave exposure Very exposed Exposed Sheltered Boulder	N/A	Beach state Reflective Intermediate Intermediate-dissipative Dissipative Estuarine	Geology Reef Hard grounds	Grain size Sandy Muddy Gravels Mixed	N/A	N/A	Geology Reef Hard grounds Bank	Geology Reef Hard grounds Canyons	N/A	Grain size Sandy Muddy Gravels Mixed	Grain size Sandy Muddy Gravels Mixed	Geology With or without Ferro-manganese deposits	
Biogeography (Marine Ecozones or Ecoregions)	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecoregions Southern Benguela Agulhas Natal-Delagoa	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecozones Namaqua SW Cape Agulhas	N/A	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecoregions Southern Benguela Agulhas Natal Delagoa	Ecoregions Southeast Atlantic Southwest Indian	Ecozones Namaqua SW Cape Agulhas Natal Delagoa	Ecoregions Southern Benguela Agulhas Natal-Delagoa	Ecoregions Southeast Atlantic Southwest Indian	

1. Estuarine Bay
2. Permanently Open Estuary
3. River Mouth
4. Estuarine Lake
5. Temporarily Open Estuary

One of the problems with this typology is that most estuaries in South Africa fall into the "temporarily open/closed estuary" category, with 75% of the 259 mapped estuaries along the coast being of this type. More recently, the classification system for the Estuary component of the 2011 NBA (Van Niekerk and Turpie 2012) was, therefore, based on a different approach that takes into account the following four key physical features:

1. Estuary size – large (>100 ha), medium (100–10 ha) or small (<10 ha)
2. Mouth state – permanently open or temporarily open/closed
3. Salinity structure – fresh or mixed
4. "Catchment type" – turbid, black or clear (referring to the dominant color and/or turbidity of the inflowing river(s))

In the NBA 2011, the categorization of these features was combined with the categorization of the biogeographical region (Cool Temperate vs. Warm Temperate vs. Subtropical), to derive a total of 46 estuarine ecosystem types for South Africa.

The classification of estuarine ecosystems, following either of the above-mentioned systems, does not in itself distinguish between those portions of an estuary that might be considered to be wetlands (as per the "narrow" definition of the National Water Act) and those that would not. Such a distinction can only be made through the delineation of habitat types within an estuary. For the mapping of estuarine habitats in the NBA 2011 (Van Niekerk and Turpie 2012), the following estuarine habitat types were distinguished: water surface (estuary channel), sand and mudflats, rock, and a number of plant community types (intertidal/subtidal macroalgae, submerged macrophytes, intertidal/supratidal salt marsh, reeds and sedges, mangroves, and swamp forest). Most of these habitat types would qualify as estuarine wetlands, except for the "water surface" (or at least the deeper portions of this) and "rock" habitats. Interestingly, these non-wetland habitat types together make up approximately 60% of the total extent of estuarine habitat mapped for the whole country (Van Niekerk and Turpie 2012), which implies that approximately 40% of the total estuarine habitat potentially represents "true" wetlands of one kind or another.

Inland Aquatic Ecosystems

Inland aquatic ecosystems include rivers, open waterbodies (permanently inundated lentic systems), and wetlands. Most historical attempts to develop a nationally applicable classification system for wetlands and other inland aquatic ecosystems in South Africa followed the so-called Cowardin approach to wetland classification (after Cowardin et al. 1979), which is based on the use of structural features to distinguish between primary types of aquatic ecosystems. In recent years, the development and use of classification systems based on the so-called hydrogeomorphic (HGM) approach to classification (after Brinson 1993) has become widespread in South Africa, particularly for inland wetlands in the "narrow" sense of the word.

In 2005, the Water Research Commission and SANBI initiated a project to develop a classification system for the South African National Wetland Inventory. One of the specific requirements for the classification system was that it had to cater for the broad suite of "wetlands" as defined by the Ramsar Convention, including estuarine and shallow marine systems. A prototype classification system was initially produced (Ewart-Smith et al. 2006), followed by further development and refinement (SANBI 2009). The refined version was used in the National Freshwater Ecosystems Priority Areas (NFEPA) project (Nel et al. 2011) and the "freshwater" component of the NBA 2011 project (Nel and Driver 2012).

The resulting classification system is six-tiered (see Fig. 1), progressing from Systems (Marine vs. Estuarine vs. Inland) at the broadest spatial scale (Level 1), through to HGM Units (Level 4) as the core units of classification. "Secondary discriminators" can be applied at Level 5 to classify the tidal/hydrological regime of an HGM Unit and "descriptors" at Level 6 to categorize a range of biophysical and chemical attributes.

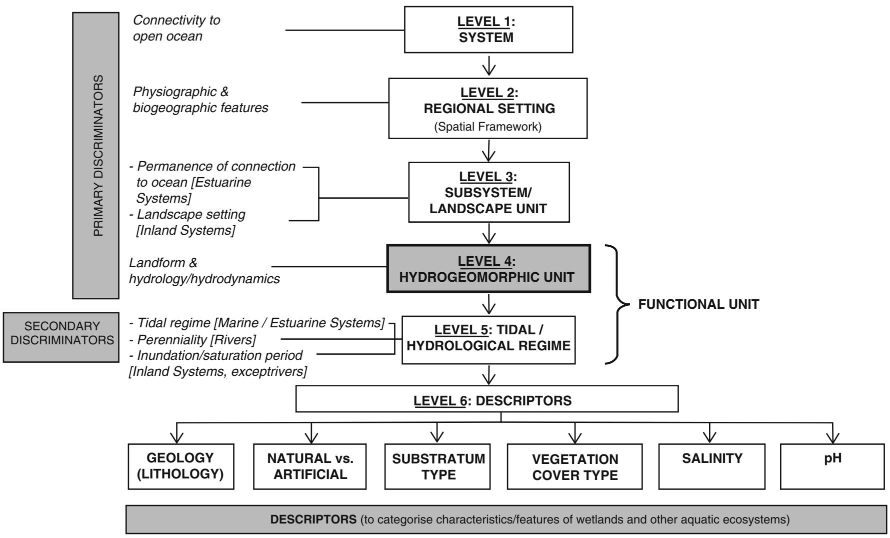

Fig. 1 Conceptual overview of the classification system for wetlands and other aquatic ecosystems, showing how "primary discriminators" are applied up to Level 4 to classify Hydrogeomorphic (HGM) Units, with "secondary discriminators" applied at Level 5 to classify the tidal/hydrological regime, and "descriptors" applied at Level 6 to categorize the biophysical characteristics of systems classified up to Level 5 (From SANBI 2009, with permission from South African National Biodiversity Institute)

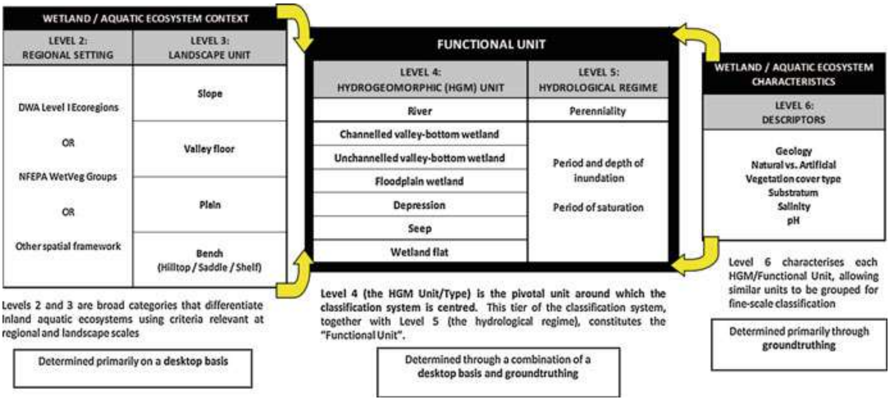

Fig. 2 Illustration of the conceptual relationship of primary HGM types (at Level 4A) to the higher and lower levels of the classification system for inland aquatic ecosystems (Aadapted from SANBI 2009, with permission South Africa National Biodiversity Institute SANBI)

The seven primary HGM types at Level 4 are the focal point of the SANBI classification system for inland aquatic ecosystems, together with the hydrological regime at Level 5 if this is known (Fig. 2; SANBI 2009; Ollis et al. 2013). The primary HGM types were derived from the HGM types recognized in the widely used WET-Health tool for assessing the present ecological condition (Macfarlane et al. 2007; Kotze et al. 2012) and WET-EcoServices tool for assessing the ecosystem services (Kotze et al. 2007) of palustrine inland wetlands in South Africa, with "rivers" and "wetland flats" introduced as additional HGM types. Levels 2 and 3 of the SANBI classification system provide the broad regional context and landscape setting for an inland aquatic ecosystem, while the "descriptors" at Level 6 provide a more detailed description of the characteristics of a particular HGM Unit or "Functional Unit." A Functional Unit is an HGM Unit and its hydrological regime, taken together to describe the functional characteristics of a particular portion of an inland aquatic ecosystem (e.g., a "seasonally inundated, permanently saturated depression").

The primary HGM types for inland aquatic ecosystems (Fig. 3) can be split into more refined HGM Units (according to Table 2), if required. This would assist, for example, in distinguishing between "exorheic" and "endorheic" depressions.

Level 5 of the SANBI classification system for inland aquatic ecosystems provides for the categorization of the hydrological regime and of the inundation depth-class for permanently inundated systems. The classification of the hydrological regime is dealt with differently for rivers than for the other six HGM types. The flow regime (i.e., perenniality) is taken as the major discriminator for the hydrological regime of rivers, whereas the inundation and saturation period (together constituting the hydroperiod) are taken as the major discriminating factors for other HGM types (see Table 3). For permanently inundated systems

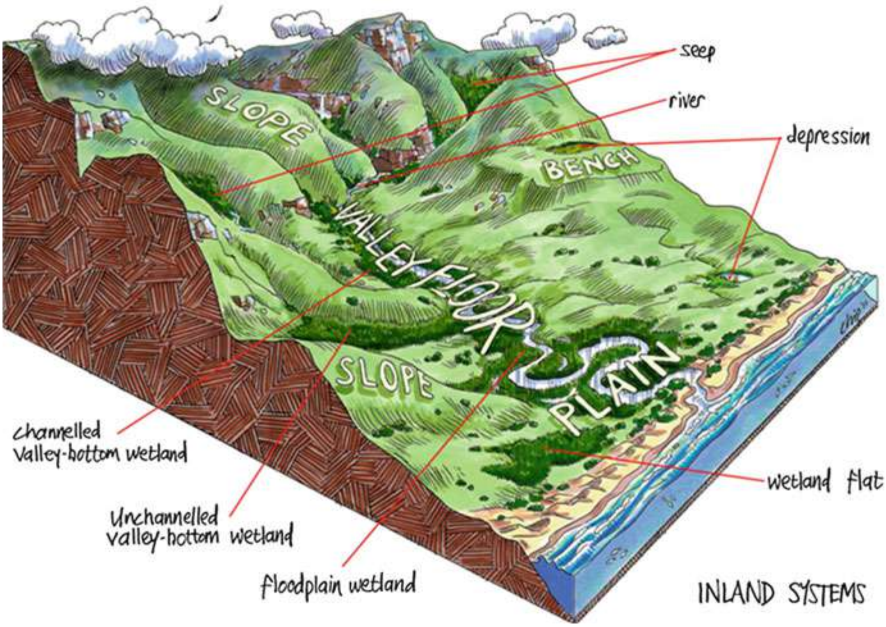

Fig. 3 Illustration of the seven primary HGM types for inland aquatic ecosystems and their typical landscape settings (By Andrew "Chip" Snaddon, from Ollis et al. 2013, with permission South Africa National Biodiversity Institute SANBI)

(i.e., open waterbodies), two depth classes are included at Level 5C to allow for the categorization of the maximum depth of inundation. A depth of 2 m (at the average annual low-water level of an open waterbody) has been used to separate deeper "limnetic" systems from shallower "littoral" systems, thus allowing for a distinction between "deepwater habitats" (sensu Cowardin et al., 1979), such as lakes or reservoirs, and more shallowly inundated areas where emergent vegetation tends to occur. This is often an important distinction to make because littoral systems/habitats are more likely to be "true" wetlands, in the "narrow" understanding of the term.

One of the important distinctions that can be made at Level 6 is between "artificial" and "natural" inland aquatic ecosystems, using the "natural vs. artificial" descriptor. The application of this descriptor is particularly important if the classification system is to be used for biodiversity/conservation planning initiatives.

A User Manual for the SANBI classification system for inland aquatic ecosystems (Ollis et al. 2013) provides illustrated, practical guidelines for applying the different levels of the system. It also includes a comprehensive glossary and a series of dichotomous keys for identifying HGM Units, Landscape Units, and the hydrological regime. A peer-reviewed paper has also been published (Ollis et al. 2015) describing the classification system and its development.

Table 2 Hydrogeomorphic (HGM) Units for inland aquatic ecosystems at Level 4 (From Ollis et al. 2013, 2015, with permission South Africa National Biodiversity Institute)

LEVEL 4: HYDROGEOMORPHIC (HGM) UNIT			
HGM TYPE	LONGITUDINAL ZONATION / LANDFORM	LANDFORM / DRAINAGE - OUTFLOW	DRAINAGE - INFLOW
A	B	C	D
River	Mountain headwater stream	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Mountain stream	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Transitional	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Upper foothills	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Lower foothills	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Lowland river	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Rejuvenated bedrock fall	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Rejuvenated foothills	Active channel	[not applicable]
		Riparian zone	[not applicable]
	Upland floodplain	Active channel	[not applicable]
		Riparian zone	[not applicable]
Channelled valley-bottom wetland	[not applicable]	[not applicable]	[not applicable]
	[not applicable]	[not applicable]	[not applicable]
Unchannelled valley-bottom wetland	[not applicable]	[not applicable]	[not applicable]
	[not applicable]	[not applicable]	[not applicable]
Floodplain wetland	Floodplain depression	[not applicable]	[not applicable]
	Floodplain flat	[not applicable]	[not applicable]
Depression	[not applicable]	Exorheic	With channelled inflow
			Without channelled inflow
		Endorheic	With channelled inflow
			Without channelled inflow
		Dammed	With channelled inflow
			Without channelled inflow
Seep	[not applicable]	With channelled outflow	[not applicable]
		Without channelled outflow	[not applicable]
Wetland flat	[not applicable]	[not applicable]	[not applicable]

Conclusions

Significant progress has been made in the classification of marine, estuarine, and inland aquatic ecosystems in South Africa, especially in the past ten years. Much of this advancement has been spurred on by national biodiversity assessments

Table 3 Hydroperiod and inundation depth-class categories for inland aquatic ecosystems other than rivers, at Level 5 of the classification system (From Ollis et al. 2013, 2015, with permission South Africa National Biodiversity Institute)

Level 5: Hydroperiod and depth of inundation		
A	**B**	**C**
Inundation period	*Saturation period (within 0.5 m of soil surface)*	*Inundation depth-class*
Permanently inundated	[not applicable]	Limnetic
		Littoral
		Unknown
Seasonally inundated	Permanently saturated	[not applicable]
	Seasonally saturated	[not applicable]
	Unknown	[not applicable]
Intermittently inundated	Permanently saturated	[not applicable]
	Seasonally saturated	[not applicable]
	Intermittently saturated	[not applicable]
	Unknown	[not applicable]
Never/rarely inundated	Permanently saturated	[not applicable]
	Seasonally saturated	[not applicable]
	Intermittently saturated	[not applicable]
	Unknown	[not applicable]
Unknown	Permanently saturated	[not applicable]
	Seasonally saturated	[not applicable]
	Intermittently saturated	[not applicable]

(undertaken in 2004 and 2011) and the rollout of a number of regional systematic conservation planning initiatives across the country. The classification system for inland aquatic ecosystems described in this article was, as a case in point, applied to the National Wetland Map generated through the National Wetland Inventory in an automated manner (using GIS modeling) to derive wetland ecosystem types. These wetland ecosystem types represent a "wetland vegetation group" derived from the grouping of vegetation types from the most recent national vegetation map for South Africa (Mucina and Rutherford 2006) (as determined at Level 2 of the classification system), together with an HGM type (as determined at Level 4A of the classification system), in an attempt to depict the diversity of wetland ecosystems across the country. The NFEPA project used the wetland ecosystem types that were derived in this manner as the basis for identifying Freshwater Ecosystem Priority Areas (FEPAs) to meet national biodiversity goals for freshwater ecosystems (Nel et al. 2011), while the freshwater component of NBA 2011 used them as the basis for categorizing, for the first time ever, the degree of threat to wetland ecosystems across the country. A rather disturbing finding of NBA 2011 was that 65% of the wetland ecosystem types derived using the classification system for inland aquatic ecosystems are threatened, making wetlands the most threatened of all ecosystems in the country (Nel and Driver 2012). The NFEPA maps have been used in the determination of management classes and the setting of resource quality objectives for inland water resources, as required in terms of the Resource Directed Measures of the National Water Act (Republic of South Africa 1998). It is envisaged that the classification system for inland aquatic ecosystems will also enable the mapping of wetlands as "ecological infrastructure," by allowing the actual or potential provision of particular ecosystem services by individual wetlands to be inferred from the hydro-geomorphological information contained in the classification.

One of the biggest challenges, and opportunities, that has arisen out of the independent development of separate classification systems for marine, estuarine, and inland aquatic ecosystems in South Africa over the past few years is the possibility of developing an integrated classification system for all aquatic ecosystems in the country. Such a classification system could build on the initial efforts by SANBI (2009) to create a structural framework for an integrated classification system for marine, estuarine, and inland aquatic ecosystems, and it could incorporate some of the advances made in the classification of these broad groups of aquatic ecosystems through the NBA 2011 project.

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USA Wetlands: Classification

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Bill O. Wilen and Frank C. Golet

Contents

Abstract

In 1979 the United States Fish and Wildlife Service (USFWS) published the Classification of Wetlands and Deepwater Habitats of the United States. In 1996, the Federal Geographic Data Committee adopted the classification as a National Standard. A revised edition of this Standard was published in 2013. The classification system permits a detailed, hierarchical description of vegetated and nonvegetated wet habitats based on vegetative life form, substrate form and composition, water regime, water chemistry, and soil properties. It has been

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used extensively in habitat mapping, as well as resource assessment and management. Most notably, the classification has supported the National Wetlands Inventory (NWI), which has been ongoing since 1976, and has performed multiple analyses of the character and dynamics of the Nation's wetlands and deepwater habitats. Digital wetlands data have been produced for the Conterminous United States, Hawaiian Islands, Guam and Saipan in the Pacific Trust Territories, Puerto Rico, US Virgin Islands, and 35% of Alaska. These data have been incorporated in five reports to Congress on the *Status and Trends of Wetlands and Deepwater Habitats of the Conterminous United States* and more than a hundred regional, state, local, watershed, and special interest reports. All of the digital wetlands data and reports may be downloaded and viewed at: http://www.fws.gov/wetlands. The classification also provides uniformity in concepts and terminology throughout the USA. In support of the classification, the US Army Corps of Engineers maintains a list of more than 8000 plants known to occur in the Nation's wetlands, while the US Department of Agriculture's Natural Resources Conservation Service has developed a definition and criteria for identifying hydric soils and has listed hydric soils at state, county, and National levels. The US wetland classification has been cited extensively in the scientific literature and applied internationally.

Keywords
Wetlands · Deepwater habitats · Wetland definition · Wetland classification · Wetland inventory · United States wetlands · National Wetlands Inventory · Wetland water regimes · Wetland water chemistry · Hydric soils · Hydrophytes · Wetland management · Wetland assessment

Development of US Classification

Worldwide recognition of the benefits of wetlands to wildlife and to humans has intensified the need for reliable information on the status and extent of wetland resources. To develop comparable information over large areas, a clear definition and classification of wetlands and deepwater habitats are required (Cowardin et al. 1979).

Historical Background

The US Fish and Wildlife Service (USFWS) conducted the first quantitative national inventory of wetlands in the mid-1950s; the results were summarized by Shaw and Fredine (1956). That inventory was based on a classification developed by Martin et al. (1953), which included 20 classes of wetlands. In 1975, the USFWS launched development of a more detailed classification designed to support a more comprehensive inventory. The first draft was reviewed by more than 150 federal, tribal, and state wetlands management personnel (Sather 1976). After major revisions, an interim classification (Cowardin et al. 1976) was tested using high- and low-altitude aerial photographs and field-visits throughout the country. The final version was published as the *Classification of Wetlands and Deepwater Habitats of the United States* (Cowardin et al. 1979). In 1996, the classification was adopted by the Federal Geographic Data Committee as a National Standard (FGDC-STD-004). The second edition of the *Wetlands Classification Standard* (FGDC-STD-004-2013) was published online in 2013 (Federal Geographic Data Committee 2013).

Objectives

The objectives of the US classification were to: (1) describe ecological units that have certain homogeneous natural attributes; (2) arrange those units in a system that aids decisions about resource management; (3) furnish units for inventory and habitat mapping; and (4) provide uniformity in concepts and terminology throughout the USA (Cowardin et al. 1979).

Basic Approach and Rationale

In most North American wetland classifications, including Martin et al. (1953), traditional terms such as marsh, swamp, bog, fen, and wet meadow played a central role. Because the definition of such terms varies widely across the USA, national compilations and regional comparisons of inventory data would be meaningless. For that reason, Cowardin et al. (1979) created a classification of individual wetland components (e.g., vegetative life form, substrate composition and texture, water regime, water chemistry, and soil). This more direct approach provides national consistency; at the same time, once a wetland's separate components have been classified, it is relatively easy to determine which of the more traditional terms would apply within a given region of the country (Cowardin and Golet 1995).

Cowardin et al. (1979) made another major departure from traditional practice during development of the US classification. They expanded the definition of wetland to include a wide variety of nonvegetated habitats, such as beaches, mud flats, and rocky shores. Their thinking was that it is hydrology, not the presence of vegetation, that determines the existence of wetland (Cowardin and Golet 1995).

Overview of US Classification

(The following is based on Cowardin et al. 1979, Cowardin and Golet 1995, and Federal Geographic Data Committee 2013.)

Definition of Wetlands and Deepwater Habitats

Marsh, bog, and swamp have been well-known terms for centuries, but only since the 1970s have attempts been made to group such landscape units under a single term, "wetlands." This general term has arisen from a need to understand and describe the characteristics and values of all types of land and to effectively manage wetland habitats as a whole. Under the US classification,

> *Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For purposes of this classification, wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year.*

The biological limits of wetlands are defined by the presence of hydrology alone or hydrology and one of the three attributes in the definition. Not all wetlands have hydrophytes, hydric soils, or even soil. Examples of these wetlands are beaches, bars, flats, and rocky shores.

In support of this definition, the US Army Corps of Engineers maintains a list of more than 8000 plant species known to occur in US wetlands (Lichvar et al. 2014). Each species is classified in terms of its regional and national frequency of occurrence in wetlands: obligate wetland (>99%), facultative wetland (67–99%), facultative (34–66%), and facultative upland (1–33%).

The US Department of Agriculture, Natural Resources Conservation Service (USDA/NRCS) has developed a definition of hydric soil and taxonomic and hydrologic criteria for identifying hydric soil (USDA/NRCS 2010), as well as lists of hydric soils of the USA and for individual states and counties (USDA/NRCS 2014). The definition of hydric soil reads as follows:

> *A hydric soil is soil that is saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions in the upper part.*

The aim of the USFWS was to map not only those areas traditionally regarded as wetlands but also those deeper waters that are frequently associated with wetlands. For that reason, the US classification distinguishes between wetlands and "deepwater habitats." The latter are defined as permanently flooded lands lying below the deepwater boundary of wetlands. In nontidal areas, the boundary between wetland and deepwater habitat occurs at a depth of 2.5 m below low water – the maximum depth to which rooted, emergent plants normally grow, and the depth beyond which "soil" does not occur, according to *Soil Taxonomy* (Soil Survey Staff 1999). In tidal areas, the boundary occurs at the extreme low water mark.

Hierarchical Structure

The classification structure consists of five levels arranged in a hierarchy, as described below. Figure 1 illustrates the classification structure to the Class level. Table 1 presents the distribution of Subclasses within the hierarchy.

The *System* (the highest level) describes the overall complex of hydrological, geomorphological, physical, chemical, and biological features that certain groups of wetlands and deepwater habitats share. Five systems are recognized: marine, estuarine, riverine, lacustrine, and palustrine.

Systems are divided into *Subsystems* primarily on the basis of water depth, surface water permanence or, in the case of the riverine system, stream gradient and extent of tidal influence. For example, the lacustrine system has two subsystems, littoral and limnetic, which are separated at a depth of 2.5 m below low water. Figure 1 shows the subsystems within each system; the palustrine system has no subsystems.

Within subsystems, the *Class* represents the basic habitat type. It describes the general appearance of the habitat in terms of either the dominant life form of the vegetation, in the case of vegetated habitats, or the form and composition of the substrate, for nonvegetated habitats. The same class may occur within two or more systems or subsystems (Fig. 1). There are six classes of nonvegetated habitats: rock bottom, unconsolidated bottom, rocky shore, unconsolidated shore, streambed, and reef. Vegetated classes include: aquatic bed, emergent (herbaceous) wetland, scrub-shrub wetland, forested wetland, and moss-lichen wetland.

Each of the 11 classes contains two or more *Subclasses* (Table 1). Subclasses are distinguished by finer differences in either vegetative life form or substrate composition. For example, forested wetlands have five subclasses: broad-leaved deciduous, needle-leaved deciduous, broad-leaved evergreen, needle-leaved evergreen, and dead. Rocky shores have two subclasses: bedrock and rubble.

Dominance Type is the lowest level in the classification hierarchy. It identifies the dominant plant or sedentary or sessile animal species within a particular subclass at a specific site. In a broad-leaved deciduous forested wetland, for example, the dominance type is the most abundant broad-leaved deciduous tree species (e.g., *Acer rubrum*). Dominance types are not listed in the classification; they are determined onsite by the user.

Modifiers

Besides vegetation and substrate composition, the classification describes water regime, water chemistry, soil type, and activity by humans or beaver (*Castor canadensis*). These features are treated as modifiers, which are applied once the habitat has been placed in the classification hierarchy.

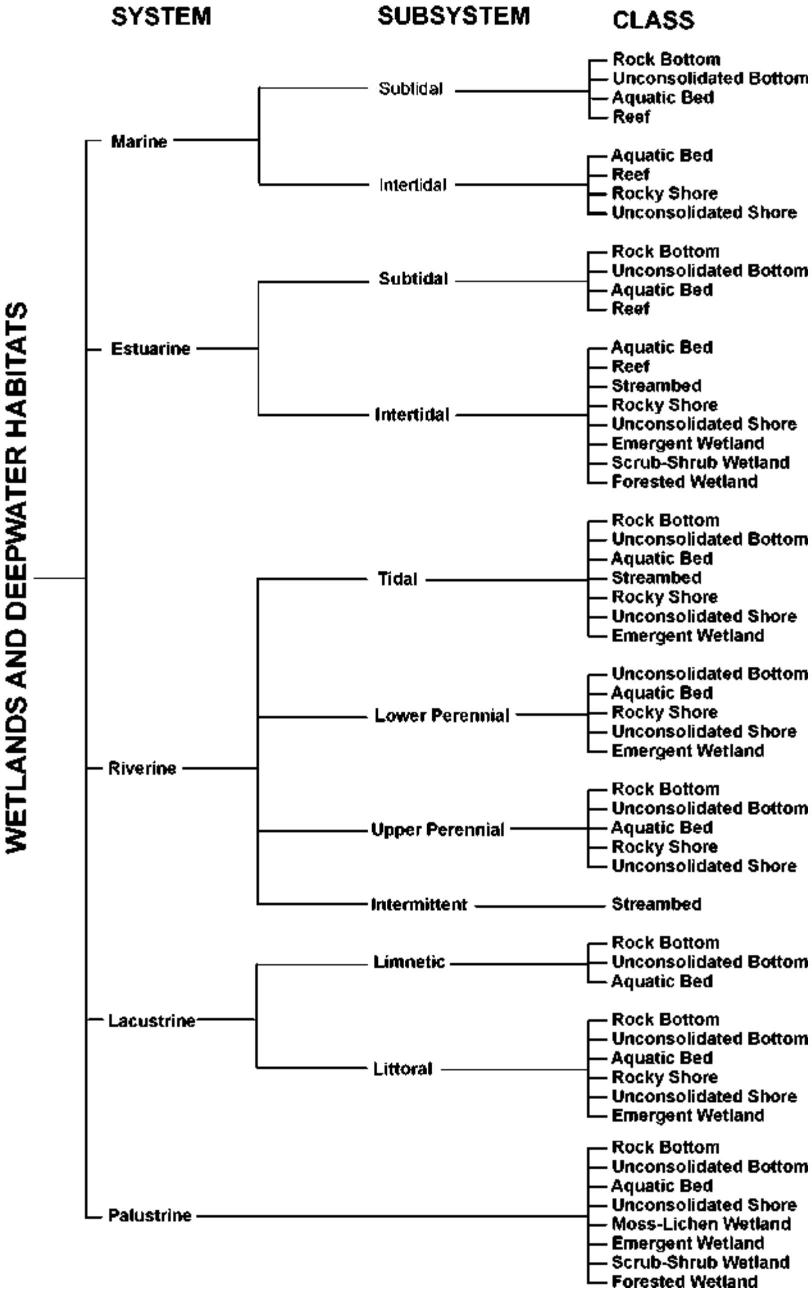

Fig. 1 Classification hierarchy of wetlands and deepwater habitats, showing systems, subsystems, and classes. The palustrine system does not include deepwater habitats

Table 1 Distribution of subclasses within the classification hierarchy

	Systems and Subsystems										
	Marine		**Estuarine**		**Riverine**				**Lacustrine**		**Palustrine**
Classes and Subclasses	**ST**	**IT**	**ST**	**IT**	**TI**	**LP**	**UP**	**IN**	**LM**	**LT**	**-**
Rock Bottom											
Bedrock	✓		✓		✓		✓		✓	✓	✓
Rubble	✓		✓		✓		✓		✓	✓	✓
Unconsolidated Bottom											
Cobble-Gravel	✓		✓		✓	✓	✓		✓	✓	✓
Sand	✓		✓		✓	✓	✓		✓	✓	✓
Mud	✓		✓		✓	✓	✓		✓	✓	✓
Organic			✓		✓	✓			✓	✓	✓
Aquatic Bed											
Algal	✓	✓	✓	✓	✓	✓	✓		✓	✓	✓
Aquatic Moss					✓	✓	✓		✓	✓	✓
Rooted Vascular	✓	✓	✓	✓	✓	✓	✓		✓	✓	✓
Floating Vascular			✓	✓	✓	✓	✓		✓	✓	✓
Reef											
Coral	✓	✓									
Mollusk			✓	✓							
Worm	✓	✓	✓	✓							
Streambed											
Bedrock				✓	✓			✓			
Rubble				✓	✓			✓			
Cobble-Gravel				✓	✓			✓			
Sand				✓	✓			✓			
Mud				✓	✓			✓			
Organic				✓	✓			✓			
Vegetated (pioneer plants)								✓			
Rocky Shore											
Bedrock		✓		✓	✓	✓	✓			✓	
Rubble		✓		✓	✓	✓	✓			✓	
Unconsolidated Shore											
Cobble-Gravel		✓		✓	✓	✓	✓			✓	✓
Sand		✓		✓	✓	✓	✓			✓	✓
Mud		✓		✓	✓	✓	✓			✓	✓
Organic		✓		✓	✓	✓	✓			✓	✓
Vegetated (pioneer plants)						✓	✓			✓	✓
Moss-Lichen Wetland											
Moss											✓
Lichen											✓
Emergent Wetland											
Persistent				✓							✓
Nonpersistent				✓	✓	✓				✓	✓

(*continued*)

Table 1 (continued)

	Systems and Subsystems										
	Marine		**Estuarine**		**Riverine**				**Lacustrine**		**Palustrine**
Scrub-Shrub Wetland											
Broad-leaved Deciduous				✓							✓
Needle-leaved Deciduous				✓							✓
Broad-leaved Evergreen				✓							✓
Needle-leaved Evergreen				✓							✓
Dead				✓							✓
Forested Wetland											
Broad-leaved Deciduous				✓							✓
Needle-leaved Deciduous				✓							✓
Broad-leaved Evergreen				✓							✓
Needle-leaved Evergreen				✓							✓
Dead				✓							✓

ST subtidal, *IT* intertidal, *TI* tidal, *LP* lower perennial, *UP* upper perennial, *IN* intermittent, *LM* limnetic, *LT* littoral

Water Regime Modifiers provide a general description of the frequency and duration of surface water inundation or soil saturation at a particular site. Three major categories of water regime modifiers are recognized: tidal salt (4), nontidal (10), and tidal fresh (5). Tidal salt water regime modifiers describe the frequency and duration of tidal flooding or exposure by ocean tides, where the salinity of the water is at least 0.5 parts per thousand (ppt); examples include: subtidal and irregularly flooded. Nontidal water regime modifiers describe hydrologic conditions during the growing season at sites not influenced by tides; semipermanently flooded and seasonally saturated are examples. Tidal fresh water regime modifiers are used at sites where water levels are affected by ocean tides, but hydrology is driven primarily by nontidal inputs, and ocean-derived salts measure less than 0.5 ppt; an example is permanently flooded-tidal fresh.

Water Chemistry Modifiers describe two characteristics: salinity and hydrogen ion concentration (pH). All habitats may be classified according to salinity; freshwater habitats (<0.5 ppt salinity) also may be classified by pH. The suffix “haline” is used for marine and estuarine habitats (e.g., Euhaline), where ocean-derived salts predominate, while the suffix “saline” is reserved for riverine, lacustrine, and palustrine habitats (e.g., eusaline). Freshwater pH may be described as acid, circumneutral, or alkaline.

There are two *Soil Modifiers*, mineral and organic, which are based on the definitions of mineral and organic soils spelled out in *Soil Taxonomy* (Soil Survey Staff 1999). Application of soil modifiers is appropriate in inland and coastal wetlands where unconsolidated substrates are capable of supporting emergent herbs, emergent mosses or lichens, shrubs, or trees, and in deepwater (subtidal) habitats within the marine and estuarine systems where the water depth is less than 2.5 m below extreme low water. In both cases, “soil” is present, according to *Soil Taxonomy*.

Special Modifiers are applied to those wetland and deepwater habitats that have been created or modified by humans or beaver. Examples include: beaver, farmed, and partly drained/ditched.

Regionalization of the Classification

The US Fish and Wildlife Service has adopted Bailey's (1976, 1995) ecoregions (see www.fs.fed.us/rm/ecoregions/products/map-ecoregions-united-states/) to provide an "ecological address" for inland wetlands and deepwater habitats in this country (Federal Geographic Data Committee 2013). Bailey's classification is hierarchical. The upper three levels are domain, division, and province; they are based on subcontinental climate, regional climate, and broad plant formations, respectively. As an example, wetlands and deepwater habitats in northern Wisconsin lie within Bailey's Humid Temperate Domain (200), Warm Continental Division (210), and Laurentian Mixed Forest Province (212). Status and trend analyses of wetlands by ecoregion are likely to become increasingly informative in light of global climate change.

Use of US Classification

National Wetlands Inventory

The classification system was developed to support a detailed inventory and periodic monitoring of the Nation's wet habitats using remote sensing. As of August 2014, the USFWS National Wetlands Inventory (NWI) had produced digital wetland data of the Conterminous United States, Hawiian Islands, Puerto Rico, US Virgin Islands, Guam and Saipan in the Pacific Trust Territories, and 35% of Alaska. These data are available for viewing and downloading from http://www.fws.gov/wetlands/ and have been incorporated in five reports to Congress on the *Status and Trends of Wetlands and Deepwater Habitats of the Conterminous United States* and more than a hundred regional, state, local, watershed, and special interest reports. The classification system (Cowardin et al. 1979) has been cited extensively in the scientific literature and applied internationally.

In 2002, the United States Office of Management and Budget (OMB) issued a revised Circular A-16, *Coordination of Geographic Information and Related Spatial Data Activities*. It established the National Wetlands Inventory as the wetlands layer data theme of the National Spatial Data Infrastructure. OMB released Circular A-16, *Supplemental Guidance*, which made the National Wetlands Inventory wetlands data layer a National Geospatial Data Asset (NGDA).

To enhance the utility of NWI data for better characterizing wetlands and for preparing preliminary assessments of potential wetland functions, "NWI+" adds landscape position, landform, water flow path, and waterbody type (LLWW descriptors) to the existing digital database codes.

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USA Wetlands: NWI-Plus Classification System

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Ralph W. Tiner

Contents

Abstract

The original classification system used to inventory and map US wetlands was based on the Cowardin system with its descriptors based on soil, vegetation, hydrology, and water chemistry. This system was extended with hydrogeomorphic descriptors (the LLWW descriptors: landscape position, landform, water flow path, and water body type) to enable functional assessment of wetlands. The resulting system, the NWI+ database, is used to classify wetlands according to both Cowardin and LLWW types and can predict function for the wetlands in a watershed. The 11 functions predicted routinely are surface water detention, coastal storm surge detention, streamflow maintenance, nutrient transformation, sediment and other particulate retention, carbon sequestration, bank and shoreline stabilization, provision of fish and aquatic invertebrate habitat, provision of waterfowl and waterbird habitat, provision of habitat for other wildlife, and provision of habitat for unique, uncommon, or highly diverse

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wetland plant communities. The LLWW descriptors can also help to predict the impact of change on wetland functions. Results of assessments are published in reports, and an online mapping tool is available.

Keywords
Wetland classification · Wetland descriptors · NWI+ · Hydrogeomorphic classification · USA wetlands · Wetland function

Introduction

Wetland mapping in the United States has largely been accomplished by the US Fish and Wildlife Service's National Wetlands Inventory (NWI) Program with cooperation and support from many state and other federal agencies (Tiner 2009). The official classification used for describing wetlands is the Cowardin et al. system (1979; herein referred to as the Cowardin system), which has been recently updated by the US Federal Geographic Data Committee (FGDC 2013). Although this system provides vital information for classifying and inventorying wetlands, additional information is needed to facilitate its use for predicting wetland functions. Dr. Mark Brinson recognized this and developed a hydrogeomorphic wetland classification (Brinson 1993). His system was not designed for inventories or as a add-on to the Cowardin system, but was presented as a concept for promoting functional assessment for evaluating site-specific impacts and determining appropriate mitigation through the use of "reference" wetlands. In 1995, the NWI Program began exploring the application of hydrogeomorphic properties for expanding the utility of NWI data for landscape level functional assessment (Tiner 1995a, b). Since then a series of hydrogeomorphic descriptors have been developed for use in wetland inventories to describe landscape position, landform, water flow path, waterbody types, and other features ("LLWW descriptors;" Tiner 2014). When LLWW descriptors are added to the categories mapped following Cowardin, the resultant database – NWI+ database – can be used to better describe and categorize wetlands and to predict wetland functions at the landscape level (Tiner et al. 2013a). When the FGDC established a federal wetland mapping standard in 2009 (FGDC 2009), it recommended adding hydrogeomorphic features to wetland inventories to increase their functionality.

While the Cowardin system includes features that describe the ecological system, predominant vegetation or substrate, hydrology, water chemistry, and other characteristics, the location of a wetland on the landscape, its physiognomy, and connectivity to other wetlands and waters can only be obtained by consulting other data sources, mainly topographic data and aerial imagery. During the mapping phase of the inventory, image analysts could add these and other features to the database. These properties include landscape position, landform, water flow path, and waterbody type ("LLWW descriptors" representing the first letter of each feature). When this information is combined with the basic wetland features from the Cowardin classification (system, class, subclass, water regime, and special modifiers), the resultant database has much greater functionality. The expanded wetlands database is called a "NWI+ database." By reviewing the literature and working with wetland specialists, a set of correlations linking the attributes in the NWI+ database to numerous wetland functions has been established (e.g., Tiner 2003, 2011). An overview of this process and applications can be found in "NWIPlus: Geospatial Data for Watershed-level Functional Assessment" (Tiner 2010).

Expanded Wetland Classification

The LLWW classification contains four major elements to describe wetlands beyond the Cowardin et al. (1979) classification: (1) landscape position, (2) landform, (3) water flow path, and (4) waterbody type (Tiner 2011, 2014). These hydrogeomorphic-type descriptors focus on abiotic properties that are vital to predicting wetland functions.

Five landscape positions describe the location of a wetland relative to a waterbody if present: (1) *marine* (along the ocean), (2) *estuarine* (along tidal brackish waters), (3) *lotic* (along rivers and streams and subject to overflow), (4) *lentic* (in basins of lakes and reservoirs), and (5) *terrene* (sources of streams or isolated – completely surrounded by upland, or not significantly affected by the aforementioned waters). Figure 1 illustrates these positions in a simplified setting.

Landform describes the physical shape of the wetland. Seven types are recognized: *basin* (depressional wetland), *flat* (wetland on a nearly level plain), *floodplain* (overflow land along rivers subject to periodic inundation), *fringe* (wetland in water, within the banks of a river, or on an estuarine intertidal plain), *island* (wetland completely surrounded by water), *slope* (wetland on a hillside), and *peatland* (formations of organic soils mainly created by two processes: terrestrialization – the in-filling of lakes and other waterbodies and paludification – the blanketing of neighboring lands by peat moss followed by a succession of vascular plants).

Water flow path defines the direction of flow of water associated with the wetlands (Table 1 for nontidal paths; Fig. 2). If the wetland is a source of a stream, spring, or a seep, it is an *outflow* wetland. River and streamside wetlands are *throughflow* wetlands with water running through them (both into and out of) during high water periods. Wetlands that only receive water from channelized flow without any outflow are considered *inflow* wetlands. Some wetlands have no channelized inflow or outflow (i.e., lack an inlet or an outlet); water rises and falls in response to changes in precipitation, snow melt, local runoff, evapotranspiration, and groundwater interactions. Water movement in these seemingly isolated wetlands is described as *vertical flow*. Wetlands along lakes and reservoirs have water levels that rise and fall with lake levels and their water flow path is classified as *bidirectional-nontidal*, while lakeshore wetlands associated with streams are *throughflow* types. Water flow path can be further modified to indicate groundwater linkages: *groundwater-connected* (e.g., *Vertical flow-outflow/ground-water connected* or

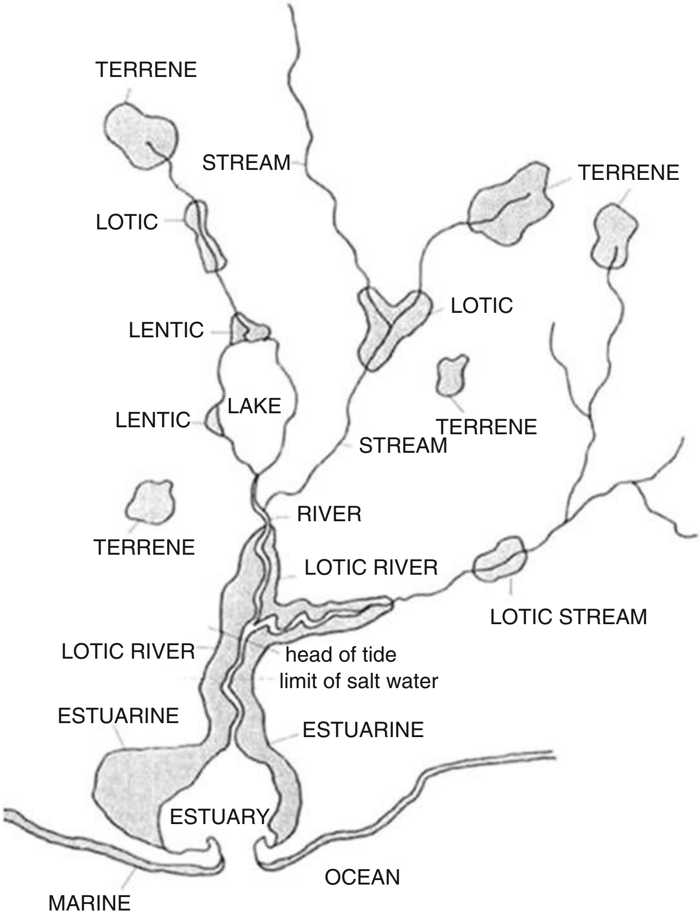

Fig. 1 General wetland landscape positions with a few waterbodies shown. Note: Streamside wetlands that are not overflowed but groundwater-dependent are classified as Terrene types, while the ones in this figure are depicted as subject to overflow, hence classified as Lotic (Source: Tiner 2014)

Vertical flow-throughflow/groundwater-connected where outflow or throughflow are accomplished by groundwater interactions such as observed in the Prairie Pothole Region of North America) or *groundwater lake-influenced* (e.g., *Bidirectional-nontidal/groundwater lake-influenced* for wetlands not directly connected to a large lake but where water levels are strongly influenced by lake levels such as experienced by wetlands formed on sandy soils along the Great Lakes in the USA and Canada). For tidal wetlands, five tidal ranges are emphasized since estuarine and marine wetlands are tidal by definition as are freshwater tidal wetlands: (1) *Nanotidal* (<0.3m), (2) *Microtidal* (0.3 to <2.0m), (3) *Mesotidal* (2.0 to <4m), (4) *Macrotidal* (4 to <8m), and (5) *Megatidal* (>8m; Woodroffe 2002). See Tiner 2014 for details.

Table 1 Brief definitions of nontidal water flow paths (Source: Tiner 2014)

Water Flow Path	Definition
Bidirectional-outflow	Water levels rise and fall with water in an outflow lake
Bidirectional-throughflow	Water levels rise and fall with water in a throughflow lake
Bidirectional-nontidal/ groundwater lake-influenced	Water levels change in response to lake level effects on groundwater
Throughflow-intermittent	Water enters from a water source above and flows out of the system via an intermittent stream; flow usually occurs during the wet season or during and shortly after heavy rains
Inflow	Water flows into an area with no surface flow outlet (a closed system); collected water is lost through evaporation, transpiration, and possibly groundwater recharge
Outflow-artificial	Water flows out of the system through a ditch or manmade channel; no direct surface water inflow
Outflow-intermittent	Water flows out of the system periodically usually during the wet season or during and shortly after heavy rains; no direct surface water inflow; typically associated with intermittent streams and groundwater discharge; may be the source of a stream
Outflow-perennial	Water flows out of the system year-round; no direct surface water inflow; typically associated with perennial streams, rivers and groundwater discharge; often the source of a stream
Throughflow-artificial	Water enters from a water source above and flows out of the system via a ditch or manmade channel or canal
Throughflow-perennial	Water flows through the system more or less year-round via a perennial stream; wetlands subject to seasonal overflow
Vertical Flow	Water levels affected by precipitation, local runoff and groundwater; no apparent surface water inlet or outlet

Uses of NWI+ Data

The NWI+ database is used to generate acreage summaries of wetlands and deep-water habitats grouped by the Cowardin et al. types and LLWW types (landscape position, landform, and water flow path) and to predict functions for the watershed's wetlands. Expanded wetland classification allowed for the relationship between wetlands and waterbodies and connectivity among wetlands to be characterized. For predicting wetland functions, relationships between properties in the NWI+ database and a variety of wetland functions were established (e.g., Tiner 2011). To date, 11 functions are routinely predicted from the NWI+ database: (1) surface water detention (for nontidal wetlands only), (2) coastal storm surge detention, (3) streamflow maintenance, (4) nutrient transformation, (5) sediment and other particulate retention, (6) carbon sequestration, (7) bank and shoreline stabilization, (8) provision of fish and aquatic invertebrate habitat, (9) provision of waterfowl and waterbird habitat, (10) provision of habitat for other wildlife, and (11) provision of habitat for unique, uncommon, or highly diverse wetland plant communities.

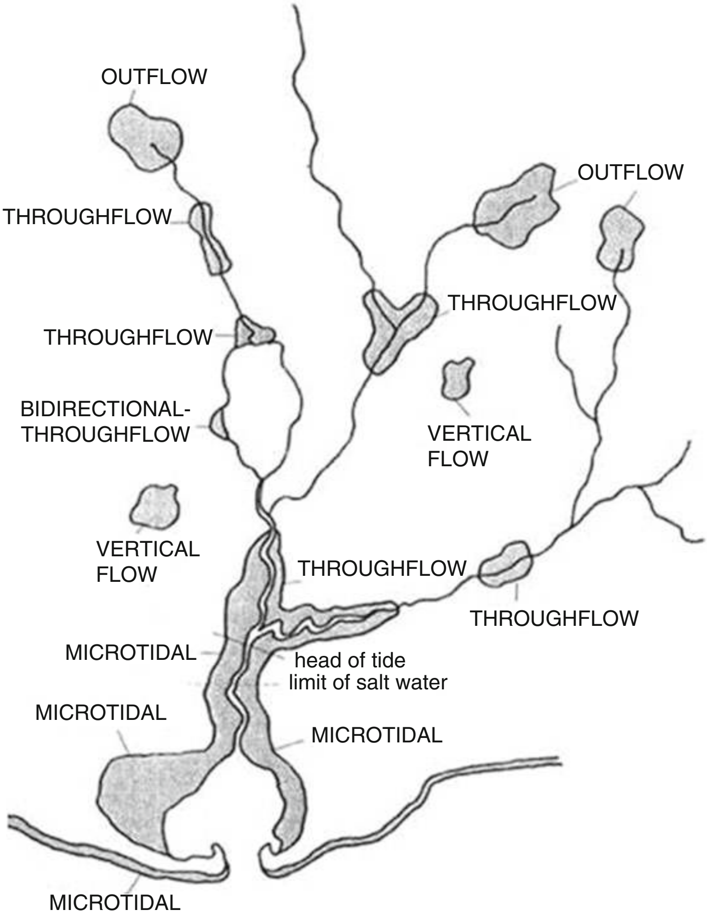

Fig. 2 General classification of common water flow paths across the landscape. Note: Direction of flow for nontidal areas is from top of page downward (Source: Tiner 2014)

Other functions can also be predicted if interested (e.g., provision of habitat for amphibians). Statewide functional assessments of wetlands have been performed for Connecticut, Delaware, and Rhode Island, with work in progress for other states and large geographic areas (see NWI+ reports online at: http://www.aswm.org/wetland-science/wetlands-one-stop-mapping/5044-nwi-reports).

When analyzing changes in wetlands over time (wetland trend studies), adding the LLWW descriptors to the losses and gains can be used to help predict the impact of these changes on functions. This type of analysis has been done for areas of variable size including, for example, the state of Connecticut from 1990–2010 (12,548 km^2; Tiner et al. 2013b), coastal Georgia (8,182 km^2; Georgia DNR 2012), Montana's Flathead Valley (3,740 km^2; Newlon and Burns 2010), and the Nanticoke watershed (Delaware/Maryland; 2,070 km^2; Tiner 2005).

Products

The results of LLWW applications are published in a series of technical reports with the geospatial data uploaded to an online mapping tool – NWI+ Web Mapper – using ESRI's ArcGIS online mapping service. Both the NWI+ reports and geospatial data are available online via the Association of State Wetland Managers' Wetlands One-Stop Mapping website: http://aswm.org/wetland-science/wetlands-one-stop-mapping. Geospatial data layers include classifications of wetlands by NWI types (Cowardin et al. 1979), landscape position, landform, water flow path, and by their predicted potential to provide various functions. The online mapper allows users to zoom into specific areas of interest and thereby see more detail than could be provided by producing maps for a report. Moreover, the tool permits the user to display the data on aerial imagery or topographic or planimetric maps and to produce custom maps for use in reports or for other purposes. The geospatial data produced for these projects allow for other geographic analyses (e.g., smaller watersheds, counties, towns, and other areas of special interest).

Future Challenges

To date, the classification has been applied mostly to temperate humid regions of the United States. More widespread application of the classification is needed in other regions. The State of New Mexico is actively applying the system to wetland mapping across the state, while the State of Montana continues to use the system in its updated wetland mapping efforts. The State of Minnesota will soon be applying it to its updates of NWI data. Michigan continues to work with other groups within the state in seeking statewide coverage. In 2015, the system will be applied to Alaska's Kodiak National Wildlife Refuge to provide a good test of the classification in the Arctic. Over the next couple of years, further refinements in the classification may be adopted based on these applications. It is hoped that other countries seeking to use wetland inventory data for management purposes will explore the use of these descriptors to enhance their wetland databases and to use such data to predict wetland functions at the landscape level.

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Wetland Classification in India

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Brij Gopal

Contents

Abstract

Wetlands in India have been classified using several different approaches which recognise a number of types whereas a hierarchical phylogenetic classification based on detailed inventory of wetlands and their characteristics has yet to be elaborated.

Keywords

Littoral and Swamp forests · Mangroves · Salt pans · Lagoons

Introduction

Although many classification schemes have been proposed at global, regional, and national levels, researchers as well as managers and policy makers often prefer to use a simple typology that groups all wetlands into a large number of broad "types" which are sometimes combined into a few larger categories. Such schemes are "artificial" classifications or typologies, whereas truly hierarchical or phylogenetic

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classifications reflect the relationships among different wetland types. However, the latter are not that common (e.g., Gopal et al. 1990). Hierarchical classification schemes are generally based on detailed inventories of wetlands along with an analysis of their major characteristics and driving variables.

In India, the term wetland appeared in the scientific literature only in 1973 (Gopal 1973) although there were many earlier studies on mangroves, marshes, and swamps. Champion (1936) recognized swamps within various forest types with Champion and Seth (1968) later recognizing "Littoral and Swamp Forests" as a major forest type of India and further classifying it into four subtypes (Tidal, Tropical Freshwater, Tropical Seasonal, and Tropical Riparian Fringing forests). The first detailed description of Indian wetlands was made by Gopal (1982) and further elaborated by Gopal and Krishnamurthy (1993). The Asian Wetland Directory undertaken in the late 1980s recognized only 22 wetland types and included 93 prominent wetlands from India, comprising mainly larger shallow water bodies, reservoirs, tanks, marshes, and mangroves, but not the freshwater swamps (Wolstencroft et al. 1989). WWF-India revised and updated the Indian section of the Directory, with information on 77 additional wetlands (WWF-I and AWB 1993). A distinction was made in India from the very beginning between natural and human-made wetlands (see Biswas 1976).

Indian Classifications

Gopal and Sah (1995) proposed the first classification of Indian wetlands. These were first grouped into saline and freshwater types and further separated on the basis of their hydrological regime, particularly the duration of flooding (permanent or seasonal), which directly controls the type of vegetation in a wetland. Wetland types were then divided into those with herbaceous or woody vegetation and further subdivided by their dominant vegetation. It was pointed out that even permanently flooded wetlands exhibit large water level changes, and a large part of their area may become exposed for a short period. Further, more than one vegetation type could occur in a wetland, but it could be categorized by the dominant vegetation type.

This scheme did not find favor with policy makers and managers who preferred to use the Ramsar typology – a number of types only which can be readily employed by nonspecialists without taking into account the hydrology. Prasad et al. (2002) observed that from a practical conservation planning perspective, a reasonably detailed classification based on a mix of habitats and aquatic vegetation was needed that could be used by both managers and academicians. They suggested that inland wetlands could be classified into: A. Ponds/Tanks/ Lakes; B. Reservoirs; C. Waterlogged; and D. Oxbow lakes/Cutoff meanders which were further divided on the basis of turbidity and aquatic vegetation cover. They recommended using remote sensing data for this purpose. The latter could have been done by referring to

Table 1 Wetland classification system used by Space Application Centre in India (Garg et al. 1998; Garg and Patel 2007)

Level I /Level II	Level III (Garg et al. 1998)	Modified level III (Garg and Patel 2007)
Inland wetlands		1000
Natural		1100
	1.1 Lakes/ponds	1101 Lakes
	1.2 Ox-bow lakes/ cut-off meanders	1102 Ox-bow lakes/ cut-off meanders
	1.3 Waterlogged wetlands	1103 High altitude wetlands
	1.4 Playas marshes/swamps	1104 Riverine wetlands
	1.5 Swamp/marsh	1105 Waterlogged
Man-made		1200
	2.2 Reservoirs	1201 Reservoirs/barrages
	2.2 Tanks	1202 Tanks/ponds
	2.3 Waterlogged	1203 Waterlogged
	2.4 Abandoned Quarries	1204 Salt pans
	2.5 Ash pond/cooling pond	
Coastal wetlands		2000
Natural		2100
	3.1 Estuary	
	3.2 Lagoon	2101 Lagoons
	3.3 Creek	2102 Creeks
	3.4 Back water (Kayal)	2103 Sand/beach
	3.5 Bay	2104 Intertidal mud flats
	3.6 Tidal flat/mud flat	2105 Salt marsh
	3.7 Sand/beach/spit/bar	2106 Mangroves
	3.8 Coral reefs	2107 Coral reefs
	3.9 Rocky coast	
	3.10 Mangroves	
	3.11 Salt marsh/vegetation	
	3.12 Other vegetation	
Man-made		2200
	4.1	2201 Salt pans
	4.2	2202 Aquaculture ponds

the fairly comprehensive assessment of wetlands undertaken by the Space Application Centre (Ahmedabad) using satellite images (Garg et al. 1998). This assessment contained 22 categories of wetland habitats and was later modified after another detailed mapping exercise which covered the entire country at a 1:50,000 scale (Garg and Patel 2007). The two classification schemes are given in Table 1.

Concluding Remarks

The above shows the development of wetland classification in India with the use and acceptance of a number of different approaches. This is not uncommon with wetland classification and supports the recommendation that classification should be systematic and directed towards specific purposes (Finlayson and van der Valk 1995).

A few further observations show that while it is common practice to group wetlands into inland and coastal wetlands, the actual differentiation is not always straightforward. The term “coastal wetlands” is generally used to refer to wetlands that are directly influenced by the ocean despite being landwards of the coastline and often above sea level. They may also be classified as being within a specific distance from the coast. Coral and limestone reefs which can be close to or far away from the coastline are also often included in the coastal category, whereas beds of seagrasses and kelps (marine algae) in shallow coastal waters are not always included. Systematic classification including clear statements about what is included or excluded has been recommended for many years in order to remove uncertainties and enable the veracity of comparisons to be ascertained.

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Brazilian Wetlands: Classification

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Contents

Abstract

Brazil contains a large number and variety of wetlands. Pronounced wet and dry seasons lead to strong fluctuations in the water levels of streams and rivers. Riparian zones and large river-floodplains as well as poorly drained, extended, flat interfluvial areas are periodically flooded and dry according to the precipitation pattern. A definition of wetlands is proposed. Basing on the floodpulse concept, the outer wetland borders are defined according to the mean maximum flood level.

The wetland delimitation also includes permanent terrestrial macrohabitats as intrinsic parts of large wetlands, essential for the maintenance of their ecological integrity and biodiversity. The peculiar hydrological conditions in the wetlands account for the conflicting positions of the different stake holders.

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The new Brazilian Forest Code (Federal Law no. 12.561/12), responding to the pressure of agrobusiness and planners, defines the protected area of wetlands depending on the width of the river channel at the "regular water level." As a result, large parts of the fringing floodplains and riparian zones are unprotected. The destruction of the wetlands will have dramatic economic and social consequences for the people, living in and around the wetlands because of the reduction of the hydrological buffer capacity of the river-wetland systems. Furthermore, the diversity of macrohabitats inside the wetlands and the related biodiversity will be destroyed. A hierarchical wetland classification system, based on hydrological parameters and higher plants, has been proposed and a macrohabitat classification of major wetland systems has been elaborated to provide a scientific basis for the sustainable management and protection of Brazilian wetlands.

Keywords
Brazilian Wetlands · Definition · Delimitation · Classification

Introduction

The Brazilian territory covers 8,514,215 km^2, corresponding to 47% of the South American subcontinent. The country is situated in the tropical and subtropical lowlands north and south of the equator. Over large areas, the relief is flat and high precipitation rates lead to the development of extended wetlands. With the signing of the Ramsar convention in 1973, the Brazilian government not only accepted the responsibility to designate specific Ramsar sites, but also to delineate its wetlands and to elaborate plans for the wise use of these areas.

Since then 11 Ramsar sites have been designated, covering a total area of 65,684 km^2. However, wetland delineation and classification are still developing as became clear in recent years, when Brazilian politicians questioned the Forest Code which regulates the protection of natural vegetation along rivers and streams. The old version of this code used the width at maximum flood level to determine the protected area. In the new version of the Forest Code agrarian pressure groups forced the government to use the width of streams and rivers at low water level, allowing the destruction of most of the riparian vegetation for agricultural purposes and eliminating the protection of riverine wetlands.

Warnings by wetland scientists about the negative impacts of wetland destruction on nature and society were not considered, in part with the argument that no definitions and delineation system were available for Brazilian wetlands (Sousa et al. 2011; Junk et al. 2012a; Piedade et al. 2012). In August of 2012, a group of Brazilian wetland scientists joined forces to elaborate a definition and classification system for Brazilian wetlands, considering the specific hydrologic and botanic conditions of the country. This paper introduces the background and details of the Brazilian approach to wetland classification.

Rationale for the Brazilian Approach

There are many definitions and classifications of wetlands as summarized by Mitsch and Gosselink (2008). Frequently cited are those of the Ramsar Convention (Scott and Jones 1995) and of the US Forest and Wildlife Service (Cowardin et al. 1979). Hydro-geomorphic approaches were proposed by Brinson (1993) and Semeniuk and Semeniuk (1995). The problems arising during the elaboration of classification systems were discussed by Finlayson and Van der Valk (1995), who pointed out the necessity of resolving differences between regional wetland definitions and regional typologies. They also drew attention to the need to standardize data collection and disseminate new technologies in order to establish ample international inventories. Indeed, many definitions and classification systems were formulated decades ago for specific purposes and do not correspond to current scientific and regulatory requirements.

For Brazilian wetlands, the primary objective is to provide a scientific basis for politicians and decision-makers to elaborate wetland-specific policies and legislation. A second objective is to offer a hierarchical classification system which provides scientists in Brazil with the means to position their wetland studies within broader national and international contexts. This would facilitate comparisons between different wetlands and improve understanding of the (dis)similarities among them.

The scientific basis for the Brazilian classification is the Flood Pulse Concept (Junk et al. 1989). The concept describes the impact of periodic flooding and drought on biogeochemical cycles, production and decomposition of organic matter, species diversity and life history traits of plants and animals in the Aquatic Terrestrial Transition Zone (ATTZ), and the exchange of water, nutrients, and organisms between the parent rivers and the ATTZ.

Developers and politicians regard the entire ATTZ as permanently dry habitats in which natural flooding is not considered an inherent attribute of the system. This view has promoted the encroachment of conventional agriculture and infrastructure development into wetlands, as was emphasized during recent disputes in the Brazilian Parliament over the new Forest Code (Piedade et al. 2012). In Europe and the USA, this lack of recognition for the wetland status of large river floodplains facilitated the exploitation of these areas as cropland protected by dikes with far-reaching adverse consequences for the flood regime, nutrient cycles, and habitat and species diversity (Mitsch and Day 2006).

Brazilian Wetland Definition, Delineation, and Classification

Because of the periodicity of rainfall, most Brazilian inland wetlands undergo large water level fluctuations. Up to 90% of the wetland area is dry during several months every year. This flood-pulsing is a common hydrological feature of tropical and subtropical river floodplains, interfluvial wetlands, and riparian wetlands worldwide

Table 1 Types of flood pulses and affected wetlands

Predictability	Frequency	Amplitude	Affected wetland type
Predictable	Monomodal	High	Floodplains along large rivers
		Low	Large interfluvial wetlands, coastal lagoon systems in large dune fields (e.g., Lencões Maranhenses)
Predictable	Polymodal	Variable	Coastal wetlands influenced by tides
Unpredictable	Polymodal	Variable	Wetlands along streams and small rivers, in small depressions and between coastal dunes
Unpredictable	Multiannual	Low	Wetlands in the north-eastern semiarid zone

(Junk et al. 1989). Coastal wetlands are subjected to the flood pulses of the tides. Wetlands with a rather stable water level are common in temperate zones (e.g., peat bogs, mires) but rare in the tropics and subtropics. Examples are palm swamps in South America and swamp forests in South-East Asia.

Because of the flood-pulsing, the determination of parameters for wetland delineation is a critical problem, which creates many controversies in practice. Under specific hydrological conditions, the occurrence of hydric soils (organic matter accumulation as a result of permanent shallow inundation or long-term waterlogging) can be a good parameter for the delineation of wetland boundaries. However, flood-pulsing wetlands with long terrestrial phases do not exhibit accumulation of organic matter as periodic aeration facilitates the decomposition of organic matter. Thus, by exclusively focusing on hydric soil indicators, we fail to achieve wetland delineation and protection in regions where there is a pronounced seasonal flood pulse. The occurrence of plant species adapted to water or waterlogged soils is frequently used for the determination of wetland boundaries, but in flood-pulsing wetlands annual wetland plants often occur only during the flood period.

Hydrology is the most important factor determining wetland characteristics and is given highest priority in the Brazilian classification. The large number of wetlands with oscillating water levels requires that greater emphasis be placed on the different types of flood pulses, which are under-represented in all of the classification systems discussed above. According to hydrologists, the active floodplain of a river can be defined as the area covered by water during a 100-year flood (Bhowmik and Stall 1979). This points to the importance of long hydrological time series for wetland delineation. A classification of flood-pulsing Brazilian wetlands is given in Table 1. While the hydro-geomorphic arguments provided by Brinson (1993) and Semeniuk and Semeniuk (1995) are very helpful from a scientific point of view, they contribute little to the continuing political discussion on wetland management in Brazil. Some Brazilian wetlands cover several tens of thousands of square kilometers. These large wetlands are characteristic landscapes with complex habitat arrangements and have a specific status as classes in the classification system. Many of them include permanently terrestrial habitats which are essential for the maintenance of biodiversity and have to be included in environmental protection plans.

This leads to the following definitions of "wetlands" and "wetland delineation" in Brazil (Junk et al. 2014):

Definition of wetlands:

- Wetlands are ecosystems at the interface between aquatic and terrestrial environments; they may be continental or coastal, natural or artificial, permanently or periodically inundated by shallow water or consist of waterlogged soils. Their waters may be fresh or highly or mildly saline. Wetlands are home to specific plant and animal communities adapted to their hydrological dynamics.

Wetland delineation:

- The extent of a wetland can be determined by the border of the permanently flooded or waterlogged area, or in the case of fluctuating water levels, by the limit of the area influenced during the mean maximum flood. The outer borders of wetlands are indicated by the absence of hydromorphic soils and/or hydrophytes and/or specific woody species tolerant to periodically or permanently flooded or waterlogged soils. The definition of a wetland area should include, if present, internal permanently dry areas as these habitats are of fundamental importance to the maintenance of the functional integrity and biodiversity of the respective wetland.

The Brazilian classification of wetlands is segregated into three levels: (1) systems; (2) units defined by hydrological parameters (subsystems, orders, suborders, classes, functional units); and (3) units defined by higher plants (subclasses and macrohabitats).

System 1: **Coastal wetlands** are defined as all wetlands, permanent or temporary, with fresh, brackish, or saline waters, under direct influence of the tides, or subject to saline intrusions, or influenced by the atmospheric deposition of dissolved or particulate substances and/or propagules from the ocean.

System 2: Inland wetlands are defined as all wetlands, permanent or temporary, with fresh, saline, or salt water, that are located in the Brazilian inland and are thus without direct or indirect marine influence.

System 3: Artificial wetlands are all wetlands, coastal or inland, derived from human activities either in organized (e.g., fish farms, rice paddy plantations) or unorganized forms (wetlands around reservoirs or those that progressively develop by the damming of streams or that form in depressions caused by the excavation of soil for road construction, etc.).

Subsystems divide coastal wetlands into wetlands subjected to (1) predictable tidal pulses, (2) unpredictable pulses, and (3) stable water level, and inland wetlands into wetlands with (1) stable and (2) pulsing water level. Most Brazilian wetlands belong to the subsystem of inland wetlands with pulsing water level, which requires further subdivision. Orders include inland wetlands with (1) predictable, monomodal long lasting pulses, (2) unpredictable, polymodal pulses of short duration, and

(3) pluriannual short pulses. Suborders include wetlands with predictable, monomodal long lasting pulses of (1) high amplitude and (2) low amplitude.

Both suborders include several classes. These classes include very large wetland landscapes, such as the large Amazonian river floodplains, the Pantanal of Mato Grosso, and the Araguaia and Guaporé River wetlands. The annual hydrologic dynamics and the morphologic heterogeneity require a subdivision of the classes into five functional units to better describe the impact of the monomodal predictable flood pulse on the macrohabitats. This approach was also used in the classification of Amazonian inland wetlands (Junk et al. 2011). The functional units reflect the different hydrological conditions: (1) permanently aquatic, (2) periodically aquatic, (3) periodically terrestrial, (4) permanently terrestrial, and (5) swampy. A sixth functional unit includes areas, strongly modified by man.

All coastal and inland wetlands show numerous subclasses and many macrohabitats, which represent the lowest category in the classification system. They are characterized by hydrological parameters and by specific vegetation types and/or indicator species. The subdivision of artificial wetlands is restricted to macrohabitat level. For management purposes, local terms were included in the macrohabitat classification as was done by Gopal and Sah (1995) for Indian wetlands. This approach is likely to be beneficial because it increases the local population’s willingness to accept the imposed regulations for the sustainable management and protection of wetlands.

Major gaps still exist at the subclass and macrohabitat level of the large inland wetland landscapes. Classification systems are already available for the Pantanal of Mato Grosso (Nunes da Cunha and Junk 2011) and for the Amazonian white water river floodplains (várzeas; Junk et al. 2012b)). Classifications of the Amazonian black water river floodplains and the floodplains of the Paraná, Araguaia and Guaporé rivers are in progress.

Conclusions

The Brazilian approach to define, delineate, and classify wetlands is a practical one. It differentiates between coastal, inland, and artificial wetlands as do other classification systems. But it considers the fact that most wetlands have an oscillating water level and that higher vegetation responds to the different types of flood pulses and levels of inundation. Furthermore, it classifies the large variety of macrohabitats in very large wetland complexes, which reach from permanent aquatic to permanent terrestrial conditions and are all essential for the maintenance of biodiversity.

Human impact on wetlands often starts with destruction of the natural vegetation cover, for example, by timber extraction, cattle ranching, and crop plantations, which in turn cause changes in the hydrological regime through water abstraction, drainage, flood control, and reservoir construction and is inevitably followed by inappropriate civil construction. These steps can be monitored by remote-sensing techniques and the consequent measures required for wetland protection, including proposals for sustainable management, can easily be explained to politicians, planners, decision-makers, and the public. Many parameters used in this approach are also used in other classification systems, however at different hierarchic levels. Therefore, the Brazilian classification allows the worldwide comparison of wetlands with similar hydrologic characteristics.

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The Canadian Wetland Classification System

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Clayton Rubec

Contents

Abstract

Until the mid-1970s, efforts to classify and map Canadian wetlands were mostly uncoordinated with a diversity of approaches, goals, and priorities. In 1976, a Canadian Wetland Classification System (CWCS) was initiated which evolved into the Federal Policy on Wetland Conservation in 1991. The CWCS has developed in close association with wetland policy in Canada since then and is based on a three-level classification: five wetland classes (bog, fen, swamp, marsh, and shallow waters); wetland forms based on surface morphology, surface pattern, water type, and underlying soil morphology; and an open-ended number of wetland types based on physiognomic characteristics of vegetation communities. The "wetland type" level recognizes that field-level wetland classification and mapping requires practical local experience and input. The currently 49 wetland types used in the CWCS reflect vegetation communities, such as shrub, treed, graminoid, moss, lichen, and aquatics. This level allows regional approaches to site-level wetland classification to be part of the system.

Keywords

Wetland classification · Wetland policy · Wetland descriptors · Canadian Wetland Classification System · Wetland mapping

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Introduction

Canada is as diverse politically as are its wetlands. About 40% of wetlands are under federal jurisdiction in Canada's northern territories and in hundreds of national parks, national wildlife areas, bird sanctuaries, and federal lands such as military bases. The other 60% are under the jurisdiction of the ten provincial and three territory governments. Unlike the United States of America, the federal government has no constitutional authority to directly regulate all of the wetlands in the nation. Thus, any approach to this concept in Canada requires cooperation across borders and jurisdictions.

The evolution of the Canadian approach to wetland classification grew out of a series of regional mapping initiatives in sectors such as soil type and land capability, forestry production, and aquatic ecosystem classification. Wetland mapping projects up to the mid-1970s examined managing ecotypes such as prairie grassland wetlands, forested peatlands, migratory bird habitat, and coastal salt marshes. Others focused on geographic units: marshes in Nova Scotia, wetlands in the Lancaster Sound Northwest Territories shipping corridor, peatlands in southern Quebec, shorezone wetlands in the Great Lakes-St. Lawrence River areas of Ontario and Quebec, wetlands in the urban-agricultural landscapes of British Columbia, and selected areas of prairie marshes. A string of publications and wetland classification efforts were created, reflecting the evolving focus on wetlands across Canada. But these many projects worked in isolation of one another, and over several decades, starting in the mid-1970s. They reveal significant differences in government priorities (such as bird habitats, engineering soil properties, cleanup of oil spill shipping disasters, urban growth) and regionally favorite, poorly defined terminology, lack of consensus on process, and an inability to readily translate ideas and terms between English and French.

A National Approach

In 1976, in an effort to bridge the significant problems of not having a national, unified approach to wetlands, a National Wetlands Working Group (the "NWWG") was created with a fluid, interdisciplinary core membership drawing on geographic diversity and significant scientific experience. It was a committee that operated through consensus with no legislative authorities and the slimmest of financial support. Participants attended on the basis of having support from their agencies rather than from a centrally provided budget. Each member brought their own networks of wetland expertise. Thus, while the total number of direct and indirect participants exceeded 100 wetland specialists nation-wide, only about ten individuals actively participated each year. In its 25 years of activity, this group's members varied year over decade, held field excursions to study wetland types in all ecoregions of Canada, met regularly, and busied themselves with an important set of projects. It was led by a Chairperson, who changed about every five years, with a National Secretariat Coordinator.

The First Edition of the *Canadian Wetland Classification System* (the "CWCS") (National Wetlands Working Group 1976) reflected consensus and was a sister project to two National Atlas of Canada initiatives: broad maps of the *Wetland Regions of Canada* and *Wetland Distribution in Canada*. This was done in concert with a major book project that saw the release of *The Wetlands of Canada* (National Wetlands Working Group 1988). The CWCS became a focal point for wetland conservation policy development by several governments in Canada. It was specifically adopted by the Federal Cabinet as a definitional source document for the creation of *The Federal Policy on Wetland Conservation* (Government of Canada 1991). It was accompanied by a long series of interpretive publications and an *Implementation Guide for Federal Land Managers*. Canada thus became one of the first nations to adopt a national-level wetland policy as envisaged by the Ramsar Convention on Wetlands.

Most provinces in Canada followed the federal initiative with wetland policies of their own, all with two common threads: environmental conservation objectives and reduction in the loss of wetland areas in each jurisdiction. This group of federal and provincial policies continues to be used widely by many agencies in scientific studies, environmental assessment projects, land management, and natural resources development. A series of wetland policies were also adopted by private sector organizations representing natural resources development companies. Examples include the oil and gas sector, forest management agencies, and peatland harvesting corporations.

The Canadian Wetland Classification System (CWCS)

After the release of the First Edition of the CWCS, a great deal of additional work went into the Classification System. A fully reorganized document with excellent diagrams, classification keys, and defined terminology was released over 20 years later as its Second Edition (Warner and Rubec 1997) in English through a joint effort of Environment Canada and the Wetlands Research Centre at the University of Waterloo. All of the documents named above are available in both English and French versions. This required a great deal of effort to standardize words and definitions.

The CWCS is based on the adoption of a three-level classification: (a) five **wetland classes** (bog, fen, swamp, marsh, and shallow waters), (b) **wetland forms** based on surface morphology, surface pattern, water type, and underlying soil morphology, and (c) an open-ended number of **wetland types** based on physiognomic characteristics of vegetation communities (Fig. 1). The first (top) level of "wetland class" is essentially a unifying grid overlying all regional and topic-specific wetland classifications in Canada that preceded the national effort. The third (lowest) level of this classification system, "wetland type," recognized that field-level wetland classification and mapping required a great deal more practical experience and input. Each of these wetland types reflected vegetation communities, such as shrub,

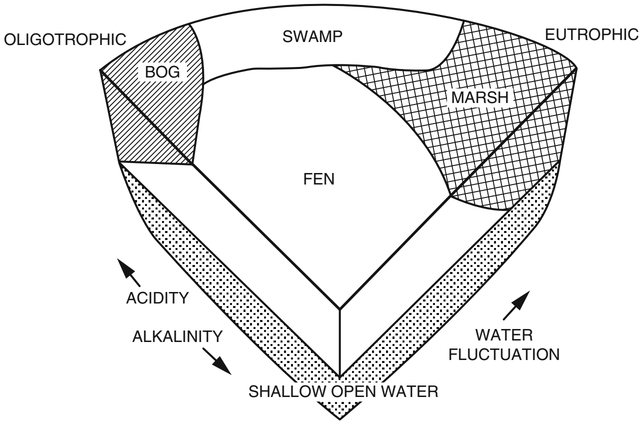

Fig. 1 The five classes of the Canadian Wetland Classification System in relation to the important chemical gradients. The vertical axis is a qualitative expression of the influence of water depth (Based on Zoltai and Vitt (1995) Reproduced with permission)

treed, graminoid, moss, lichen, and aquatics. It is at this level that many regional approaches to site-level wetland classification take over.

In the First Edition in 1976, there were 68 wetland forms defined. However, in the Second Edition in 1997, this list was simplified into a new set of 49 wetland forms. Each is discussed and defined in the text and has illustrative diagrams, classification keys, and photographs to explain each to the reader. Two appendices complete the text. Appendix 1 provides terminology crossovers between the first and second editions as well as the Ramsar Convention's descriptive wetland terminology as adopted to that point in time. Appendix 2 provides a listing of each wetland class and form in English and the adopted terms in French (in Canada).

Thus, a wetland in the CWCS approach is based on three sets of words, with the following as an example: "Treed Riverine Fen".

- Class: Fen
- Form: Riverine.
- Type: Treed.

An important feature of the development of the classification was its close association with the development of wetland policy in Canada. A review paper on the Canadian experience with wetland regulation was released in 2005 (Rubec and Hanson 2008); however, a national overview paper on the current status of wetland policy efforts in Canada has not been undertaken since a major report by Lynch-Stewart et al. (1999).

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Section XV

Earth Observation Methods for Wetlands

Richard Lucas

Earth Observation Methods for Wetlands: Overview

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Richard Lucas

Contents

Abstract

Across their range, wetlands are highly complex and dynamic and have been observed by a wide and diverse range of ground, airborne, and spaceborne sensors. The methods applied for characterizing, mapping, and monitoring mangroves are therefore diverse but have focused primarily on mapping state (i.e., water, ice, or snow) and extent as well as persistence and duration, sediment loads, substrate characteristics, and tidal fluctuations. A number of indices, algorithms, and models have been specifically developed to understand the changing states of wetlands, with these including mangroves, sea grasses, bogs, mires and fens, tropical floodplains, and semiarid wetlands. Many wetlands are also subject to anthropogenic disturbance as well as natural events and processes. Remote sensing data provide a unique opportunity to track such changes but also to classify these according to the different disturbance types. A number of international programs have also been put in place to advance the use of remote

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sensing data for wetland observations, with these including the European Space Agency's (ESA) Globwetland (I, II), the Japan Aerospace Exploration Agency (JAXA's) Kyoto and Carbon (K&C) Initiative, and the NASA's Making Earth System Data Records for Use in Research Environments (MEaSUREs) projects.

Keywords
Remote sensing · Water states · Biophysical characteristics · Human disturbance · International projects

Regions of the Electromagnetic Spectrum

At local to global scales, wetlands can be observed, characterized, mapped, and monitored using a diverse range of ground, airborne, and spaceborne sensors operating in different modes and across different spatial and temporal scales. Sensors that are generally more familiar to those involved with wetlands assessment operate in the *spectral* (reflected visible to shortwave infrared) regions of the electromagnetic spectrum, with these allowing identification of open water, determination of water state and quality, discrimination of different aquatic environments and vegetation types, and tracking of vegetation phenology and water dynamics. Sensors operating in the *thermal* regions provide information on the temperature variations of wetlands and particularly the water surface. *Microwave* sensors (on the order of cm wavelength) typically facilitate the mapping of open water and inundation and also provide information on the three-dimensional structure of wetland vegetation.

Main Sensor Types of Relevance

For wetland mapping, key sensors include the Landsat series, which has provided a global record of wetland characteristics and extent through observations at 30–80 m spatial resolution from the late 1970s onwards (Williams et al. 2006; Markham and Helder 2012). With the recent release of this archive, considerable opportunities for understanding the dynamics of wetlands are provided. Sensors operating at coarser spatial resolution have included the NOAA's Advanced Very High Resolution Radiometer (AVHRR) and the Terra-1 Moderate Resolution Imaging Spectroradiometer (MODIS), with these being particularly beneficial for mapping the dynamics of flooding over large areas through regular (at least daily) observations over several decades (Takeuchi et al. 2003; Yan et al. 2010). These and the Landsat sensors also provide observations in the thermal region. More detailed assessment of wetlands have been conducted using very high resolution (VHR) optical sensors such as Worldview, Quickbird, and IKONOS (e.g., Laba et al. 2008; Dillabaugh and King 2008; Magumba et al. 2014; Salari et al. 2014), which are increasingly available at a global level, with the constellation of five RapidEye sensors providing higher frequency (daily) observations over smaller areas. Sensors on board

Unmanned Airborne Vehicles (UAVs) are also being used more regularly for observing wetland areas in detail and on demand (Jensen et al. 2011).

Observations by many optical and also thermal sensors are limited by cloud. For this reason, Synthetic Aperture Radar (SAR) operating in the microwave region has commonly been exploited, with these also providing information on vegetation types and the extent of inundation. Different levels of information can be obtained by using data acquired at different frequencies and polarizations, with C-band (~6 cm) SAR facilitating discrimination of herbaceous vegetation and L-band SAR allowing detection of water under vegetation, including that which is woody (Hess et al. 2003; Silva et al. 2008). Airborne Light Detection and Radar (LIDAR) has also being used to provide detailed three-dimensional, high resolution representations of the terrain surface and surface features, including vegetation (Cook et al. 2009).

Remote Sensing of Water Dynamics

A wide range of remote sensing methods are available to determine the physical state of water (liquid, snow or ice) and water column characteristics and to quantify patterns of inundation and extent, persistence and duration, sediment loads, substrate characteristics, and tidal regimes.

Using spectral wavebands, water can generally be distinguished from ice and snow (Dozier 1989). Open water itself is best mapped using thresholds or classifications that utilize the near infrared and shortwave infrared bands or derived indices (e.g., the Normalized Difference Water Index). At SAR frequencies, water exhibits a relatively low backscattering coefficient allowing discrimination from nonwater surfaces. However, confusion with rough water and macrophytes often occurs (Costa and Telmer 2006).

Water column characteristics are best retrieved through spectral analysis, with the visible channels providing best opportunity for retrieval. Empirical relationships with measured properties or bio-optical models are most commonly used for retrieval (e.g., Zhang et al. 2008).

Depending upon the cover of vegetation and depth of water, the extent of *inundation* can be mapped using combinations of remote sensing data, with SAR providing the best opportunity for mapping inundation beneath vegetation. The combination of satellite-derived gravity and altimetry data has also been used to quantify inundation and water dynamics (Alsdorf et al. 2007).

Detection also depends upon the frequency of coverage. By comparing maps of inundation extent, information on the *persistence and duration* of water within wetland systems has been obtained (Milne and Tapley 2004). The MODIS sensor, for example, provides excellent mapping of open water over large areas and on a frequent (i.e., daily basis). However, other sensors such as the Landsat provide observations every 16 days (cloud cover permitting) and hence temporary bodies of water may go undetected. The repeat coverage of many SAR is also relatively low (e.g., 44 days for the ALOS PALSAR).

Sediment loads and types have been mapped primarily by considering the reflectance characteristics in the visible regions (Stumpf and Goldschmidt 1992) and, when the water is optically clear, *substrate cover types* and their properties have also been retrieved (Dekker et al. 2005). Tidal regimes have often been inferred through reference to inundation detected under mangroves or maps of tidal flats (e.g., Murray et al. 2012).

Remote Sensing of Natural and Seminatural Wetlands Types

A wide range of remote sensing techniques have been developed since the beginning of earth observation from the ground and by airborne and spaceborne sensors with these often being specific to particular regions and wetland types.

In the *arctic and boreal regions*, SAR data have often been preferentially used because of the prevalence of cloud. A range of techniques has been used to differentiate wetland types, including rule-based classifications. Using Alaska as an example, elevation and slope data obtained from the Shuttle Radar Topographic Mission (SRTM) have been combined to identify areas where wetlands occur, with a Random Forests algorithm then applied to provide more detailed classifications of wetland types (Whitcomb et al. 2014). For more mapping of wetland dynamics, object-based classifications of time-series of high to moderate resolution data based on supervised methods or decision trees have been used (Tehrany et al. 2014).

In the *temperate regions* of Europe and North America, many wetlands have been fragmented through human activity and typically moderate resolution to VHR remote sensing data have provided the level of detail needed for their mapping and monitoring. In these regions, consistent mapping of seminatural and natural habitats and their differentiation from those that are cultivated, managed, or artificial is desirable. The classification schemes ideally need to be consistent within and between sites. One such taxonomy is the Food and Agricultural Organisation's (FAO) Land Cover Classification Scheme (LCCS) (Di Gregorio and Jansen 2000), which has been applied to wetland habitats in protected areas and their surrounds at sites across Europe (Lucas et al. 2014a). A particular advantage of using taxonomies such as the LCCS is that they can be applied at any scale and use a diversity of remote sensing inputs.

In the *semiarid regions*, wetlands are often periodic and water may inflow from regions within different climatic zones. In these cases, wide swath ScanSAR data and MODIS data are useful as more frequent observations of wetlands systems are obtained with these allowing better detection of inundated areas and water flows (Bartsch et al. 2009; Moser et al. 2014). Often, data from optical sensors such as MODIS and Landsat sensor data can be used to capture the vegetation response to environmental flows (Shaikh et al. 2010). Techniques used have included simple thresholding, supervised/unsupervised classifications, and machine learning (Ozesmi and Bauer 2002; Liu et al. 2008).

In the *tropical regions*, flooded forests are extensive with significant tracts occurring in the Amazon and Congo river basins (Melack and Hess 2010; Mayaux et al. 2002). Within these areas, the dynamics of flooding has a large influence on biogeochemical and nutrient cycles as well as on biodiversity distributions. To map the extent of inundation beneath the forest canopy, the use of L-band SAR has proved useful (Hess et al. 1995), with time-series allowing the changes in flooding patterns to be discerned. Through knowledge of the extent and condition of wetlands and how these have changed over time, better information on the changing distribution of energy and gaseous exchange (e.g., carbon, methane) can be obtained (Richey et al. 1997; Potter et al. 2014).

SAR data have also been integrated for mapping the extent of *herbaceous vegetation types*, including within the varzea regions of the Amazon where an object-based approach to classification has been recommended (Silva et al. 2008). By using multifrequency, polarimetric data, a greater level of discrimination can be achieved. For example, near Ankor Wat in Cambodia, NASA JPL fully polarimetric airborne SAR (AIRSAR) data have been used to differentiate macrophytes, flooded grasslands, and both trees and shrubs (Milne and Tapley 2004).

In many tropical regions, *peat swamp forests* are common and are of particular importance because of their unique biodiversity and the large amounts of carbon they contain (Jauhiainen et al. 2005). As cloud cover is persistent in these regions, data from moderate resolution optical sensors are often limited for characterizing these forests and detecting changes in their extent, although dense time-series can be used to map their extent and detect disturbance (e.g., logging). However, these peat swamps are particularly distinct within L-band SAR data and some significant changes in their state have been observed by comparing time-series of these data (Hoekman 2007).

Throughout the tropics and subtropics, *mangroves and sea grasses* occur in the coastal zones and are considered as wetland habitats. For mapping mangroves, moderate resolution optical data has been advocated with several studies (e.g., Spalding et al. 2010; Giri et al. 2011) generating regional to global maps. As cloud is often persistent in mangrove regions, the use of these data for detecting change has been limited. However, time-series of JERS-1 SAR and ALOS PALSAR data can be used for monitoring losses and gains in mangrove extent and associating this with a particular cause (e.g., agriculture, aquaculture, provision of sediments; Thomas et al. 2015; Lucas et al. 2014b). Given the homogeneity of their canopies and their adjacency to the sea, the height of mangroves has been estimated using SRTM data together with ICESAT GLAS waveform data (Simard et al. 2008). For detailed mapping of mangroves (e.g., for the establishment of baseline maps of extent, height and species composition), the use of stereo aerial photography as well as airborne or spaceborne visible and near infrared data has been advocated (Lucas et al. 2002).

Remote sensing allows large areas of *sea grass* habitat to be monitored. A common approach has been to use aerial photography but the use of hyperspectral data has proved to be more effective in terms of mapping different species and determining cover and other biophysical attributes (e.g., Leaf Area Index) (Phinn et al. 2008). Time-series of moderate spatial resolution data (e.g., Landsat and SPOT) have also allowed trends in extent and condition to be described. A recent development has been the use of acoustic remote sensing for mapping seagrass extent as well as height and structure (Montefalcone et al. 2013).

For characterizing lakes in the Pantanal region of subtropical South America, a combination of spaceborne C-band and L-band data has been used in an object-based approach (Costa and Telmer 2006). Using these data, differences in backscatter between brackish and freshwater lakes have been observed. These data were also used to indicate differences in lake water geochemistry as inferred from vegetation types and amounts.

Remote Sensing of Anthropogenic Activities

Whilst many wetlands are still observed in their natural state, humans have exerted a considerable influence on their extent, condition, and dynamics. Such impacts, many of which can often be detected from remote sensing, arise from agricultural production, aquaculture, dam and reservoir construction, drainage, harvesting of aquatic resources (e.g., wood and charcoal), hydrological and sediment diversions, and salt production. However, humans also contribute to the creation and restoration of wetlands.

In many regions, aquaculture is prevalent with mangroves being the most common ecosystem within which ponds are placed. These ponds are particularly evident within SAR data with time-series of JERS-1 SAR and ALOS PALSAR data being particularly useful for discriminating aquaculture ponds that are pre-1990s and post-1990s in their formation (Lucas et al. 2014b). Rice paddies are also widespread and SAR data have often been used again for mapping extent, inundation, and cropping intensity (Rosenqvist 1999). In many regions, the SAR signal follows a distinct trend over the cropping cycle that can be used to indicate growing season length and the frequency of cycles within any one year. Coarse spatial resolution optical sensors (e.g., MODIS) have also been used for rice mapping using wavelengths and indices that are sensitive to water and vegetation (Xiao et al. 2005).

Regional to Global Wetland Mapping Programs with Remote Sensing Contributions

To ensure conservation, wise and sustainable use of mangroves, a number of regional to global initiatives have been established, with these including the European Space Agency's (ESA) Globwetland (I and II), JAXA's Kyoto and Carbon (K&C) Initiative, and the NASA's Making Earth System Data Records for Use in Research Environments (MEaSUREs) projects.

The *Globwetland projects* were designed to demonstrate the use of EO data for wetland mapping and monitoring over large areas. Globwetland I (Jones et al. 2009) was aimed at developing and demonstrating products and services based on remote sensing data for wetland managers and national authorities with Globwetland II

(Paganini et al. 2010) focusing on the provision of decision support tools for wetland management and conservation activities. In this second stage, the aim also was to generate maps of land use, land cover, change and water cycle regimes for 200 wetlands. The third Globwetland project, GlobWetland Africa, seeks to address wetland conservation and management within Africa and using specifically data from the Sentinel constellations of the Copernicus initiative.

The JAXA K&C (Rosenqvist et al. 2007) aimed to demonstrate the benefit of using data from the JERS-1 SAR and ALOS PALSAR data for conservation, international conventions, and carbon cycle science. This initiative has resulted in regional products, including maps of inundation in the Sudd Wetlands, Pantanal, and Amazon; maps of rice paddy cycles in southeast Asia and the United States; and the classifications of habitats of international importance including peat swamp forests in Indonesia. The science undertaken as part of this initiative forms the basis of applications using the ALOS-2 PALSAR-2 launched in 2014.

The NASA MEaSUREs projects aims to advance the use of earth observation data and pioneer scientific use of satellite measurements to better understand the earth system. One project is the Inundated Wetlands Earth System Data Record (IW-ESDR), which seeks to monitor wetland extent and dynamics and understand their role in, for example, greenhouse gas and water cycling, climate impacts and feedback, ecosystem health and management of water resources. For this purpose, focus is on the use of SAR data.

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Electromagnetic Spectrum: Regions Relevant to Wetlands

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Richard Lucas

Contents

Abstract

Given the diversity of wetlands, active and passive remote sensing data acquired in different regions of the electromagnetic spectrum are needed. Optical remote sensing data are particularly useful for retrieving information on the characteristics of the water column (e.g., total suspended matter, chlorophyll-a) but also the subsurface (e.g., substrate) and associated vegetation types. A number of indices and models have been established specifically for retrieval from optical data, with these relating to, for example, water state and vegetation productivity. Thermal infrared data also provide information on the temperature of the land and water surfaces. Data acquired in the microwave regions provide information on the three-dimensional structure of surfaces but also moisture contents and relative roughness. While each region provides useful information on particular aspects of wetlands (e.g., productivity, water pollutants), the integration of data from sensors operating in different modes and regions is advocated.

Keywords

Remote sensing · Electromagnetic spectrum · Thermal · Optical · Radar · Data integration

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Introduction

The electromagnetic spectrum describes the frequencies of electromagnetic radiation with the spectral, thermal, and microwave regions being of key importance for characterizing, mapping, and monitoring the components of wetlands (Fig. 1). A diverse range of ground, airborne, and spaceborne active and passive sensors are available to provide data in these regions, with these operating across a broad range of spatial resolutions and coverages and temporal frequencies.

The Spectral Regions

The spectral region covers the visible (0.4–0.7 μm) to the shortwave infrared (2.5 μm) and observations are provided primarily by passive optical (e.g., multi-spectral sensors, including aerial photography). These sensors record the solar energy reflected from the earth's surface and the atmosphere, and data are often calibrated to radiance (W m^{-2} sr^{-1} μm^{-1}). Conversion to surface reflectance (%) is achieved using empirical methods or ray tracing models that correct for the effect of the atmosphere such as the Second Simulation of a Satellite Signal in the Solar Spectrum or 6S) (Vermote et al. 1997). Differences in illumination as a function of viewing and solar angles (i.e., the bidirectional reflectance distribution function or BRDF) and topographic slope and aspect can also be accounted for (Shepherd and Dymond 2003). Cloud screening is a frequent requirement.

In the spectral region, the reflectance properties of inland and transitional waters depend upon the inherent optical properties (IOPs) of the water medium itself and the materials contained, with these including the optically active constituents of phytoplankton (commonly associated with a high concentration of green photosynthetic pigments), colored dissolved organic matter (CDOM), and triptons, with this being suspended nonliving debris (mineral matter, humus, or organic remains). Apparent optical properties (AOPs) are also distinguished with these being a function of the medium (the IOPs) and the directional structure of the ambient light field. These properties include irradiance reflectance and the (vertical) diffuse reflectance. Reflectance also depends upon the roughness of the water surface itself and temperature variations.

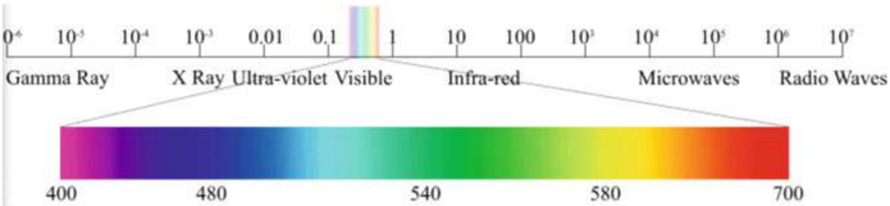

Fig. 1 The electromagnetic spectrum

Fig. 2 Waters of the Rio Solimoes and Rio Negro, Manaus, Brazil. The Rio Solimoes (*lowest* in image) is tan colored because of the large amount of sediment eroded from the Andes Mountain. The waters of the Rio Negro are black because of the low levels of erosion; instead, organic matter from the forest vegetation leads to the black color of the river (Source: NASA Earth Observatory)

The transmission of energy through water is typically greater towards the blue end of the electromagnetic spectrum, allowing detection of subsurface features (e.g., bedloads, sediment types, or vegetation such as sea grasses and algae). Depending upon the type and loadings of sediments, the reflectance is also generally greater in the visible region compared to the longer near and shortwave infrared wavelength regions where it often is $<1\%$ or approaching zero. An example of differences in the visible reflectance of water is shown in Fig. 2 where the mixing waters of the Rio Solimoes and Rio Negro in Brazil highlight the ability to differentiate between sediment and tannin-laden waters.

To retrieve the concentration of optically active water components from spectral data, empirical relationships or inversions of calibrated bio-optical models are often used (Hoogenboom et al. (1998). Using empirical methods, parameters such as chlorophyll a, cyanobacterial pigment phycocyanin (PC), total suspended matter (TSM), absorption by colored dissolved organic matter (a_{CDOM}), Secchi disk depth (Z_{SD}; or water clarity), turbidity, and water temperature have been retrieved (Matthews 2009). For the empirically based algorithms, single bands, ratios of bands or multiple bands, or derived measures are used in linear, multiple linear, or nonregression analyses. These empirical approaches are more widely adopted as they have been shown to provide reliable estimates, have been used to derive standard products from certain satellite sensors, can, in some cases, be used independently of in situ data, and have a biophysical explanation. The bio-optical approach is often combined with an approximation of the radiative transfer model to model the reflectance of water as a function of the water IOPs or a direct solution to the equation is used. The inversion of these forward models, through, for example, mathematical optimization or multiple nonlinear regression procedures, allows a number of IOPs to be retrieved from the remote sensing data themselves. A critical component of these models is compensating for atmospheric effects.

A further property of water is the Total Suspended Matter (TSM), which describes (on the basis of mass or optical properties) the amount of organic and inorganic material. TSM is important in wetlands as it regulates light attenuation and impacts on the distribution and type of submerged aquatic vegetation. TSM is estimated based on the optical properties of the suspended particles and often using a constant relationship between particle mass and the particulate scattering or absorption, although this varies with particle size and composition. Methods for estimation include single band algorithms (e.g., based on empirical relationships with visible channels), band ratios (with less sensitivity to natural variability), and multispectral algorithms. Commonly used algorithms include the ratio of the form (R_{560}-R_{520})/(R_{560}+R_{520}) for relatively low levels of TSM (<66 g m^{-3}) or the ratio of the near infrared (R_{850}) to the visible green (R_{550}) where higher concentrations exist (Matthews 2009). Band ratios are preferentially used over single bands as variable particle grain sizes and sediment refractive indices are normalized. Turbidity may be a more relevant attribute for certain water quality applications where optical properties (e.g., transparency) are important.

Within many inland waters, the spectral reflectance of water is influenced by aquatic algae and the pigments they contain, and particularly chlorophyll a. In studies using optical or hyperspectral data, the concentrations of chlorophyll a within estuarine and inland waters have been best predicted using ratios of the red and near infrared reflectance (approximately R_{665}-R_{674} and R_{700}-R_{713} nm, respectively) (Lunetta et al. 2009; Matthews 2009). TSM and Chromophoric (colored) Dissolved Organic Matter (CDOM) were also predicted by Lunetta et al. (2009) from these data using a single parameter approach (Fig. 3), although improvements could be made through inversion of bio-optical models. The MEdium Resolution Imaging Spectrometer (MERIS) standard chlorophyll a retrieval algorithm is also based upon the R_{665}/R_{709} ratio with a correction for water absorption based on the R_{776} channel. The strong correlations observed are attributable to the increases in backscattering from phytoplankton and absorption by water towards the near infrared. Other estimates have been based on the fluorescence maximum near 685 nm, regressions based on single bands, or ratios of moderate resolution optical sensors similar to those outlined above (although retrieval is compromised by the relatively broad band widths and large signal to noise), or more advanced neural networks, genetic algorithms, or multivariate regression (Matthews 2009).

CDOM (gelbstoff or yellow substances) is comprised of humic and fulvic acids that color the water brown because of strong absorption in the blue region. The slope of the absorption curve from 440 nm to $>\sim$550 nm is quite predictable for most inland and coastal waters and as it is similar to that of other detrital material, it can make a contribution to CDOM absorption. Where CDOM is large (e.g., in humic lakes), retrieval of other biophysical attributes will be affected (Matthews 2009). A problem with retrieval is that many sensors do not have a blue observation channel,

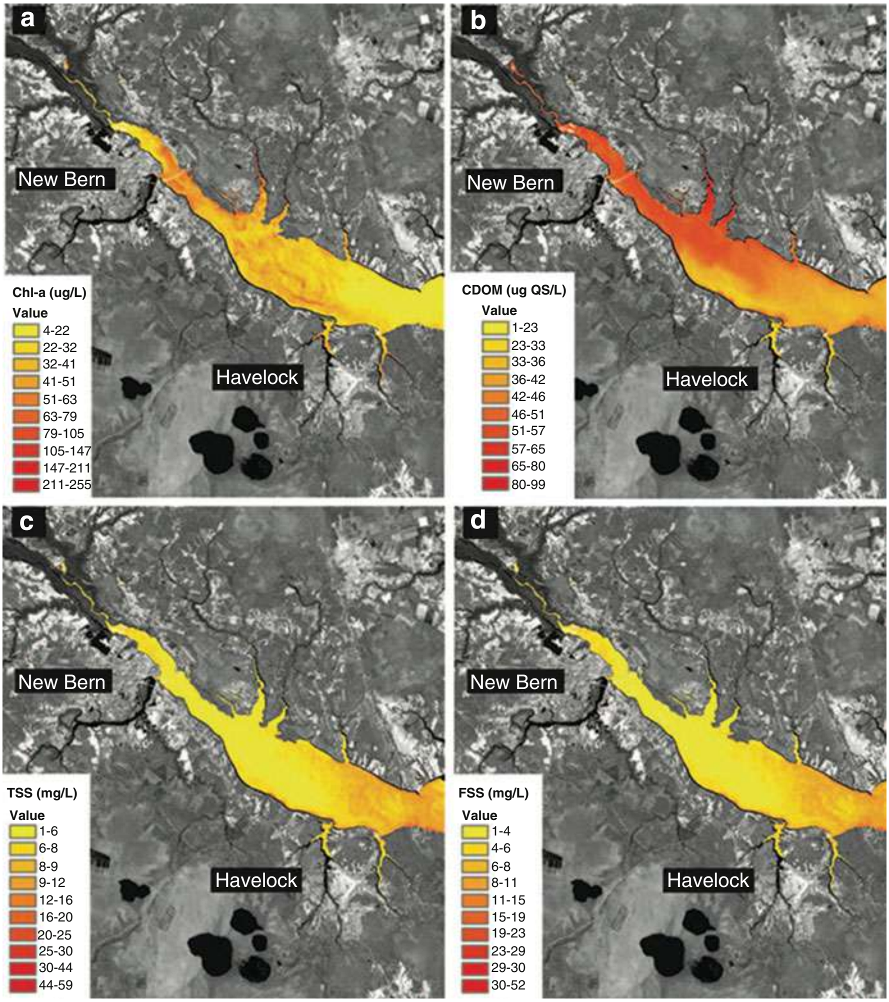

Fig. 3 Concentrations of (**a**) chlorophyll a, (**b**) CDOM, (**c**) TSM, and (**d**) fixed suspended solids (FSS) predicted from AVIRIS data (Lunetta et al. 2009)

or this band is affected by atmospheric conditions. Retrieval algorithms often use ratios of the blue reflectance with that of the green or red and perform best when the amount of suspended sediment is low. Ratios of the MERIS bands (R_{665}/R_{490} and R_{665}/R_{550}) are also used with the 665 nm band associated with maximum chlorophyll a absorption, which normalizes the effects of this and also backscattering by particulate matter.

Water quality constituents (WQSC) and optically active substances (OAS) can also be retrieved using remote sensing data. As an example, the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO) has developed an algorithm based on a physics-based adaptive linear matrix inversion (aLMI) that simultaneously derives concentrations of water quality variables over different water types. Satellite sensors with capability for observation of water quality include the MERIS and MODIS-AQUA sensors.

A number of indices have been designed for detecting vegetation liquid water from optical remote sensing data, including the Normalized Difference Water Index (NDWI), which uses two near infrared channels centered at approximately 0.86 μm (where absorption by water is minimal) and 1.24 μm (Gao 1996). Other measures include canopy water content (g m^{-2}), which is the product of the mean leaf equivalent water thickness (EWT, g cm^{-2}; the amount of leaf water divided by its area) and leaf area index (m^2 m^{-2}).

Infrared Regions

While many studies have focused on the spectral (and primarily visible) components of the electromagnetic spectrum, the thermal infrared proportion (from 3 to 35 μm) provides useful information on water state and condition. Sensors operating thermal bands include those on board the Landsat and MODIS satellites. For open water in more expansive areas, thermal infrared sensors have been used to map circulation and temperature patterns, typically in lakes (e.g., Anderson et al. 1995). Torgersen et al. (2001) also used airborne data to measure the thermal properties of rivers and streams.

Microwave Regions

The microwave region includes both a passive and active component. Active microwaves are emitted from human-made sensors while passive microwaves (0.15–30 cm) emit naturally from the Earth's surface. The passive microwave brightness temperature is related to kinetic temperature through the emissivity of the material and varies considerably as a function of surface materials, with sensitivity to moisture of the material. Passive microwaves are also attenuated by atmospheric components. The active microwave region extends from 1 mm to 1 m, and energy is typically transmitted artificially from radar antennas. Most sensors operate as Synthetic Aperture Radars (SAR) at X-, C-, L-, and P-band. SAR remote sensing instruments overcome the limitations of optical imagery in that they are unaffected by atmospheric water vapor or cloud cover (with the exception of very short wavelengths) and provide information on land cover and inundation mapping (Henderson and Lewis 2008). Using SAR data, open water as well as snow and ice are readily detected. Where expanses of smooth open water occur, microwaves at all frequencies are reflected away from the sensor resulting in low returns. However, similarly low returns are often observed (depending on polarization) for bare soil, short vegetation, grasslands, and shrublands, particularly where surfaces are smooth and the size, dimensions, and density of plants elements is low;

hence, confusion with open water is commonplace. During windy conditions or when water is turbulent (e.g., as a consequence of tidal or freshwater currents), the increased roughness of the surface enhances backscatter. Hence, open water may become difficult to distinguish from surfaces such as woody vegetation or rough ground. Lower frequency radar are more useful for detecting water bodies because of reduced sensitivity to the surface roughness of the water and ability to penetrate through sparse and/or nonwoody (e.g., herbaceous) vegetation.

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Remote Sensing Instruments: Sensor Types Relevant to Wetlands

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Richard Lucas and Maycira Costa

Contents

Abstract

A wide range of sensors are available for providing information on wetlands, both in the past and in near real time. The Landsat series of sensors has been providing data since the 1970s, and these provide a unique record of the changing extents and states of water across the globe. Using these data, a number of regional or global maps of annual water inundation have been generated. Coarse resolution optical sensors such as MODIS provide more regular observations of water and have been used to better understand water flows and snow accumulation and melt across landscapes. Microwave sensors operating at X, C, and L band have also provided high to moderate resolution cloud free observations of wetlands on a regular basis since the early 1990s and are particularly useful to quantifying changes in wetland extent and dynamics, including in areas where forest cover is dense. Airborne and spaceborne LIDAR have mainly been used to retrieve information on the three-dimensional structure of vegetation. Stereo-optical data,

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lidar, and radar can also indicate the topography of the underlying surface, which is useful in hydrological modeling. A number of missions are anticipated to increase the amount of information on wetlands, with these including the Copernicus Sentinel missions.

Keywords
Optical · Lidar · Radar · Copernicus · Future missions

Sensors in the Spectral Region

A wide range of sensors operate in the spectral region of the electromagnetic spectrum where they provide information that allows characterization, mapping, and monitoring of wetlands. Key among these are the Landsat sensors and Moderate Resolution Imaging Spectroradiometer (MODIS). The Landsat series of sensors have been the Multispectral Scanner System (MSS), Thematic Mapper (TM), Enhanced Thematic Mapper (ETM+), and the Operational Land Imager (OLI). The Landsat MSS was launched on July 23, 1972 and provided the first moderate resolution global observations of the world's surface in the visible and near infrared regions. Three more MSS sensors (Landsat 2-4) followed. While the radiometric range was low, these data provide the oldest and most detailed record of the extent of wetlands. First launched on 16 July 1982, the TM operated in six spectral bands and one thermal. The Landsat ETM was launched in 1999 and again observed in six spectral bands (visible to shortwave infrared) with an additional panchromatic band as well as a thermal band. Following the failure of the Landsat-6 at launch on October 5, 1993, users were dependent upon the Landsat-5 and 7 and observations were compromised by a partial failure of the latter. However, as part of the Landsat Continuity Mission (LCM), the Landsat 8 OLI was launched on 11 February 2013 and operated in the same wavelength regions as the ETM+. In addition to the bands used in the previous TM and ETM+ missions, the coastal blue channel was included as well as a cirrus channel (1.36–1.38 μm), with the latter facilitating better atmospheric correction because of sensitivity to atmospheric constituents. A particular advantage of the Landsat series is that a long historical record of the extent and characteristics of wetlands globally is provided, although observations have been compromised by the relatively low temporal frequency (every 16 days with this increasing when more than two sensors are operating) and cloud and cloud shadow in many regions.

The MODIS is on board the Terra (EOS AM) and Aqua (EOS PM) satellites, which collectively view the earth's surface every 1–2 days in 36 spectral bands (2 at 250 m resolution at nadir, 5 at 500 m, and 29 at 1000 m spatial resolution) or groups of bands. Observations are in the visible, near infrared, short/mid-infrared and thermal bands. For wetland mapping, this sensor provides opportunity for capturing short- to long-term flooding events across large areas. A number of products are commonly used including the Nadir BRDF-Adjusted Reflectance (NBAR) and the 8-day land surface temperature product (MYD11A2) (Ordoyne and Friedl 2008). Other sensors providing useful information for wetland mapping include the moderate spatial resolution SPOT, High Resolution Geometric (HRG), the ALOS AVNIR-2, and the Terra-1 ASTER. At finer spatial resolution, Very High Resolution (VHR) spaceborne multispectral data allow considerable detail to be resolved in wetland environments. Among these are the Quickbird and IKONOS sensors, although the Worldview-2 is particularly useful because of the presence of coastal, yellow and red edge bands, which allow better detection and description of water and vegetation. Stereo viewing capability also allows the extraction of digital elevation models (DEMs) that are useful for hydrological analysis.

Microwave Sensors

For mapping the extent of wetlands, Synthetic Aperture Radar (SAR) technology has proven to be effective as persistent cloud, smoke, and haze prevents observations using optical sensors. Furthermore, microwaves are able to penetrate vegetation and interact with the underlying surface, thereby allowing detection of flooding. SAR data have therefore been used for mapping inundation and land covers, and retrieving biophysical properties. Key spaceborne SAR operating at C-band (5.3 GHz) have included the European Space Agency's (ESA) European Remote-Sensing (ERS) satellite-1 (1991–2000) and -2 (1995–2011) and the Canadian Space Agency's (CSA) RADARSAT-1 (1995–2013) and -2 (2007–present). The ENVISAT Advanced Synthetic Aperture Radar (ASAR; 2002–2012) was also launched in 2002, with this operating in imaging mode (IM; 30 m) and two ScanSAR modes, the Wide Swath Mode (100 m) and the Global Monitoring Mode (1000 m). These latter sensors allowed observations up to 10 times per month when descending and ascending acquisitions were combined and were particularly beneficial for mapping and monitoring the extent of inundation. A limitation was that the large scan width lead to variations in backscatter because of large incidence angle differences across the swath and also because of varying surface moisture conditions between observation dates. The resolution was also insufficient to detect small features (e.g., ponds, rivers). The Tandem-X mission, operating at X-band (9.6 GHz), was also instigated by the German Space Agency (DLR) and integrated the TerraSAR-X mission (launched in 2007). By using these paired satellites, digital elevation models (DEMs) of the global land surface are being generated with these complementing that provided by the Shuttle Radar Topographic Mission (SRTM) in 2000. The DEMs provided by both sensors are particularly useful for indicating the distribution of wetlands as well as river networks.

A number of L-band (1–2 GHz) SAR have been launched, primarily by the Japan Aerospace Exploration Agency (JAXA). The first of these was the Japanese Earth Resources Satellite (JERS-1) SAR, which was in operation between 1992 and 1998, and the Advanced Land Observing Satellite (ALOS) Phased Arrayed L-band SAR (PALSAR; 2006–2010). JAXA also implemented a systematic observation strategy that ensured complete continental coverage of these data at regular intervals and at times when data collection was most beneficial (e.g., for flood inundation mapping).

The ALOS-2 PALSAR-2, their successor, was launched in 2014 and is providing a combination of spotlight, fine beam mode, and wide swath (ScanSAR) mode data. Airborne systems operating at X, C, L, and/or P-band include the AIRSAR, GEOSAR, UAVSAR, and EUSAR.

LiDAR

Airborne Light Detection and Ranging (LiDAR) is being increasingly acquired over large landscapes, thereby allowing generation of digital elevation models (DEMs) of the terrain (DTMs) and surface (e.g., of canopies; DSMs). LiDAR operates in both discrete and full waveform mode, with the former recording several reflections for each transmitted pulse while the latter measures the entire return waveform at very high sampling frequencies. In both cases, significant information on the underlying surface, the canopy, and the vertical distribution of vegetation is obtained.

Future Missions

A number of future missions are planned which will benefit the observation of wetlands. They among these are Sentinel 1 and Sentinel 2 constellations of the Copernicus initiative which will provide long-term access to enhanced radar observations and high-resolution super-spectral data, respectively, opening a new scenario for mapping, assessment, and monitoring of wetlands worldwide. The C-band imaging radar of the Sentinel 1 mission will provide all-weather day-and-night imagery which will be extremely useful for monitoring wetlands in cloudy conditions. The Sentinel 2 mission will provide systematic optical observations of all terrestrial and coastal zones, at 10 m spatial resolution, with a swath width of 290 km. Together with its twin satellite it will cover the Earth's surface with a repeat cycle of 5 days at the equator. The impressive footprint of Sentinel 2 along with the short revisit time and its systematic acquisition policy will allow rapid changes in ecosystems to be precisely monitored and is ideally suited to monitor sensitive habitats such as wetlands. It will allow for seasonal and permanent changes in wetlands to be monitored with higher accuracy. The GMES sentinel data policy, with its full and open access for all users, is an important incentive that would largely facilitate the uptake of these new technologies by the wetland communities.

While each of these sensors can be used alone, combinations of these data provide greater capacity for characterizing, mapping, and monitoring wetlands (e.g., Bwangoy et al. 2009; Michishita et al. 2012). The number and diversity of sensors is also increasing as more countries obtain capability for their launch and operation. A comprehensive overview of trends and the status of sensors is provided by Belward and Skøien (2014).

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Remote Sensing of Water in Wetlands: Inundation Patterns and Extent

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Bruce Chapman, Laura Hess, and Richard Lucas

Contents

Abstract

Seasonally varying inundation extent and duration are key properties of wetlands, but are poorly quantified, particularly in tropical, boreal, and coastal regions. Optical sensors such as Landsat are limited by cloud cover, although sensors such as MODIS, with high repeat frequency, partly compensate for this limitation. Synthetic aperture radar (SAR) sensors are insensitive to cloud cover, and at longer wavelengths (C-band and L-band) are capable of detecting water beneath vegetation canopies. Time series of SAR data are effective for monitoring seasonal inundation dynamics, and combinations of different SAR wavelengths

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and polarizations can discriminate vegetation structure. Optical, SAR, and passive microwave sensors are being employed at global scale to characterize the role of wetlands in global hydrologic and biogeochemical cycles.

Keywords
SAR · AIRSAR · ALOS · PALSAR · UAVSAR · Napo river · Pacaya-samiria · Pantanal · Double-bounce

Introduction

Inundation, whether permanent or temporary, influences biogeochemical cycles, trace gas fluxes between land and atmosphere, nutrient uptake, and sediment amounts and redistribution, and determines the types, extent, and condition of aquatic vegetation (Alsdorf et al. 2007; Frappart et al. 2005). Inundated areas may be connected to or isolated from the main river channels (Ward et al. 2013; e.g., those that are rainfed or remain following water recession). The levels, depths, and movements of water depend largely on the nature of the elevation, slope, and morphology of the terrain (Alsdorf et al. 2000).

Despite their importance, timely and accurate information on inundation is not available for many regions. As illustration, across much of Africa, the hydrologic dynamics of seasonally flooded wetlands and floodplains remains poorly quantified (Lee et al. 2011) despite these occupying over 1 million km^2 and contributing to the livelihoods of millions of people. Equally, in northern latitudes, the extent and dynamics of wetlands are not well known (Whitcomb et al. 2009) and large gaps exist in relation to wetland surface elevation changes in tidally inundated coastal wetlands (e.g., saltmarshes and mangroves; Webb et al. 2013). Monitoring of inundation may also be compromised by urbanization of proximal environments, as in the case of temperate peatlands (Andersen et al. 2011).

Methods for Detecting and Mapping Inundation

For detecting, mapping, and monitoring inundation, a wide range of methods have been developed using a diverse range of sensors. Landsat sensors at moderate (<30 m) spatial resolution have frequently been used for large scale mapping of the extent of open water but the dynamics are often not captured because of the 16-day repeat frequency, which is further reduced when cloud cover occurs; water beneath vegetation (particularly dense vegetation) is also difficult to detect (Smith 1997; Alsdorf et al. 2000; Vanderbilt et al. 2002). Greater success in mapping inundation extending over large areas has been achieved using lower resolution optical sensors, such as MODIS, which provide imagery on at least a daily basis and at a global level although the time-series may still be compromised by image quality (hence composite images are often used). The relatively coarse (>250 m–1 km) spatial resolution limits detail in the delineation of open water, although several studies have used a suite of complementary satellite measurements to estimate the fractional extent of inundation (Prigent et al. 2007). Inundation extent is also often underestimated with detection depending upon the reflectance of the background and also the depth of water. Methods for mapping inundation from optical data have typically been based on density slicing (e.g., Shaikh et al. 2001) or classification (e.g., Rebelo et al. 2009) of single or combinations of bands and/or derived indices (e.g., the modified Normalized Difference Water Index or Open Water Likelihood Algorithm) (Chen et al. 2013).

Synthetic Aperture Radar (SAR) data have been shown to be useful for mapping and monitoring the extent of both open water and subcanopy inundation at relatively high resolution (100m or better) (e.g. Hess et al. 1990; Hess et al. 1995; Smith 1997; Kasischke and Bourgeau-Chavez 1997; Kasischke et al. 2003; Frappart et al. 2005; Arnesen et al. 2013). Due to its insensitivity to cloud cover, complete SAR coverage can be obtained over continental scales within relatively short time intervals (12–46 day intervals), making it possible to monitor regional scale inundation dynamics (Rosenqvist et al. 2000; De Grandi et al. 2011). The ability of SAR to detect open water is attributable to specular reflectance of the radar waves by the smooth reflecting water surface, resulting in a dark radar image pixel (Telmer and Costa 2007). There can be ambiguity, however, as water can become brighter if it is wind roughened, and ground barren of vegetation can also be quite dark. By carefully selecting the incidence angle range that is used in the analysis, the effect of wind roughening can be reduced; the ability of the SAR to distinguish between open water and bare ground is partially dependent on the Signal to Noise Ratio (SNR) and noise level of the SAR, its wavelength, and its polarization.

Vegetated environments with subcanopy inundation may be identified if the wavelength of the SAR is greater than about 5 cm (C-band). The ability of SAR to detect inundation in the presence of vegetation is attributable to the transmitted microwaves scattering off both the flat open water surface and the vertically emerging vegetation (i.e. a "double bounce" reflection). This results in a large fraction of the transmitted power being reflected directly back to the radar. At wavelengths shorter than 5 cm, unless the vegetation is sparse or the canopy broken, specular or volume scattering from the surface or canopy layer will generally dominate (see Fig. 1 for an illustration of the scattering mechanisms). But as the wavelength increases from 5 cm to 20 cm (L-band) to a meter or more (i.e. P-band), penetration through the vegetation (including woody vegetation) to the underlying water layer is more common, with double bounce reflections becoming the dominant signal in these inundated environments. Despite volume scattering and attenuation of the radar signal by the vegetation canopy, a strong return back to the radar can be received. Figure 2 illustrates the changing dominance of double-bounce reflections with changing wavelength.

As exemplified by Hess et al. (2003), image brightness thresholds may be used to detect inundated vegetation, as the double bounce areas are significantly brighter than areas dominated by specular and volume scattering. However, similar brightness values can occur in urbanized areas (e.g., particularly if streets are orientated parallel to the flight track of the radar) or where the surface is rough (e.g., lava flows). Figure 3 shows an example SAR image from the JAXA ALOS PALSAR instrument

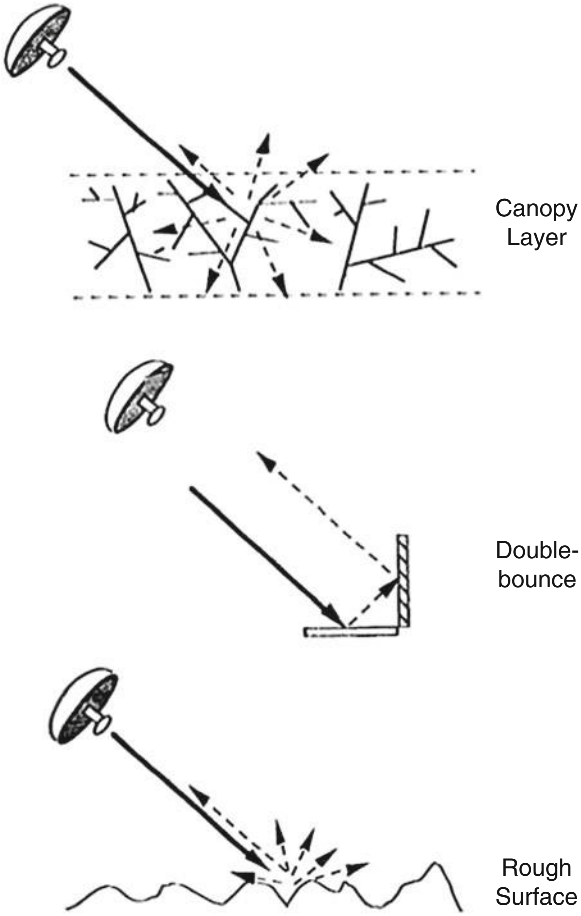

Fig. 1 An illustration of the dominant scattering mechanisms in SAR imagery. Double bounce reflections can occur when level water (which is a strong specular scatterer) surrounds emergent vegetation (Chapman and Blom 2013)

and an associated inundation classification. As can be seen, brighter image areas may be associated with inundated vegetation, and darker image areas of may be associated with open water. This image was acquired in a wide-swath imaging mode called "ScanSAR".

The configuration of the SAR includes the polarization as well as the wavelength. We have assumed a typical polarization configuration of horizontally polarized on transmit and receive (called "HH"), which is optimally sensitive to double bounce reflections. However, a SAR may be capable of acquiring other polarization combinations. Using polarimetric decomposition techniques, a fully polarimetric radar (collecting four orthogonal polarizations) is capable of clearly identifying the scattering mechanism (Cloude and Pottier 1996). Figure 4 shows an example of a multipolarization SAR image for a vegetated environment that is inundated, illustrating the complexity of the scattering behaviors in these types of regions.

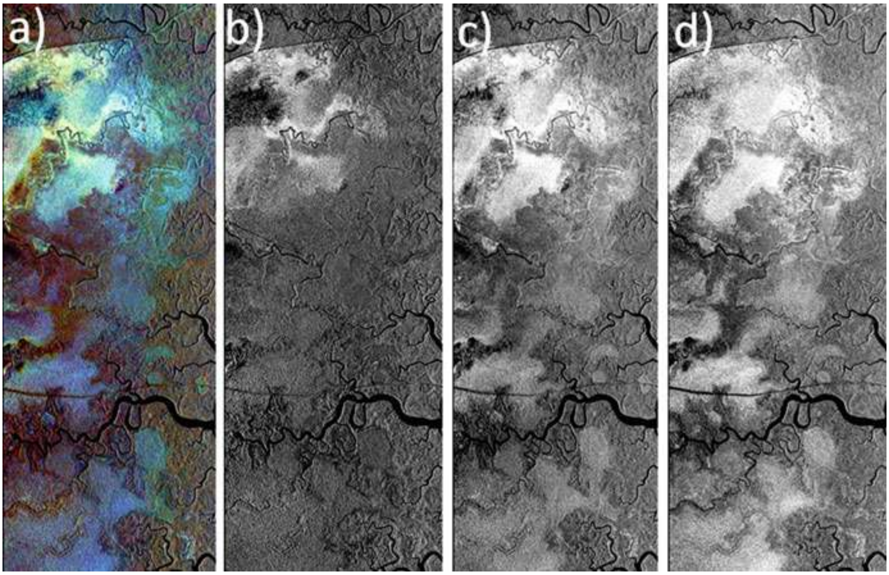

Fig. 2 (**a**) Color overlay of C-band HH (*white*), L-band HH (*cyan*), P-band HH (*blue*); (**b**) C-band HH image *greyscale*; (**c**) L-band HH image *greyscale*; (**d**) P-band (68 cm) HH image *greyscale*. Inundated mangrove canopy best revealed as areas of high backscatter in P-band and 3-band multiwavelength composite image (AIRSAR image courtesy of NASA/JPL)

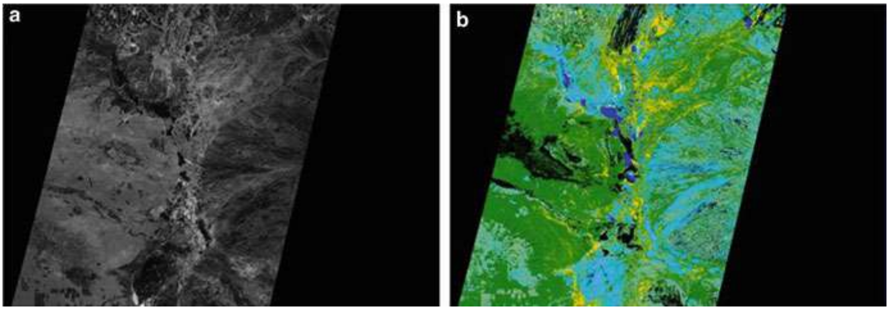

Fig. 3 Pantanal Wetlands, South America. (**a**) A portion of a single ALOS PALSAR SCANSAR terrain corrected image from mid-2008, approximately 400 km wide, *greyscale*; (**b**) corresponding maximum inundation classification, where *dark blue* and *light blue* areas are open water, *yellow areas* are inundated vegetation, *green areas* are not inundated vegetation, and *black areas* have high slopes and are not expected to be inundated (ALOS image copyright JAXA/METI)

There is a particular advantage of using time-series of SAR data, in that they provide the opportunity to understand and quantify the dynamics of inundation across large and often inaccessible areas, with such information providing input to, for example, habitat maps and models or estimates of methane emissions from wetlands.

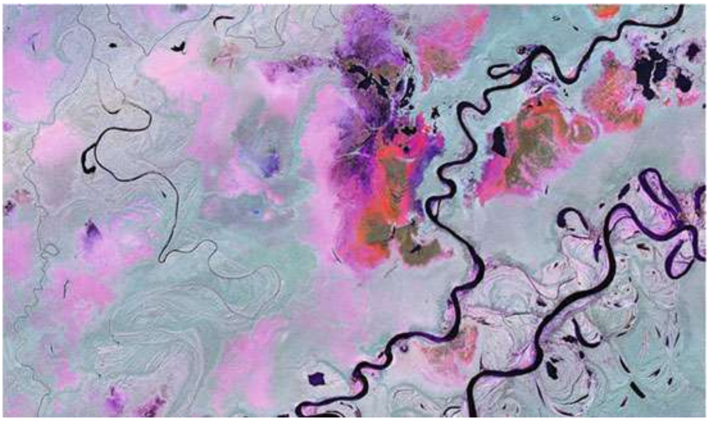

Fig. 4 Mosaic of 4 UAVSAR fully polarimetric L-band SAR images covering a portion of the Pacaya–Samiria wetlands in Peru, acquired March 2013. HH is *red*, HV is *green*, VV is *blue* in this color overlay of those three polarimetric channels. The colors indicate the variable brightness in each polarization channel due to differences in electromagnetic scattering by the inundated vegetation, open water, and not-inundated vegetation. UAVSAR is a fully polarimetric airborne SAR operated by NASA (UAVSAR image courtesy of NASA/JPL)

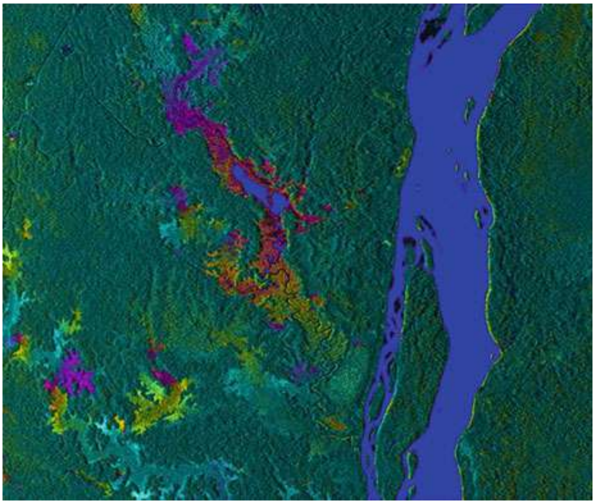

Fig. 5 Overlay of L-band HH interferometric phase (color) and image amplitude (brightness) over a portion of the Napo River in Ecuador, during a period of changing inundation level. Two data acquisitions, separated by two weeks and forming an interferometric pair, were obtained by NASA's UAVSAR, an airborne SAR operated by NASA, in March 2013. The colors are quantitatively related to the change in the water level of the subcanopy inundation (on the order of centimeters) between the two acquisition dates. *Solid blue* are areas of open water (Image courtesy of NASA/JPL)

Interferometric SAR (InSAR), in which two SAR images are acquired at different times and interfered, may be used to measure small changes (a fraction of a wavelength) in water level between the two observation dates in the presence of double-bounce scattering (e.g. inundated vegetation). This was first demonstrated by Alsdorf et al. (2000). Figure 5 illustrates this sensitivity of a SAR interferogram to slight changes in water level.

Alternative approaches used for detecting inundation have included the use of Light Detection and Ranging (LiDAR) intensity data following enhanced Lee filtering (Lang and McCarty 2009) for local area mapping. Gravity data from the NASA Grace mission were used in combination with satellite altimetry data and other sources to estimate inundation and also the volume of water filling and draining from large wetlands (Alsdorf et al. 2010; Lee et al. 2011). However, these data sets necessitate the masking of nonwetland areas and difficulty arises when detecting inundation under dense forest canopies in tropical regions (Melton et al. 2013). Papa et al. (2006) also highlighted the use of the dual Ku (13.6 GHz) and C (5.4 GHz) frequency radar altimeter on board the Topex-Poseidon satellite for detecting inundation over boreal regions, with monthly flooded areas determined from flooded pixel fractional coverage based on the altimeter C-band magnitude and vegetation effects considered by applying a linear mixing model to the C-Ku backscatter difference. Approaches to mapping inundation extent at a global level have included the use of passive microwave land-surface emissivities (e.g., from the Special Sensor Microwave/Imager or SSM/I) following removal of atmospheric, cloud, and rain contamination with active microwave (ERS-1) scatterometer data (at 5.25 GHz) and/or NOAA AVHRR Normalized Difference Vegetation Index (NDVI) data used in combination to assess vegetation contributions to the passive microwave signal (Prigent et al. 2007).

Future Challenges

The remote sensing techniques described here continue to be developed and expanded as new instruments are developed and launched into Earth orbit. As these products are produced, the uncertainties in estimates of inundation extent and dynamics are clarified. In turn, the uncertainties in biogeochemical models, estimates of trace gas fluxes between land and atmosphere, and knowledge of types, extent, and condition of aquatic vegetation and associated habitats may be reduced.

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Remote Sensing of Water in Wetlands: Persistence and Duration

222

Tony Milne

Contents

Abstract

The extent, duration and recession pattern of water over a landscape are major determinates of vegetation response and growth and species composition. Optical data and synthetic aperture radar (SAR) in particular, can be used to map vegetation response as well as separate perennial from non-perennial wetlands. Parameters including both the spectral and spatial resolution of remote sensing systems as well as the frequency of their data acquisition cycle all influence the capacity to detect and map phonological response.

Keywords

Classification · Flooding · River regimes · Synthetic aperture radar · Wetland vegetation

Introduction

The persistence of water refers to the duration of inundation and can be dependent on natural (e.g., rainfall, snowmelt) and/or human-induced activities (e.g., release from reservoirs, construction of levees, and damming). Persistence can be perennial or nonperennial. Establishing whether a wetland is persistent or otherwise can be

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problematic, particularly as a different surface is observed during the times when inundation does not occur. This surface is either nonvegetated (e.g., bare rock, soil, or sand) or vegetated, with herbaceous or woody plants dominating. For example, in the Manas National Park of India, extensive flooding occurs, which is followed by recession and exposure of the soil, subsequent regeneration of dense grasslands, and then fire before further flooding takes place. Hence, persistent areas of water in its various states are observed in each image within a time series, provided that time intervals between observations are sufficiently short.

In many natural systems, the duration of water, and also ice and snow, is often dependent upon seasonal patterns in climate whereas those of managed systems may conform to or may be independent of these seasons. The persistence and also duration of water also depends on tidal regimes, which leads to different levels of substrate exposure and water depth. This, in turn, influences vegetation distributions. Areas can be subtidal or regularly flooded (e.g., on a daily basis).

Tracking the Persistence of Water

Tracking the persistence of water in its various states from earth observation data generally requires that these aquatic surfaces can be detected but also that the timing of observation corresponds exactly or approximately with the timing of inundation or snow and ice cover. Few sensors can track the level of tidal inundation as most observe less regularly than and out of sequence with the tidal regimes. Nevertheless, in some ecosystems, the extent of inundation can be inferred from the increased backscatter at lower frequencies (e.g., L-band) as a result of the increased L-band double bounce reflections. Polarimetric SAR data acquired at mid to low tide can also be used to track differences in the amount of water retained at or below the surface in exposed areas. Tidal levels have also been detected through reference to TOPEX POSEIDON datasets (Egbert et al. 1994).

Moderate spatial resolution optical and radar sensors tend to observe less frequently (e.g., 16 days for the Landsat sensors and 44 days for the ALOS PALSAR) although if used in a time series, water that is persistent or otherwise can be detected from which the duration of flooding can be inferred. As an example, Rosenqvist et al. (2002) used all acquisitions of JERS-1 SAR to determine the extent and duration of inundation in the Jau River catchment in Amazonia, allowing the duration of water to be quantified. Radar data can also be used to identify and describe seasonal change and the impact of flood levels, as illustrated for the Mekong River which, together with runoff from local catchments, flows into Lake Tonle Sap during periods of flooding. Enhancements of radar data have the capacity to map and monitor the incidence of seasonal flooding in the Lower Mekong Basin as well as observe the larger regional responses to seasonal change. A 100 m resolution ScanSAR image acquired during the latter stage of the wet season on 5 November 2006 records the extent of flooding in Tonle Sap Lake and beyond into the wetland forest and adjacent rice fields (Fig. 1; Milne and Taply 2004). Areas of open water are black and clearly identifiable due to their specular surfaces. In

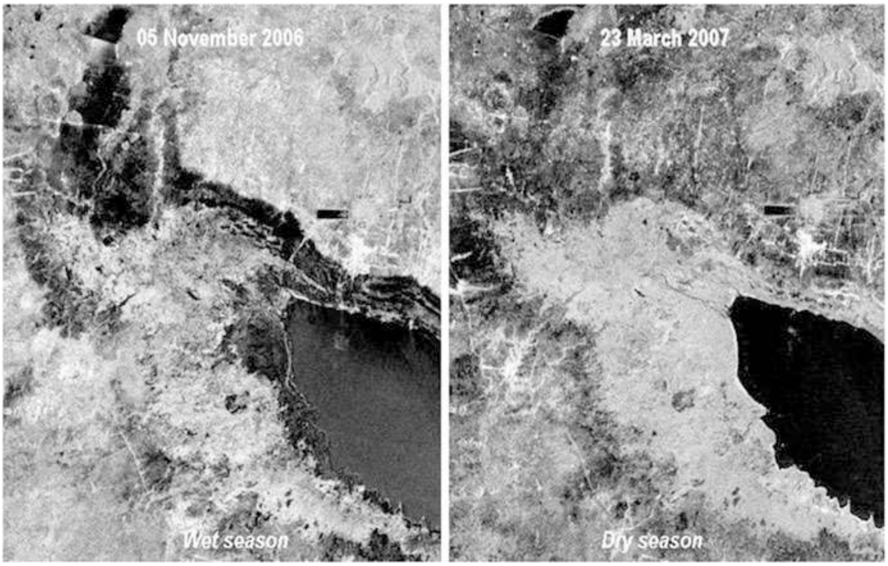

Fig. 1 North-western end of Tonle Sap Great Lake – ScanSAR images acquired during the wet (November) and dry (March) seasons show clearly the seasonal differences in the level of water in the lake, and also highlight flooding under tree canopies of the wetland forest, especially apparent in the wet season as bright (*white*) signatures (Milne and Taply 2004) © JAXA/METI

contrast, the dry season image for 23 March 2007 shows a shrunken lake area as the floodwater drains back into the main channel of the Mekong River. This annual reversal of flow dominates the ecological and human response to the prevailing environmental conditions. The expanded dark areas in the March image away from the lake and adjoining wetlands are a backscatter response (at L-band wavelengths) from dry rice paddies and bare fields. Images show in detail the impact of flooding and the effect of falling lake levels along the western end of the Tonle Sap Lake. Extensive areas of wetland forest are covered by water in the wet season but have "emerged" as the water level in the lake falls to its lowest in the dry season.

While moderate spatial resolution data typically provide detail in the extent of water, they lack temporal coverage. For this purpose, coarse spatial resolution sensors such as NOAA's Advanced Very High Resolution Radiometer (AVHRR) and the Terra and Aqua Moderate Resolution Imaging Spectrometer (MODIS) data are used as these observe the earth's surface on at least a daily basis and, provided cloud conditions allow, the extent of flooding over time can be mapped over time periods that range from daily to multiannual depending upon the lifetime of the observing sensors.

An alternative is to use digital terrain models (DTMs) of the ground surface generated using, for example, lidar or interferometric SAR, in conjunction with past records of observed inundation and/or river gauge data to infer the extent of inundation.

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Remote Sensing of Anthropogenic Activities: Agricultural Production

223

Nathan Torbick, Bill Salas, and Xiangming Xiao

Contents

Abstract

The development of aquaculture in many parts of the world has led to significant loss or conversion of wetlands. Both radar and optical remote sensing data have been used to map the extent and also development of aquaculture, which is particularly prevalent in tropical regions. Aquaculture systems are particularly distinct because of their distinct geometry. A number of studies have focused on assessing the impacts of aquaculture development on wetland ecosystems, including mangroves.

Keywords

Satellite remote sensing · Aquaculture · Ecosystems

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Introduction

Rice is an important anthropogenic wetland as paddies occupy nearly 11% of the Earth's arable land with more than 700 million tons produced annually. Rice agriculture has large demands for land and water resources with millions of hectares of irrigated lowland rice worldwide. Rice is grown in more than 100 countries around the world with the majority coming from South and Southeast Asia followed by Brazil, Japan, and the USA (Table 1). Rice provides livelihoods to hundreds of millions of people and is the fastest growing food staple in many developing nations.

Rice production necessitates intense water resource demands for multiple rice stages and managements such as seeding, straw decomposition and soil preparation, and waterfowl habitat. Rice paddy hydroperiod, or flood frequency and duration, is also arguably one of the most significant drivers of anthropogenic greenhouse gas (GHG) emissions (Li et al. 2005; Salas et al. 2007). Methane (CH4) from rice accounts for 20% of global sources with irrigated rice accounting for 80% of rice CH4 emissions (Wassmann et al. 2000). Alternative Rice Managements (ARM) - such as dry seeding and midseason drainage - are being evaluated as potential adaptation and mitigation strategies. At the same time, pressures on rice production ranging from drought to urban sprawl are threatening food security and thus the livelihoods of hundreds of millions of people. Resilient and sustainable agricultural development is critically needed to ensure more self-sufficiency in crop production and food security. To address these needs updated and accurate decision support information on rice extent, crop calendar, hydroperiod, and cropping intensity and production are required.

Table 1 Rice production by country

Countries	2011 (tons)	Countries	2011 (tons)
China	202,667,270	Republic of Korea	6,304,000
India	155,700,000	Pakistan	6,160,400
Indonesia	65,740,900	Egypt	5,675,030
Bangladesh	50,627,000	Madagascar	5,078,420
Viet Nam	42,331,600	Nigeria	4,567,320
Thailand	34,588,400	Nepal	4,460,280
Myanmar	32,800,000	Sri Lanka	3,874,800
Philippines	16,684,100	Iran (Islamic Republic of)	3,217,250
Brazil	13,477,000	Lao People's Democratic Republic	3,065,760
Cambodia	8,779,000	Malaysia	2,665,100
Japan	8,402,000	Peru	2,624,450
United States of America	8,391,870	Colombia	2,543,710
		Democratic People's Republic of Korea	2,479,000

Mapping Rice

Keys to mapping rice include accurate characterization of rice growth stages and managements. Matching sensors that are sensitive to these key stages and managements is a requirement for accurate and timely information. Several efforts over the past decade have utilized Synthetic Aperture Radar (SAR) observations for mapping rice areas (Figs. 1 and 2). Often highlighted is the all-weather capability and active collection system that operates independent of sun illumination. For rice

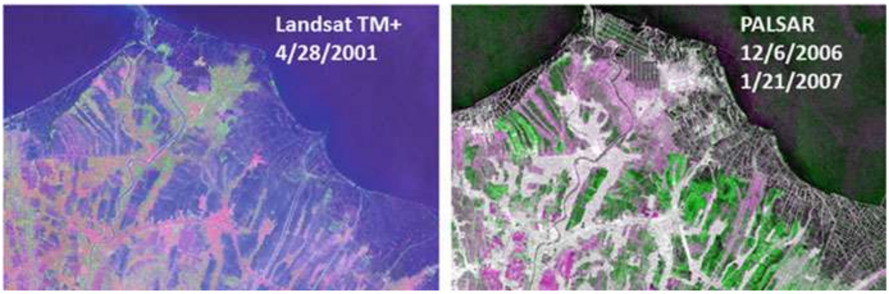

Fig. 1 Landscape in Kendal, Java, Indonesia. The *left* scene is the false-color composite of Landsat ETM+ image acquired on April 28, 2001, *Red*: Band 7, *Green*: Band 4 and *Blue*: Band 3. The *right* scene is two fine-resolution PALSAR images (HH) acquired on 6 December, 2006 and 21 January, 2007, respectively. Open water (*dark color*), fish ponds and rice paddies (*green* and *pink* color) could be visually detected easily from this two-date composite image. *Green* color indicates crop fields flooded in December 2006 but planted in January 2007; *Purple* color indicates crop fields planted in December 2006 but harvested in January 2007 (© JAXA/METI)

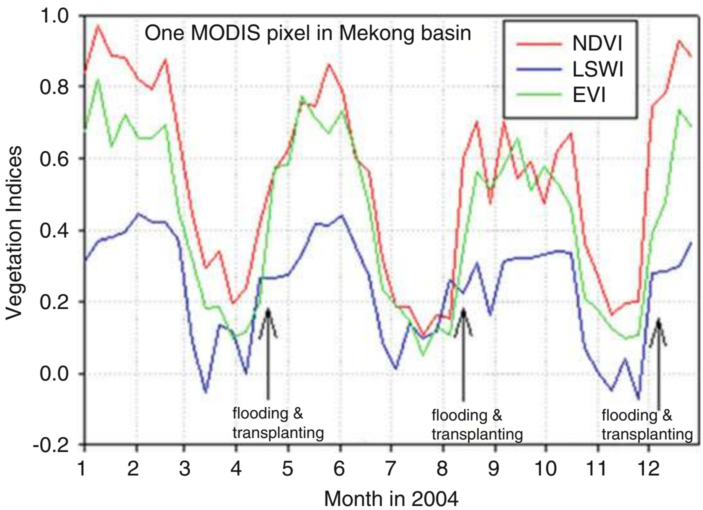

Fig. 2 MODIS dynamic indices can be used to identify key rice growth stages

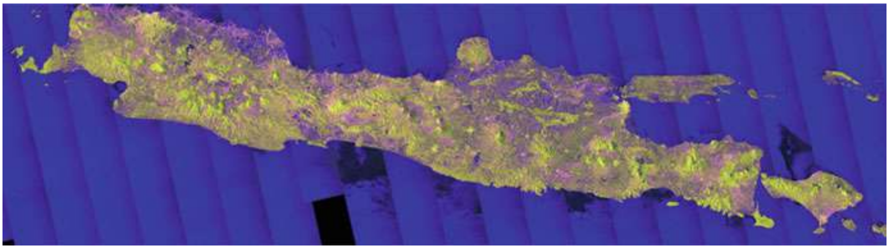

Fig. 3 2008 PALSAR FBD 50m Orthorectified Mosaics from dry and wet seasons are useful for mapping rice, especially in cloud prone regions such as Monsoon Asia (© JAXA METI)

applications, microwave observations at the relevant configuration can be sensitive to growth stages, soil moisture, and inundation frequency and duration (Chen and McNairn 2006; Inoue et al. 2002; Le Toan et al. 1997; Ribbes and Le Toan 1999; Torbick et al. 2011). Considering the dynamic range and scattering mechanisms of the rice life cycle, SAR can be particularly useful for mapping rice extent, inundation, and cropping intensity. During rice transplanting periods, the surface contribution of a paddy causes low backscatter. As rice plants accumulate biomass, the backscatter response increases until paddies are drained and crops reach harvest which tends to decrease backscatter. These patterns can be timed with knowledge of growing season length which is usually between 90–140 days depending on the type of rice and region. Overpass dates can thus be timed with these growth patterns and inundation periods to extract the unique SAR signature of rice. Output products from the SAR data have included fractional cover (Fig. 3).

The use of high temporal frequency optical imagery, such as MODIS, for rice mapping has recently been exploited as a monitoring tool. Primary advantages of MODIS measurements are the frequent repeat intervals of the satellite platform. The MODIS Science Team provides a suite of products including surface reflectance with bands that are sensitive to water and vegetation. Xiao et al. (2005) developed an approach based on 8-day MODIS indices to map rice and agro-ecological attributes. The algorithm, based on multitemporal MODIS (Fig. 2), has been successfully employed in regions in South and Southeast Asia and California for mapping rice and rice hydroperiod over large areas (Xiao et al. 2006; Torbick et al. 2011). Sakamoto et al. (2008) also found multitemporal MODIS useful to map rice and assess flood patterns agriculture in the Mekong Delta.

One established technique (Fig. 4) uses the dynamic relationship among the Land Surface Water Index (LSWI), Enhanced Vegetative Index (EVI), and Normalized Difference Vegetation Index (NDVI) generated from optical data such as 8-day or 16-day MODIS (Biradar and Xiao 2011, Xiao et al. 2005, 2006; Torbick et al. 2011). LSWI is an indicator of equivalent water thickness of the presence of water, that is generated by ratioing observations that are sensitive to vegetation, moisture, and water properties. EVI ratios observations that are sensitive to vegetative surface conditions and incorporates additional spectral information to normalize soil and

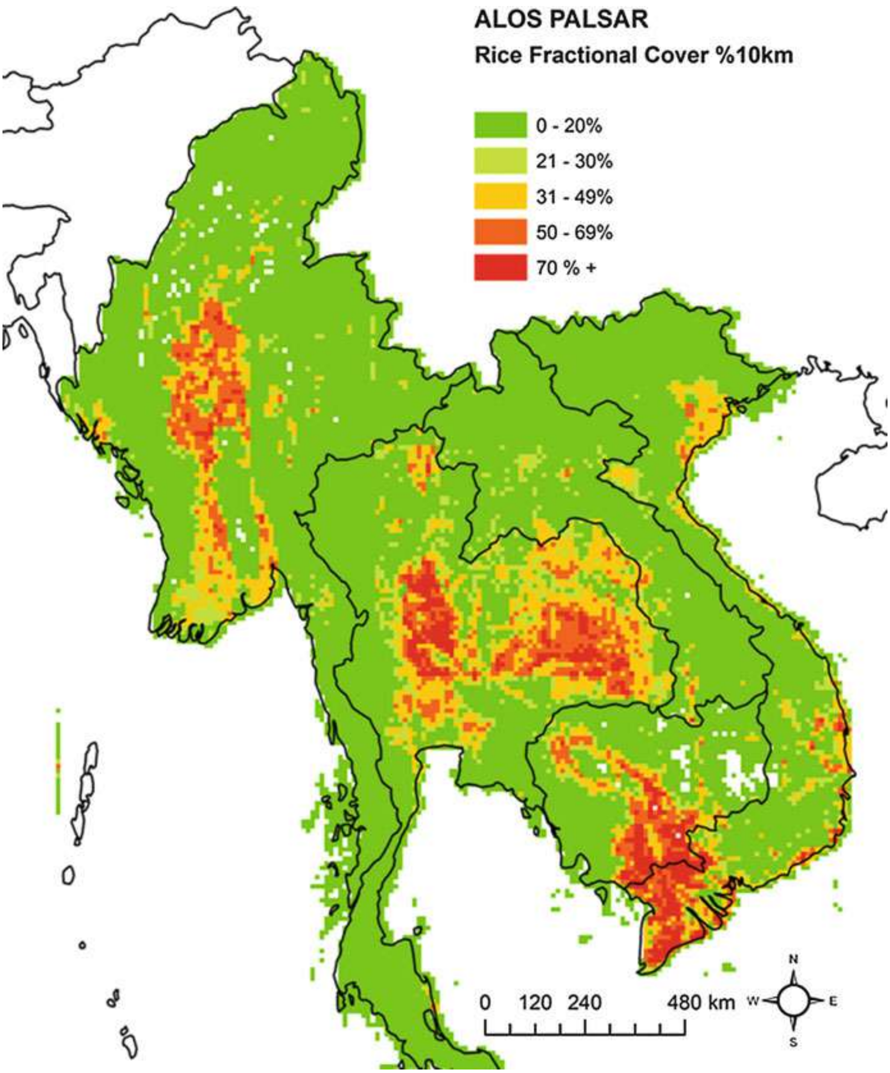

Fig. 4 Fractional cover % at 10 km for Indochina using ALOS PALSAR. The major growing areas are well captured by the multitemporal imagery with dense rice regions mapped across the Mekong and Red River Deltas, Chao Phraya and I-San Plateau, and coastlines

background effects (Xiao et al. 2005). By recording the inversion of LSWI and NDVI, an indicator of paddy inundation can be mapped; however, the LSWI requires spectral information from the shortwave infrared band which is available at 500 m when using MODIS. Spectral matching techniques and wavelet approaches have also utilized similar scale data for large area rice mapping (Sakamoto et al. 2008). By generating accurate maps of rice extent and cropping intensity, estimates of rice production are feasible.

Future Challenges

Mapping hydroperiod, cropping intensity, and production with operational approaches is a near term challenge. Food security programs desire accurate and rapid information to address famine, disaster, and response planning. As more operational optical and SAR programs come online, operational map products can support food security needs. Greenhouse gas (GHG) emissions are becoming a driving factor that influence flood management decisions in many regions. As policies and management practices that abate emissions are developed, spatially explicit mapping tools will be needed to support cost effective assessment. Alternative Rice Managements (ARM) – such as bailing rice residue, winter flooding practices, dry seeding, and early drainage prior to harvest- are all being considered as potential adaptation and mitigation strategies. Remote sensing tools are being evaluated as components of Monitoring, Reporting, and Verification (MRV) protocol and mapping ARMs will be an integral part by providing low cost spatiotemporal records for accounting platforms. Finer resolution, near-real time, and accurate rice maps will continue to be useful information products to many end users.

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Remote Sensing of Anthropogenic Activities: Aquaculture

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Lisa-Maria Rebelo

Contents

Abstract

While many wetlands are still observed in their natural state, humans have exerted a considerable influence on their extent, condition, and dynamics. This section focuses on remote sensing techniques that have been used to characterize, map, and monitor these impacts which arise from agricultural production, aquaculture, dam and reservoir construction, drainage, harvesting of aquatic resources (e.g., wood and charcoal), hydrological and sediment diversions, and salt production. Several techniques have also been developed to contribute to the restoration and creation of wetlands.

Keywords

Human impact · Agriculture · Aquaculture · Hydrological modification · Restoration

Introduction

Earth observation from satellites offers potential for aquaculture applications, including the selection of sites, management of aquaculture ponds, and assessment of impacts. While there are many forms of aquaculture in coastal areas as well as in many inland lakes and wetlands, here the focus is on shrimp farming in ponds.

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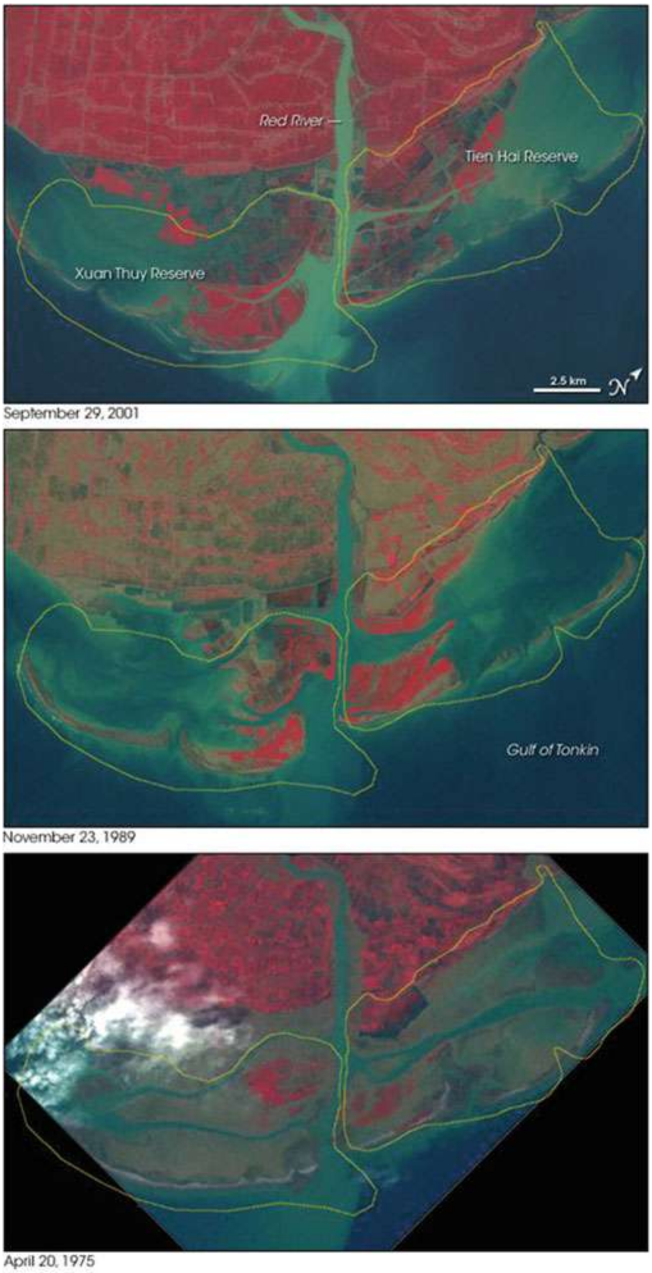

Fig. 1 Landsat false colour composites of the Xuan Thuy Wetland Reserve (Ramsar site) and Tien Hai Nature Reserve, Vietnam, acquired in April 1975, November 1989, and September 2001. *Red* indicates vegetation, *brown* indicates relatively bare ground, *blue-green* indicates shallow and/or sediment-laden water, and *deep blue* indicates deeper, clearer water. Both nature reserves are outlined in *yellow* (NASA 2008)

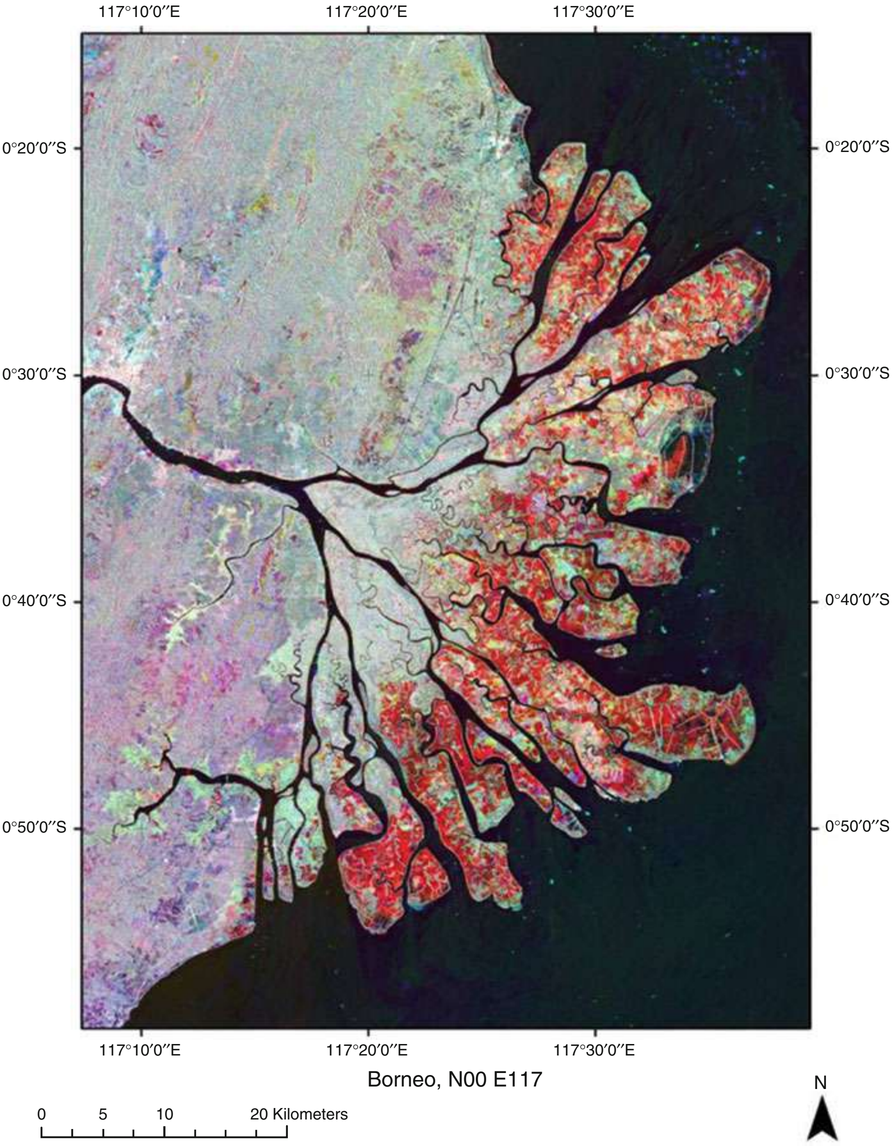

Fig. 2 A composition of JERS-1 SAR (1996) and ALOS PALSAR data (2007 and 2010) showing the loss (coloured *red*) of mangroves to aquaculture in the Mahakam Delta of Indonesia (© JAXA METI)

Mangroves are commonly replaced by shrimp farms although often it is the flats behind mangroves that are exploited. Both optical and radar remote sensing data have proved popular for inventorying, monitoring and assessing the impact of aquaculture ponds in mangroves. These ponds are typically rectangular in shape and filled with water and are thus readily detected using both moderate spatial resolution optical and radar imagery because of the distinct signatures in both. Various operational decision support systems exist using remote sensing data as input.

Monitoring Shrimp Farm Development Using Remote Sensing

The European Space Agency (ESA), Food and Agricultural Organization (FAO) and the Government of Sri Lanka undertook one of the first projects to create an operational system to address the uncontrolled proliferation of shrimp farms, which were having a detrimental impact on the environment. This involved regular mapping of the shrimp farms from ERS SAR data to register their increase and growth, and to monitor encroachment into conservation areas (Travaglia et al. 1999), and has subsequently been expanded to include Southeast Asia (Travaglia et al. 2004). While radar data have received more attention for mapping aquaculture in mangroves as these ecosystems are frequently cloud covered, optical remote sensing images have also been used typically in the assessment of long term change. Images from NASA's Landsat (NASA 2008; Fig. 1) have been used to quantify the changing extent of aquaculture within Ramsar wetland sites containing mangroves in Vietnam over a 25-year period (Seto and Fragkias 2007). Time-series of JERS-1 SAR and ALOS PALSAR data have also been used for mapping the changing extent of aquaculture and mangroves in the Mahakam Delta of Indonesia (Fig. 2; Thomas et al. 2014). Here, substantial losses of mangroves occurred since 1990, with this also highlighted by time-series comparison of Landsat sensor data.

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Remote Sensing of Wetland Types: Arctic and Boreal Wetlands

225

Daniel Clewley

Contents

Abstract

Initiatives aimed at studying change in arctic and boreal wetlands aim at (1) producing snapshots of lake change; (2) upscaling analyses to regional and panarctic levels; (3) linking earth observation data to permafrost degradation modelling. Because of the limitations of optical data, radar data (notably SAR) have become popular for mapping and change detection in boreal and arctic regions. A wide of methods has been used, including combinations of optical and SAR data and different classification algorithms to separate vegatation and hydrogeomorphic characteristics. Examples of applications to wetlands in Alaska and eastern Siberia are reviewed.

Keywords

Arctic wetlands · Boreal wetlands · Earth observation · Synthetic aperture radar · Permafrost

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Introduction

A number of initiatives have been aimed at establishing baseline and change maps of wetlands for the arctic and boreal regions, with these including the MEaSUREs project and the European Space Agency (ESA) Data User Element Permafrost (DUE Permafrost). This latter project is focusing on five to seven monitoring sites with the aim of providing: (a) snapshots of lake-change magnitudes and directions (including fine scale thermokarst changes), (b) options for upscaling analyses to regions and the pan arctic, and (c) linkages with permafrost degradation modeling. These projects have used combinations of optical and radar data at varying temporal and spatial resolutions.

Optical Data

Optical data have been used widely for observing arctic and boreal wetlands. However, limitations are that cloud cover often prevents observation and water levels change between seasons (e.g., because of snow melt). Long time-series are also often needed to generate large area mosaics. However, these issues can be largely overcome using Synthetic Aperture Radar (SAR) data because of its ability to penetrate through cloud and operate regardless of illumination conditions. Mosaics that are relatively consistent in time can also be generated across large regions.

Moderate (5–30 m) spatial resolution data are often used for detecting water bodies in these regions, with the exception of those that are small in area. For this purpose, subpixel methods or textural measures have been used. Within fine (<5) m resolution data, very shallow lakes are difficult to detect from panchromatic data (as provided by the historical Key Hole sensors, e.g., Corona and Hexagon) but can be discerned in near infrared data (e.g., as provided by RapidEye sensors). There is also a general lack of archival EO data available for change detection, although the "Key Hole" (Corona, Hexagon, Argon, and Lanyard missions and the KH Mission 1-9) series of very high resolution (VHR) data is of particular importance considering archives extend back to the 1960s and 1970s.

Radar Data

Over the boreal regions, the use of Synthetic Aperture Radar (SAR) data for mapping wetlands and detecting change has been advocated. As an example, a Random Forests technique was applied to Japanese Earth Resources Satellite (JERS-1) SAR and Advanced Land Observing Satellite (ALOS) Phased Arrayed L-band SAR (PALSAR) to map wetlands across Alaska, building on previous work undertaken by Whitcomb et al. (2009a, b, c; Fig. 1). For this purpose, mosaics of these data for Alaska and also Canada have been generated through the Global Boreal Forest Mapping (GBFM) project and also the Kyoto and Carbon (K&C) Initiative. These data were used for classification, following refinement of registration procedures, with the acquisition date also considered because of differences in backscatter between image dates as

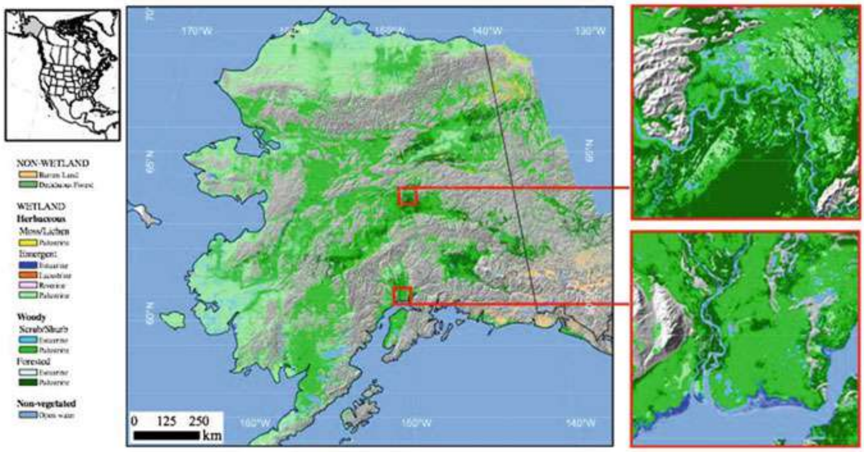

Fig. 1 Map of wetlands in Alaska (boundary shown by *polygon*) and extending into Canada (Based on 2007 ALOS PALSAR data; Clewley et al. 2015)

a function of environmental (e.g., surface moisture) conditions. Elevation and slope data were also included as a means of separating wetlands occurring in the lowlands and uplands and confining the analysis to only those areas where wetlands occurred (i.e., slopes $<3.8^{o}$). A water mask was generated a priori by applying a rule-based classification to the JERS-1 SAR and ALOS PALSAR data (thresholds of -12 dB and -14 dB, respectively), giving consideration to the slope layer (with water occurring on slopes $<3^{o}$). Distance to water was also included. The classification itself was based on the Cowardin system which separated wetlands into estuarine class (tidal, riverine, lacustrine, and palustrine), and within each geomorphological class, the vegetation was divided according to whether it was dominated by mosses or lichens, emergent vegetation (primarily sedges and cotton grass), scrub/shrub, or forest. A modifier was then applied to each geomorphic and vegetation class to indicate hydrological changes that were occurring, although this required temporal data which was difficult to obtain based on the SAR data alone. Non-wetland areas were also included in the classification process. Separate maps were generated using the Random Forests approach and based on the JERS-1 SAR and ALOS PALSAR data, with overall accuracies of 85.3 and 85.4%, respectively. The comparison of the maps allowed wetland dynamics to be considered across Alaska, with many lakes appearing to have reduced in size or disappearing altogether and a transition from emergent vegetation to shrub/scrub and from shrub/scrub to forest also occurring.

Classification Methods

For mapping surface waterbodies in boreal and arctic regions, a wide range of methods have been adopted. For example, Duguay and Pietroniro (2005) mapped lake depth and lake ice thickness by integrating optical and SAR data. Focusing on the coast of north eastern Siberia, Grosse et al. (2005) used manual interpretation and thresholding techniques to classify water bodies from Corona data. Thaw lakes and drained thaw lake basins were mapped in Alaska using object-oriented methods by Frohn et al. (2005). Hese et al. (2010) also used RapidEye data to differentiate water areas, with the spatial resolution being superior to Landsat sensor data.

Sören and Schmullius (2009) used declassified photo-reconnaissance imagery from the 1964 Corona Key Hole (KH-4A) sensor with Landsat MSS data and also 2004 Quickbird data to map change in the Lena River delta of northern Siberia. A similar approach by Hese et al. (2010) used 1973 Hexagon (1970s) and RapidEye data (2009/2010), with the inclusion of multispectral data (through object-based segmentation) considered essential for differentiating very shallow lake areas from deep water and also vegetation that is spectrally similar. Registration was achieved using centroid points from lake polygons and a spatial join based on distance. To create normalized grey values for all objects with each set of imagery, the water mean normalization (WMN) technique was used whereby:

$$\mathrm{DN}_{water\_form} = (\mathrm{DN}/\mathrm{DN}_{lake\ mean}) * 100$$

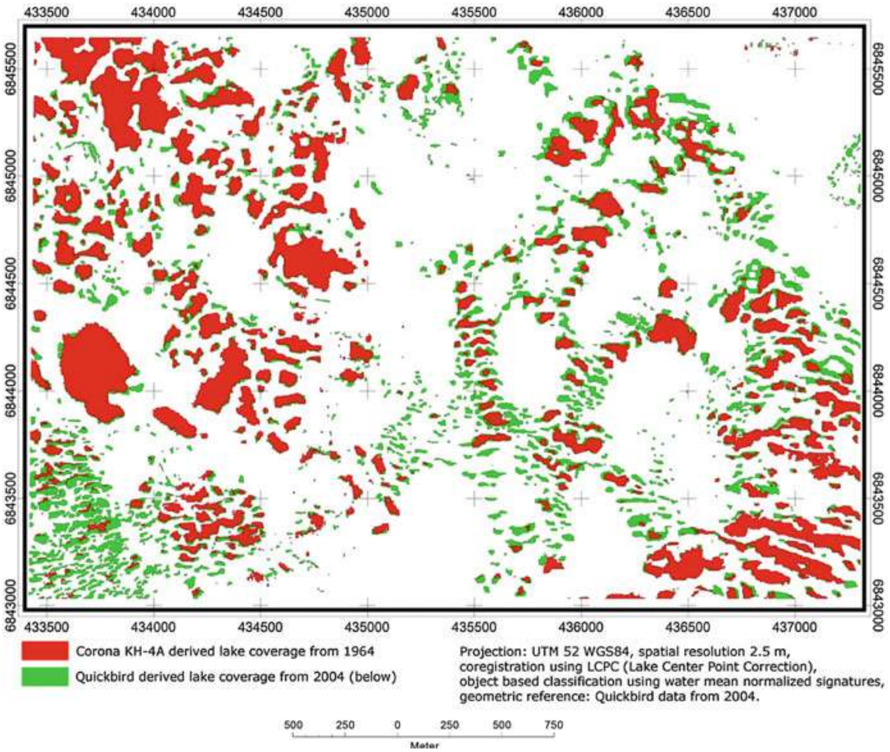

Fig. 2 Changes in the distribution and dimension of lakes generated by comparing Corona and Quickbird data from 1964 to 2004, respectively (Hese and Schmullius 2009)

where DN_{water_form} is the normalized reflectance scaled for the 16 bit data space (as floating point), the DN is the original reflectance value, and DN_{lake_mean} is the mean grey scale value for water. Changes in the lake objects were based on lake area (m^2), border index values (i.e., the ratio of the perimeter of a fitted box and the actual object perimeter), the elliptical fit (the ratio of the length to width of an ellipse built from the same area of the polygon and comparing the area outside and inside of the ellipse), the length to width ratio (using a bounding box concept), the object direction (the main direction of an object generated using the larger of two Eigen vectors from the covariance matrix) (0–180°), and roundness (a measure of fit of a bounding ellipse). Two structural lake features include the number of neighboring lakes within varying search radii (e.g., 30 m, 60 m, etc.) and the standard deviation of the area values of all lakes within 200 m of every lake, which is described as the homogeneity of the size of lakes. Within multitemporal data, differences in the homogeneity/heterogeneity of area values in neighboring lakes indicate a change. Water objects observed in both or either 1964 and 2004 were extracted using intersect functions and queries, with the mapping indicated in Fig. 2. The comparison indicated an increase in lake density, variability in lake sizes, and changes in shape direction and density (Fig. 3). The change in lake direction was attributed in part to a change in wind direction.

For classifying wetland vegetation in the Lena River, Hese et al. (2010) used RapidEye data and a rule-based approach based on thresholds of the Normalised Difference Vegetation Index (NDVI), a combined ratio with the RapidEye red edge channel and a supervised nearest neighbor nonparametric classification based on all

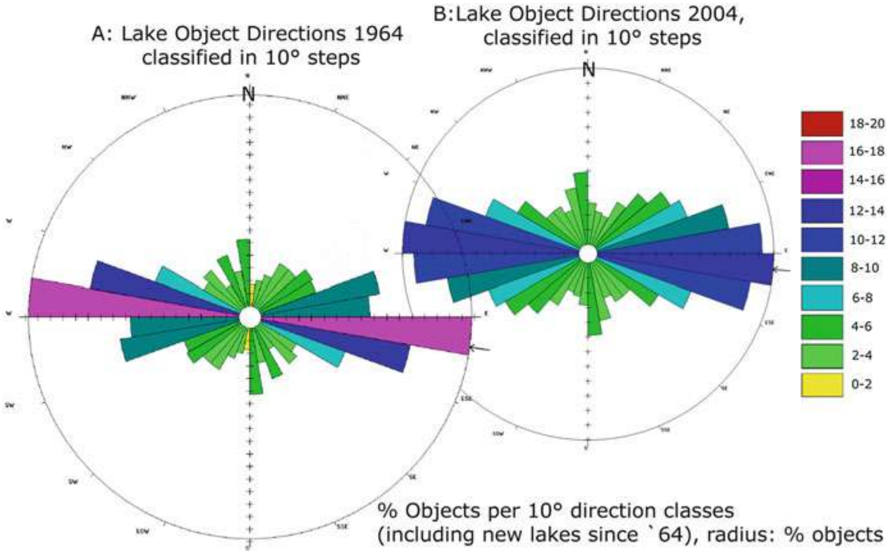

Fig. 3 *Rose* diagram showing the main lake directions (classified in 10° intervals). The radius represents the percentage of lake objects (color codes) with lake directions in 1964 and 2004 indicated (Hese and Schmullius 2009)

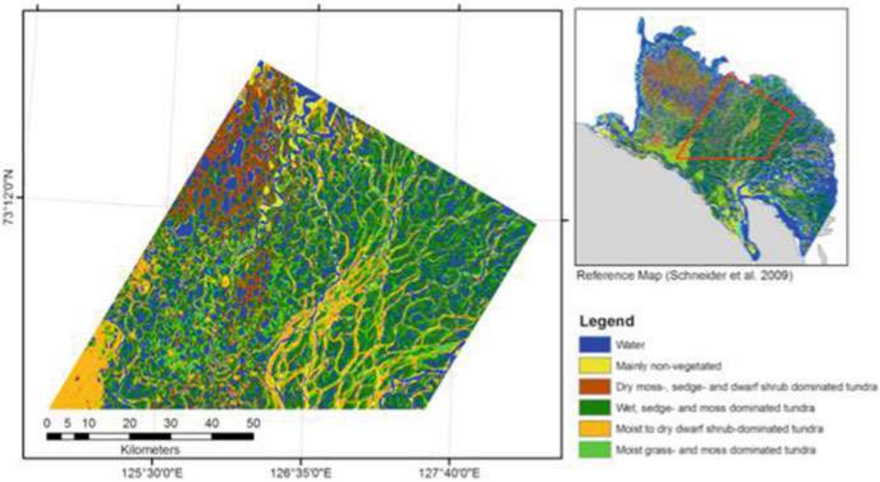

Fig. 4 Classification of the Lena River generated using RapidEye data

RapidEye channels (excluding the green). Separation was also increased with the GLCM Homogeneity and correlation measures (Fig. 4).

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Remote Sensing of Wetland Types: Mangroves

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Richard Lucas, Lola Fatoyinbo, Marc Simard, and Lisa-Maria Rebelo

Contents

Abstract

Mangroves are characteristics of coastlines and occur primarily in the tropics and subtropical but extend to temperate regions. Remote sensing radar and optical and lidar data can be used to provide information on mangrove extent and also biophysical characteristics. Optical data are the most useful for differentiating species type, while radar and lidar data can be used, either singularly or in combination, to retrieve the three-dimensional structure (canopy height profiles,

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cover, and stem number density). Data from these sensors also provide information on changes in extent and biophysical properties, which can be applied at local to global levels. Lower frequency L-band radar data are particularly useful given their ability to operate regardless of weather and illumination conditions and significant efforts are being made to map the changing extent of mangroves at a global level using these data. Examples of capability are provided for Brazil, Australia, and South America.

Keywords
Mangroves · Radar remote sensing · Optical remote sensing · Lidar data

Introduction

The characterization, mapping, and monitoring of mangroves based on remote sensing data has proved challenging as the prevalence of cloud cover in many areas they inhabit limits observations by optical sensors, diurnal changes in tidal inundation lead to variability in their appearance, and similarities with other proximal vegetation types often compromises their discrimination. The spatial distribution and geometric arrangement of mangroves is also highly variable and ranges from small, isolated clumps of trees and narrow strips that often parallel the coast to contiguous forests covering hundreds of kilometers. For these reasons, a wide range of remote sensing data has been exploited for mapping extent, retrieving important biophysical variables and monitoring change.

Extent

For mapping extent, optical data are typically used with the inclusion of shortwave infrared channels often providing best discrimination from adjoining natural, managed, or plantation forests and shrublands. By comparison, SAR data acquired at X-, C-, and L-band backscatter data have proved to be less useful because of similarities in response. Data from very high resolution (VHR) multispectral sensors allow complex distributions of mangroves to be resolved and are frequently exploited of mapping and monitoring at local sites. However, the majority of mapping has been undertaken using data from Landsat sensors, with these often collated into regional or global products (e.g., Spalding et al. 1997, 2010; Giri et al. 2011).

Biophysical Characteristics

Key structural descriptors of mangroves include cover and height. Mangroves typically support a high density of trees and leaves/branches within the canopy and hence canopy cover often exceeds 80%. Using Landsat-derived Foliage Projective Cover (FPC), for example, discrimination from forests of lower cover can be achieved. For height retrieval, stereo photography (Lucas et al. 2002; Mitchell et al. 2007) and both airborne and spaceborne InSAR and LIDAR have been used. For example, Mitchell et al. (2007) and Held et al. (2003) demonstrated the use of historical stereo photography and airborne InSAR data for height retrieval. Other sources of elevation information include stereo PRISM and Tandem-X InSAR data. Simard et al. (2008) and Fatoyinbo and Simard (2013) utilized SRTM corrected with IceSAT GLAS data to generate regional maps of mangrove canopy height. Based on an established relationship with ground-measured height, maps of biomass were generated first for areas in South America (Simard et al. 2006) and Africa (Fatoyinbo et al. 2008) and subsequently globally. The aboveground biomass (AGB) of mangroves can also be retrieved from lower frequency L and P-band SAR as the backscattering coefficient typically increases asymptotically until the signal is saturated (typically at ~50–100 Mg ha^{-1} and 100–150 Mg ha^{-1}, respectively). However, for mangroves with large prop roots systems, typically associated with *Rhizophora*

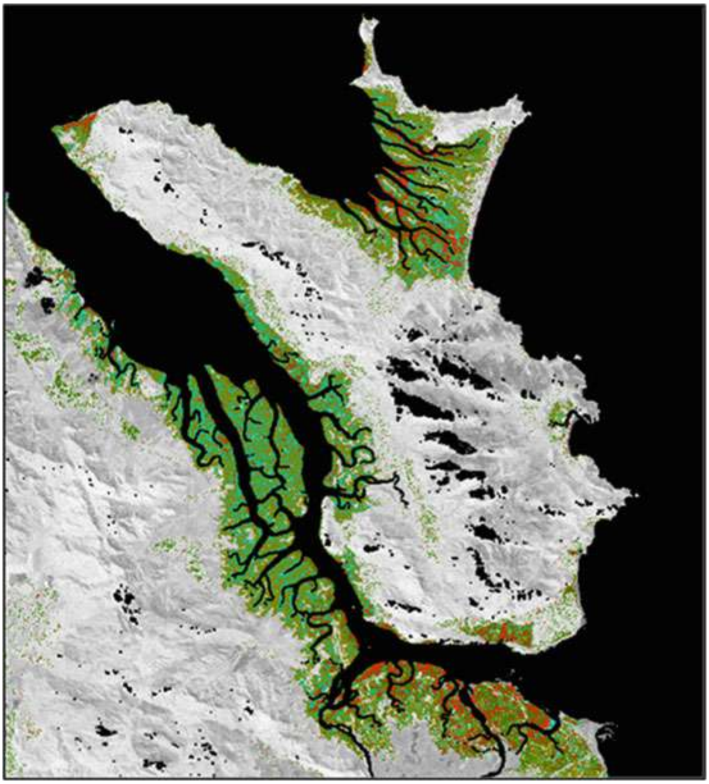

Fig. 1 Classification of mangroves biomass classes generated using a combination of SRTM height and ALOS PALSAR L-band data. Mangroves of lower biomass are defined as having a tree height of <=10 m while those above 10 m and of higher biomass are differentiated into those with (*red*) or without (*olive*) prop root systems, with the former typically dominated by *Rhizophora* species (© JAXA METI)

species, the backscattering coefficient is often lower compared to those without and decreases as the AGB increases (Lucas et al. 2007). This peculiarity of mangroves can also be exploited to assist the discrimination and mapping of tree species that dominate the mangrove zones (Fig. 1), although the diversity of types able to be detected increases when optical imagery are integrated into classification procedures. The ability to discriminate species types improves where optical data are acquired at higher spatial resolutions and hyperspectral sensors are used.

Change

Across their geographical range, mangrove forests are responding to natural changes in the coastal environment although these can be exacerbated by climate change. Changes are most notable in areas where mangroves remain relatively undisturbed from human activities. In many regions, mangroves have provided protection against storm surges and tsunamis, reducing loss of life and damage to infrastructure, agriculture, and livelihoods. For these different rates and patterns of change to be

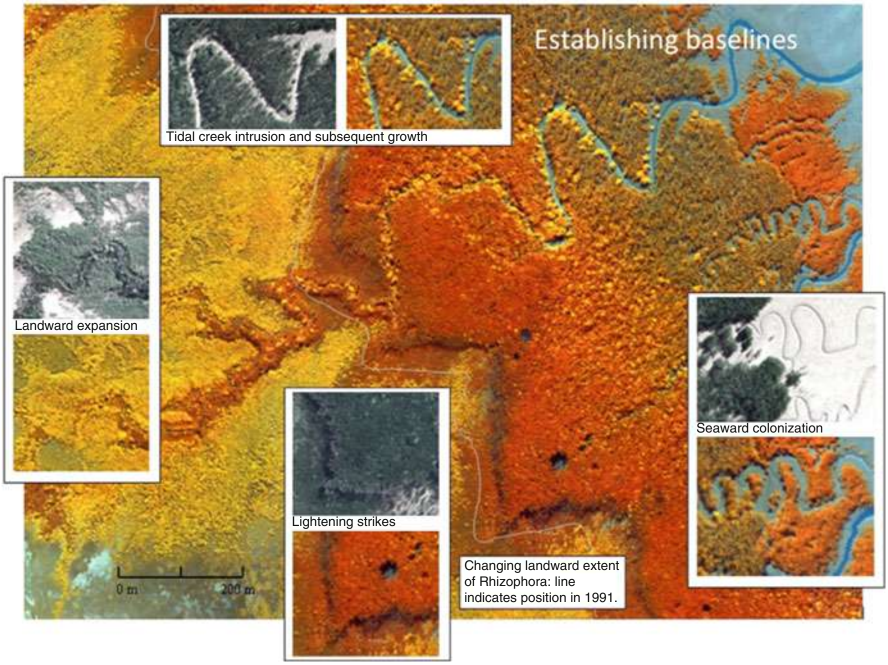

Fig. 2 Mangroves along the western bank of the West Alligator River, Kakadu National Park, northern Australia, as observed from CASI data. Dominating species are *A. marina* (*yellow*), *R. stylosa* (*red*), and *S. alba* (*olive*). Mangroves on the seaward side are dominated by younger colonizing *S. alba*. The primary changes observed when compared with aerial photography acquired in 1991 include landward and seaward expansion and loss of cover through lightning strikes and storm surges (Reproduced with permission from Lucas et al. (2017))

detected, sensors operating at a range of spectral, spatial, and temporal frequencies are required. VHR airborne or spaceborne imagery provide the level of detail necessary for the detection of small-scale and relatively localized change as exemplified in Fig. 2. In this example, which compares true color aerial photography and hyperspectral Compact Airborne Spectrographic Imager (CASI) data acquired over the West Alligator River in Kakadu National Park, northern Australia, mangroves dominated by *Sonneratia alba* are colonizing mudflats on the seaward margin while *Avicennia alba* is advancing inland and expanding in cover. Significant losses of the taller mangroves dominated by *Rhizophora stylosa* are the result of lightning strikes and storm surges.

Time consistent detection of change using moderate spatial resolution optical (e.g., Landsat) sensors is often compromised by cloud cover, but SAR allows observations under such conditions and also at night. For monitoring of mangroves, data acquired by the Japanese Earth Resources Satellite (JERS-1) SAR and Advanced Land Observing Satellite (JERS-1 SAR) have proved beneficial as the systematic observation strategy has allowed regular and consistent global observations. In French Guiana, significant redistribution of mangroves has been observed as a consequence of sediment erosion and accretion. By comparing time-series of JERS-1 SAR and ALOS PALSAR data, the gains and losses of mangroves can be observed (Fig. 3). However, and is the case of French Guiana, optical data are generally required for generating baselines of mangrove extent against which to assess change because of the difficulty in mapping mangroves using SAR data alone. These data can either be used singularly (e.g., Spalding et al. 2010; Giri et al. 2011)

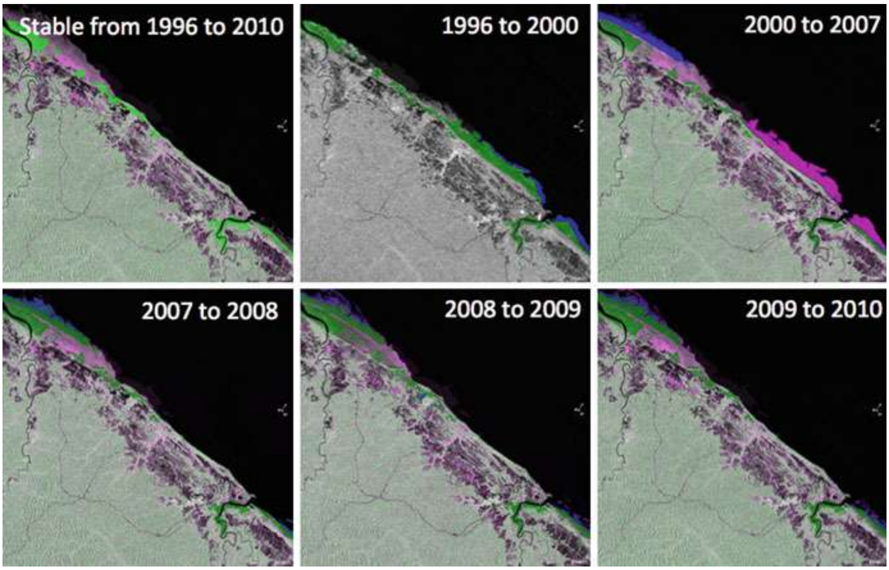

Fig. 3 Changes in the mangrove area from an existing baseline (Giri et al. 2010) showing losses (*blue*), gains (*magenta*), and stable areas (*green*) (© JAXA METI)

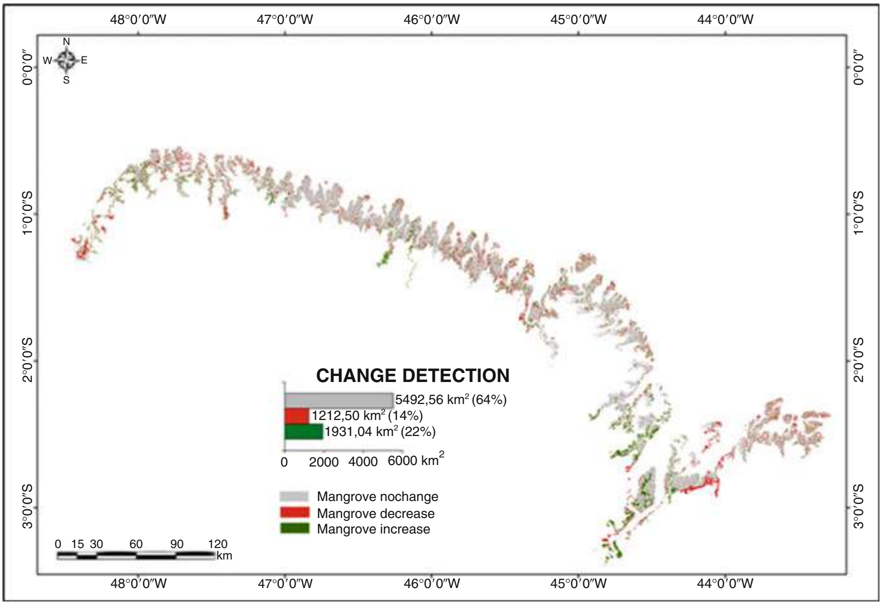

Fig. 4 Amazon mangrove forest change from 1996 to 2008 (Nascimento et al. 2013)

or in combination, with Nascimento et al. (2013) using Landsat TM/ETM+, JERS-1 SAR, and ALOS PALSAR data to confirm an overall increase in the area of mangroves along the Amazon Macrotidal Coastal Zone of Brazil (Fig. 4).

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Remote Sensing of Wetland Types: Peat Swamps

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Dirk Hoekman

Contents

Abstract

Deposits of peat underneath peat swamp forests are among the world's largest reservoirs of carbon. Although tropical peatlands occupy only about 0.3% of the global land surface, they could contain as much as 20% of the global soil carbon stock, representing 63–148 Gt of carbon.

Peat swamp forests are among the worlds most threatened and least known ecosystems. In Southeast Asia large areas of peat swamp forest have been deforested, converted for agricultural projects or into plantations (such as oil palm).

Drainage through canalisation has frequently severely disrupted water table level dynamics, resulting in CO_2 emissions due to oxidisation and vulnerability to fire, especially during 'El-Niño' years. Water management is essential in addressing disturbances and rehabilitation of degraded tropical peatlands.

Radar satellite observations can be made frequently, also in the wet season. Because of a certain level of penetration of the radar waves, also observation

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below the canopy is possible. Particularly the L-band sensors on board the former JERS-1 and ALOS-1 satellites are superior to all other spaceborne sensors for assessment of flooding and drought conditions and, thus, hydrological cycles.

Keywords
Tropical peat swamp forest · Ecosystem · Deforestation · Drainage · Satellite observation · SAR

Introduction

In the millennia after the last ice age, thick deposits of peat developed in different periods and under different abiotic circumstances. Due to these distinct characteristics, these deposits are classified in three types: high peats, coastal peats, and basin peats (Rieley 1991; Table 1). Formation of high peats started about 9,000 years ago in the interior of Kalimantan, in elevated watersheds (30–50 m asl), after long periods of rainfall. Formation of coastal peats started more recently, around 5,000 years ago, after sea level ceased rising (Rieley 1991; Silvius et al. 1984). Coastal accretion took place at a rate of 9–20 m per year (Silvius et al. 1984; Whitmore 1975), but in certain areas and periods accretion up to 100 m per year occurred (Whitten et al. 2000). Initially, coastal plains were colonized by mangroves. Within these mangrove swamps, abiotic conditions eventually became unfavorable for micro-organisms and the process of biodegradation. Lack of oxygen, constant water logging, and the presence of toxic substances decreased the rate of decomposition. In this way, topogenous peat was formed and other plants gradually replaced typical mangrove species, which prefer a clayey soil. As accumulation of organic materials proceeded, the peat became ombrogenous (purely rainwater-fed and nutrient poor). As the peat layer developed further and the distance to the sea increased, a transformation to basin peats took place (Whitten et al. 2000).

Extent

Deposits of peat underneath peat swamp forests are among the world's largest reservoirs of carbon. Although tropical peatlands occupy only about 0.3% of the global land surface, they could contain as much as 20% of the global soil carbon stock, representing 63–148 Gt of carbon (Rieley and Setiadi 1997; MacDicken 2002). It should be noted that tropical peat swamps are unevenly distributed across the globe. The tropical peat swamp forests of Southeast Asia account for approximately 26.5 Mha of the total tropical resource of 38 Mha, with Indonesia alone contributing an estimated 17–27 Mha (Waldes and Page 2002). Following internationally accepted criteria for the definition of peat (i.e. at least 30% organic matter in a layer of at least 40 cm), the total area of peat soil in Southeast Asia is estimated at

Table 1 Different types of Southeast Asian peat (cf. Whitten et al. 2000)

Type	Hydrology	Location	Sub-soil
High peats	Ombrogenous	Elevated watersheds	Podsolised sandy soil
Coastal peats	Starting topogenous, evaluating into ombrogenous	Coastal areas	(Reduced) clay
Basin peats	Ombrogenous	Coastal areas and inland	Coastal peat and clay

Table 2 Estimated pre-disturbance tropical peat land area in millions of hectare (cf. Whitten et al. 2000)

Region	Country	Area
Africa, America		17 Mha
SE Asia	Brunei	0.01 Mha
	Indonesia	27 Mha
	Malaysia	2.7 Mha
	Papua New Guinea	2.9 Mha
	Philippines	0.24 Mha
	Thailand	0.07 Mha

33 Mha. In other areas of the world, small tracts of peat swamp forests are located in South and Central America (Brazil, Guyana, Costa Rica), in parts of the Caribbean (Jamaica, Cuba), and in Africa (Burundi, South-Africa). The estimated pre-disturbance tropical peat land area is shown in Table 2 (Rieley et al. 1994).

Biophysical Characteristics

A distinction is made between ombrogenous "peat swamps" (rain fed) and topogenous "freshwater swamps" (flooded by rivers). Within the ombrogenous peat swamp, there is a decrease of mineral nutrients towards the center (Polak 1933). This trend of increasing infertility towards the center is reflected in the vegetation by decreasing canopy height, girth (of certain species), and biomass and increasing leaf-thickness. Thus, there is no single type of peat swamp forest but a gradual change of forest types (Anderson 1964; Whitmore 1975), covering a large range of variation in biophysical parameters (such as biomass) as well as a large range of growth conditions (reflected in growth rate and carbon sequestration potential). Accurate vegetation maps showing these wide ranges of conditions within the peat swamp vegetation type are still lacking. Peat swamp forest in Sumatra and Borneo are often compared to lowland Dipterocarp forests as both have many plant and animal species in common. However, being poor in nutrients, peat swamp forests support far less species than lowland Dipterocarp forests. The level of endemism is low as the ecosystem is very young (Whitmore 1975).

Change

Peat swamp forests are among the worlds most threatened and least known ecosystems. In Southeast Asia, large areas of peat swamp forest have been deforested (for timber), converted for agricultural projects (even though the soil is too acid), or converted into plantations (such as oil palm, Borneo rubber). Nevertheless, peat systems are fragile and sensitive to hydrological disturbance (e.g. Hoekman 2007). Drainage through canalisation has frequently severely disrupted water table level dynamics, causing the peat layers to dry out and trees to collapse over large areas. Besides resulting in CO_2 emissions due to oxidisation (Harris et al. 2012; Zarin 2012) this process makes them particularly vulnerable to fire, especially during 'El-Niño' years (Van der Werf et al. 2009). When peat catches fire, a low-intensity, incomplete combustion process is triggered, producing large amounts of smoke and haze. The smoke and haze resulting from combustion of peat has been the primary cause of the high emission levels and severe economic and human health damage during the 1997–1998 fire catastrophe in Indonesia (BAPPENAS-ADB 1999).

Uncontrolled human-induced fires have become a common phenomenon in tropical forest regions throughout the world during the last two decades. Although these wildfires occur over relatively small areas, their impact on the global carbon cycle is enormous (Van der Werf et al. 2009). Emissions from the fires in Indonesia during 1997–1998, for example, have been estimated at 0.8–2.5 gigatonnes (Gt) of carbon. This is equivalent to 13–40% of annual emissions from anthropogenic fossil fuel combustion during that same period resulted in the biggest annual increase in atmospheric CO_2 levels since records began more than 40 years ago (Page et al. 2002; Kool et al. 2006).

Water management is essential in addressing these disturbances. However, the relationship between spatial and temporal dynamics of peat swamp forest hydrology, carbon content and forest health would need further study. Such understanding would not only support the conservation of peat swamp forest, but also the rehabilitation of degraded tropical peatlands, which may significantly reduce carbon emission and fire risk.

Spaceborne Remote Sensing

In the humid tropical regions, optical remote sensing systems largely fail because of persistent cloud cover. LANDSAT is most commonly used but fails to provide useful data every year (e.g. Gastellu-Etchegorry 1988). A recent study using the optical RapidEye data showed a good potential for peat swamp forest inventory and disturbance mapping (Franke et al. 2012). Spaceborne radar observation is not hindered by adverse atmospheric conditions (such as clouds, smoke and haze) and can be made frequently and repetitively, but is still not widely used and relatively unknown. The advantages are considerable however. Observations can be made frequently, also in the wet season, and because of a certain level of penetration of the radar waves, also observation below the canopy is possible. Particularly the L-band sensors on board the former JERS-1 and ALOS-1 satellites (Rosenqvist et al. 2007) are superior to all other spaceborne sensors for assessment of flooding and drought conditions and, thus, hydrological cycles. Moreover, radar signals are sensitive to forest structure and biomass level (Hoekman and Quinones 2002; Hoekman et al. 2010; Englhart et al. 2012; Schlund et al. 2014). This offers unique opportunities for applications such as peat swamp forest health and fire susceptibility monitoring as well as fast illegal logging response monitoring.

As examples of the use of SAR data, Fig. 1 shows that the temporal dynamics in flooding intensity can be related to the hydrology of ombrogenous peat swamp forests and, indirectly, to peat depth (Hoekman 2007). Figure 2 shows interannual variability in flooding in Borneo as observed from time-series of ALOS PALSAR data. The time-series of JERS-1 SAR in Fig. 3 shows the fast succession of events at

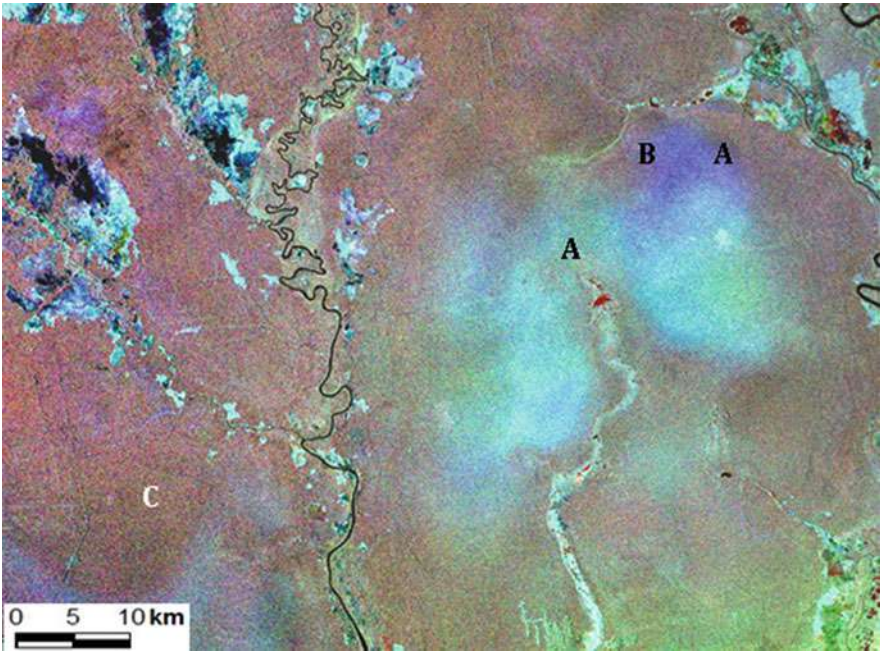

Fig. 1 Temporal dynamics in flooding intensity can be related to the hydrology of ombrogenous peat swamp forests and, indirectly, to peat depth. The *blue* areas labelled as *A* are flooded parts of the relatively flat tops of a complex of two peat domes, with a river originating from a central depression (*B*). The feature labelled as *C* shows the relatively flat and wet fringe of a dry peat dome. Mawas area, Central Kalimantan; JERS-1 SAR multi-temporal composite image (*Red* 7 Sep 1994; *Green* 12 Jul 1995; *Blue* 4 Jan 1996) (© JAXA METI)

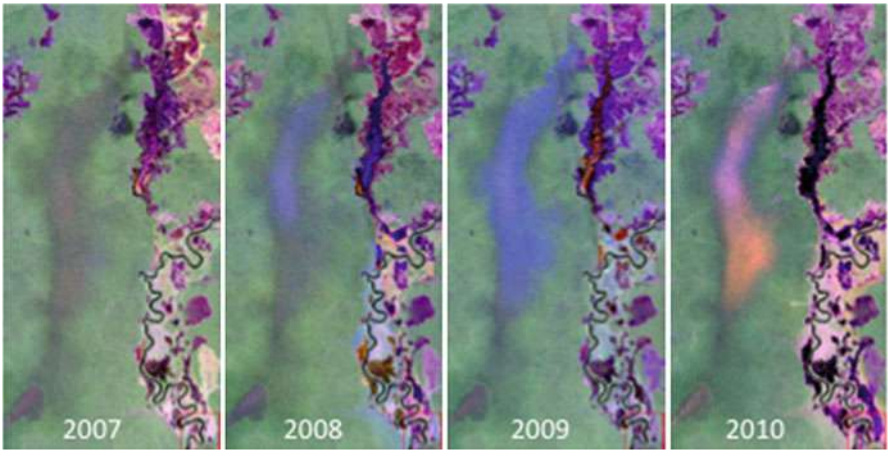

Fig. 2 In peat swamp forests and floodplains, the backscatter dynamics is high and shows major inter-annual variation related to flooding events. PALSAR FBS-FBD; 2007–2010; Borneo (© JAXA METI)

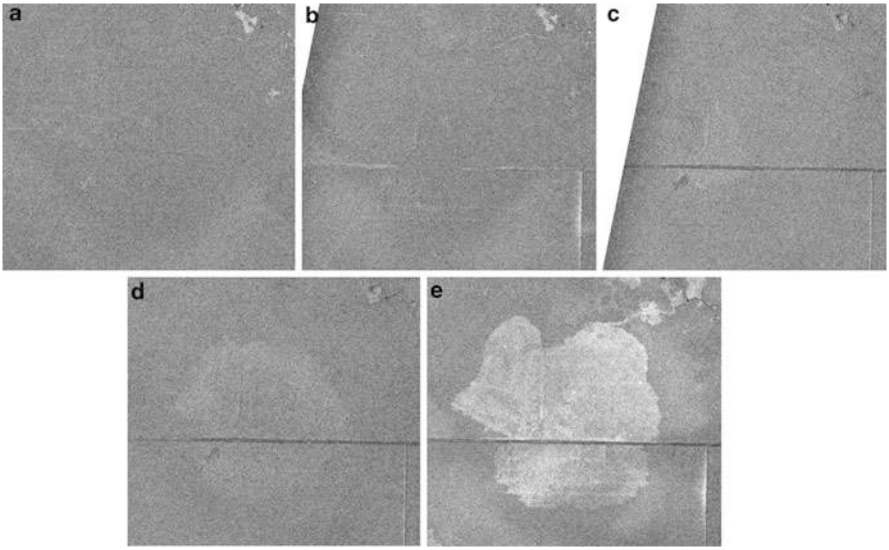

Fig. 3 JERS-1 SAR time series of a forest on top of a peat dome in Central Kalimantan, Indonesia. Dates: (**a**) 950712; (**b**) 970319; (**c**) 970911; (**d**) 971025; (**e**) 980121; (image width 21 km) (© JAXA METI)

a central area of a peat swamp forest (which forms a dome) where all trees fell down because of underground peat fires. This forest collapse can be shown as a time sequence of events. Until 1996 the dome was still hydrologically intact. In 1997, the construction of a very wide canal through the dome is visible. In the third image of

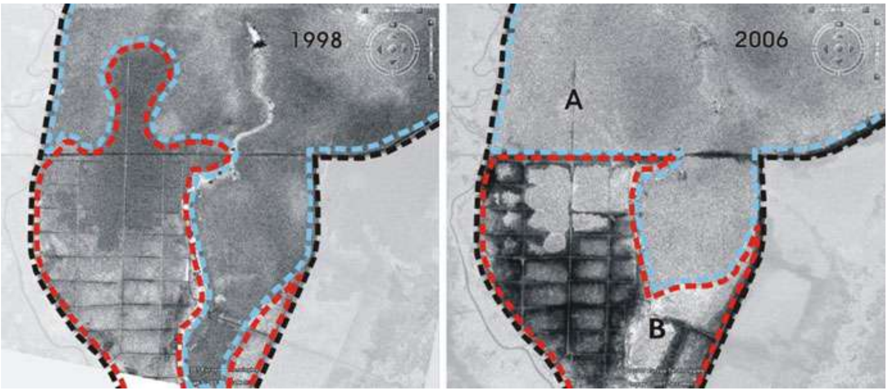

Fig. 4 Peat swamp degradation (**b**) and restoration (**a**) in the Mawas area between 1998 (JERS-1) (*left*) and 2006 (PALSAR) (*right*). The *red* area is degraded, the *blue* area is intact or regenerating (© JAXA METI)

the sequence (September), the canal is filled with water (the canal becomes black) and a small somewhat brighter area appears. This area grows very fast and becomes even brighter until the forest collapse is completed (January 1998). The obvious cause of the collapse is the huge drainage caused by the wide canal. The coinciding strong ENSO period may have accelerated the process (Hoekman 2007).

A lot of information can be derived from analysis of time series of L-band radar data, such as ALOS PALSAR. For many peat swamp areas in Borneo and Sumatra, large series of historical JERS-1 images (i.e. 15–30) collected in the period 1992–1998 exist. Degradation of peat swamps is also shown in Fig. 4. In the JERS-1 image of January 1998 (dry period), the area demarcated by the red line is an area within the Mawas area suffering from excess drought. In the PALSAR image of 9 November 2006 (dry period), this area has decreased above the main east-west canal because of the construction of dams in the canal going North (canal Neraka). In the area south of the main east-west canal, a large network of canals is still present and the continued drainage has worsened the situation. Note the very low radar backscatter (intense black) caused by very dry bare peat areas and the bright white area, which is a strongly degraded open forest with fire damage. The areas demarcated in blue are hydrologically intact, allowing forests previously damaged to regenerate. A classification of the peat swamp forests using TSX/TDX monostatic and bistatic features is provided in Fig. 5 (Schlund et al. 2014).

Since it may be beneficial to stop or even reverse trends of peat swamp forest destruction (see above), it is of utmost importance to have quantitative and accurate data on land cover change processes over the recent past (last decade) into the present. The study of spatial and temporal drainage patterns using remote sensing may lead to better understanding of underlying dynamics in hydrology. This may improve the capacity of monitoring peat swamp forest health and fire susceptibility

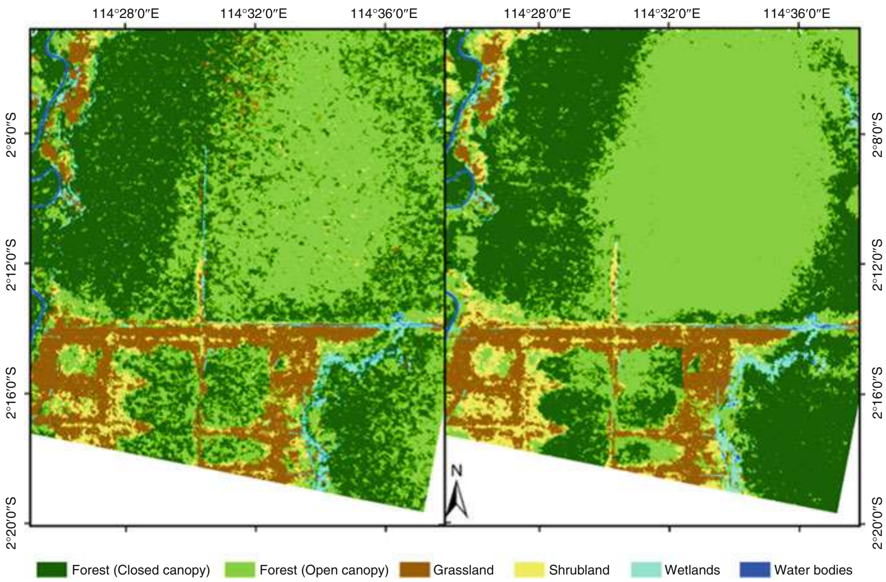

Fig. 5 TSX/TDX classification results for a peat swamp forest area using monostatic (*left*) and bistatic features (*right*). The TSX (monostatic) overall accuracy of 75±4% increases to an (bistatic) overall accuracy of 85±3% for the TSX/TDX combination, showing the importance of interferometric coherence as additional feature

using remote sensing. More specifically, peat swamp forest areas that are high risk for degradation, fire and consequent carbon release can be identified (Hoekman 2007; Hoekman and Vissers 2007).

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Remote Sensing of Wetland Types: Sea Grasses

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Mitchell Lyons and Richard Lucas

Contents

Abstract

Sea grasses are characteristic of coastal waters and play an important role in sustaining biodiversity and also facilitate uptake and storage of carbon. These grasses largely occur in shallow waters that are often clear. Hence, mapping can be achieved using optical remote sensing data with the intricacies of sea grass beds best observed at higher spatial resolutions. Hyperspectral data are especially useful as these can be used to better differentiate sea grass species and productivity types and also facilitate better discrimination from other subsurface environments. Acoustic methods (e.g., SONAR) have also been used and provide additional information on the structure of sea grasses and the underlying topography. Remote sensing data can also be used to indicate human-induced disturbance.

Keywords

Sea grasses · Optical remote sensing · Hyperspectral data · Acoustic methods

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Introduction

Seagrasses occur in shallow coastal waters across the world, and are comprised of single or mixed species meadows that are often interspersed with various other benthic cover types, including macroalgae, mangroves and corals (Green and Short 2003). Seagrass habitats play key trophic and structural roles in maintaining marine ecosystems and associated economic resources (e.g., fisheries) (Short and Coles 2001). With recent emphasis on "Blue Carbon," the importance of seagrass habitat as a carbon sink is also increasingly being recognized (Nellemann and Corcoran 2009). Knowing the changing extent, composition and condition of seagrasses, as well as differentiating these from other submerged aquatic vegetation (SAV) and benthic cover types is important for their conservation and management. Remote sensing provides an opportunity to monitor large areas of seagrass habitat without having to physically survey these and, in many cases, remote sensing approaches are more cost-effective (Mumby et al. 1999). Remote sensing can also enable monitoring when it is infeasible (e.g., too deep, too turbid) or dangerous (e.g., crocodiles, stingers) to survey in the water.

Remote Sensing Techniques for Sea Grass Monitoring

The traditional approach to monitoring seagrasses has been to use vertical multi-spectral (3–4 bands) aerial photography, because of the flexibility of timing and scale of exposure (Short and Coles 2001; Fig. 1). More recent hyperspectral aerial photography have dramatically improved the ability to discriminate seagrasses, as well as their biophysical features such as species composition, percentage cover/

Fig. 1 Sea grasses as observed from the IKONOS sensor, Morten Bay, Queensland

Fig. 2 Spectral measurement of sea grasses

density and leaf area index (Dierssen et al. 2003; Pu et al. 2012; Mishra et al. 2007). Subsurface spectral measurement has also increased understanding of the reflectance characteristics of different species and growth stages (Fig. 2). Early multispectral satellite sensors were less suitable, particularly as data were often acquired at times of the day when conditions were suboptimal (e.g., in terms of cloud cover, tides and wind conditions; Ferguson and Korfmacher 1997). Green et al. (1996) also reported that seagrass is difficult to distinguish from other substrates including mussel beds, algal blooms and detritus and coral reefs. However, more recent satellite image sensors have overcome many of these limitations (Phinn et al. 2008). Very low seagrass density and turbid water can also completely inhibit discrimination from the substrate, which is exacerbated by attenuation of the spectral signature and strength with increasing depth (Lee et al. 1998).

Seagrasses have been observed, mapped and monitored from a wide variety of optical satellite sensors, most of which have not been designed for coastal observations (e.g., Landsat, SPOT). Traditionally, studies have aimed to discriminate seagrass from non-seagrass and establish whether these are submerged or exposed (Green et al. 2008). More recent studies have aimed to also discriminate density of cover and species composition (Gullström et al. 2006; Phinn et al. 2008). Availability of long term satellite image data sets (e.g., Landsat archive) has enabled the use of remote sensing for quantification of long term trends and dynamics in seagrass extent and condition.

Differentiation of species composition within seagrass meadows is problematic at moderate spatial (i.e., pixel size) and spectral (i.e., number of spectral bands) resolutions (Phinn et al. 2008). Fyfe (2003) demonstrated that seagrass species are indeed spectrally distinct, however, optical sensors often do not contain the appropriate spectral or even spatial resolution to capture distinguishing features. Nevertheless, many studies have demonstrated that a range of tropical and temperate species of seagrass can be discriminated in both northern and southern hemisphere coastal environments, with overall accuracies in the range of around 50–80%. The most reliable approach appears to be the use of hyperspectral aerial photography, although the SPOT, IKONOS, Quickird-2 and Worldview-2 multispectral scanner have been used (Fig. 1), while multispectral satellite sensors have been shown to be the most cost-effective (Mumby and Edwards 2002).

Seagrass habitats have also been mapped using acoustic remote sensing methods, most notably Multi-beam Sonar (Komatsu et al. 2003) and Side Scan Sonar (Montefalcone et al. 2013; Sagawa et al. 2008). Similarly to passive optical remote sensing approaches, acoustic mapping methods have been shown to be able to discriminate seagrass extent and species type, with the added potential to measure canopy height and structure. Acoustic data may also be integrated with optical remote sensing data to improve seagrass mapping approaches. Use of acoustic sensors may enable monitoring when the water is otherwise too dark or too turbid for optical sensors, although many acoustic sensors require several meters of water depth to operate effectively, which inhibits their use in many seagrass environments, particularly intertidal habitats.

Application of Remote Sensing for Sea Grasses

Remote sensing data can be used for detecting changes in seagrasses, which have been substantial and attributed, in part, to human-induced disturbance including increased turbidity due to eutrophication and sediment load as well as physical removal such as dredging and clearing. For example, Dekker et al. (2005) used multi decadal Landsat Thematic Mapper (TM) imagery to detect changes in the extent of various seagrass species and community types. Increasing availability of imagery and methods are providing monitoring solutions at a large range of spatial and temporal resolutions. Detailed spatial and temporal information about seagrass communities derived from remote sensing approaches are one of the fundamental data sets required for understanding both the natural and anthropogenic causes and drivers of seagrass dynamics.

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Remote Sensing of Wetland Types: Semiarid Wetlands of Southern Hemisphere

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Richard Lucas and Tony Milne

Contents

Abstract

A number of substantive semiarid wetlands occur, with notable examples being in southern Africa and Australia. The seasonal nature of these wetlands requires regular observations with both optical and radar data providing different but often complementary information. Focusing on the Okavango Delta in Botswana, the Sudd Wetlands in South Sudan, and the Murray Darling Basin in Australia, the role of these different sensor types has been used to classify wetlands and understand dynamics and pressures. A wide range of descriptors of wetlands ares also provided by integrating data of differing spatial resolutions (e.g., RapidEye to MODIS) and temporal frequencies (daily to seasonal).

Keywords

Semiarid wetlands · Optical remote sensing · Radar remote sensing · RapidEye · MODIS

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Introduction

Expansive semiarid wetlands include those within the Okavango Delta in southern Africa, the Nile Basin of northern Africa, and those within the Murray Darling Basin in Australia, although smaller wetlands are commonplace. Many of the wetlands in these regions tend to be periodic rather than seasonal and may depend on rainfall or inflow from catchments that extend into other climatic zones. Water flows are generally slow moving and many are described as endorheic (i.e., they do not flow out to external water bodies (rivers or oceans) and evaporate or internally drain into lakes or swamps).

The Okavango Delta, Botswana

The Okavango is one of the largest wetlands in the semiarid regions of Africa, with between 4,000 and 13,000 km^2 inundated annually through incoming water flow and local precipitation. The area consists of a network of river channels and wetlands with the natural vegetation including grasslands and woodlands. The complexity of the landscape in terms of landscape features and vegetation compromises classification from remote sensing data although several elements (e.g., open water) are relatively distinct. McCarthy et al. (2005) nevertheless distinguished up to 12 ecoregions (Fig. 1) using a supervised maximum likelihood and unsupervised classification of Landsat Thematic Mapper (TM) data (from 1989 to 1994), a postclassification update based on multitemporal low resolution images of flooding (1972–2000 derived from NOAA Advanced Very High Resolution Radiometer (AVHRR), ERS-Along Track Scanning Radiometer (ATSR) and reduced resolution Landsat sensor data) and a contextual postclassification procedure (e.g., based on distance from digitized rivers and area/ perimeter ratios and also focal features). The flooding frequency was divided into five classes of inundation namely permanent swamps (>80% of the time), seasonal swamps (30–80%), regularly flooded areas (10–30%), sparsely flooded areas (5–10%), and areas flooded in singular events. Separate classifications were performed for three distinct subregions and then merged. Overall classification accuracies were least when all 12 ecoregions were classified but were increased (to 74%) when these were merged into six classes. An assessment of local rainfall impacts on soil moisture and vegetation response was also undertaken for the Okavango Delta by Wagner et al. (2005), who showed that changes in both were manifested within ENVISAT ASAR Global Monitoring datasets by a transition from multiple scattering to primarily specular reflection. Such changes might also be expected within the ScanSAR modes of the Sentinel-1 C-band SAR mission.

The Sudd Wetlands, South Sudan

The Sudd Wetlands in South Sudan are expansive with a large seasonal flood pulse that determines much of the hydrological, geomorphological, and ecological processes that occur (Rebelo et al. 2012). Habitats occurring include open water and

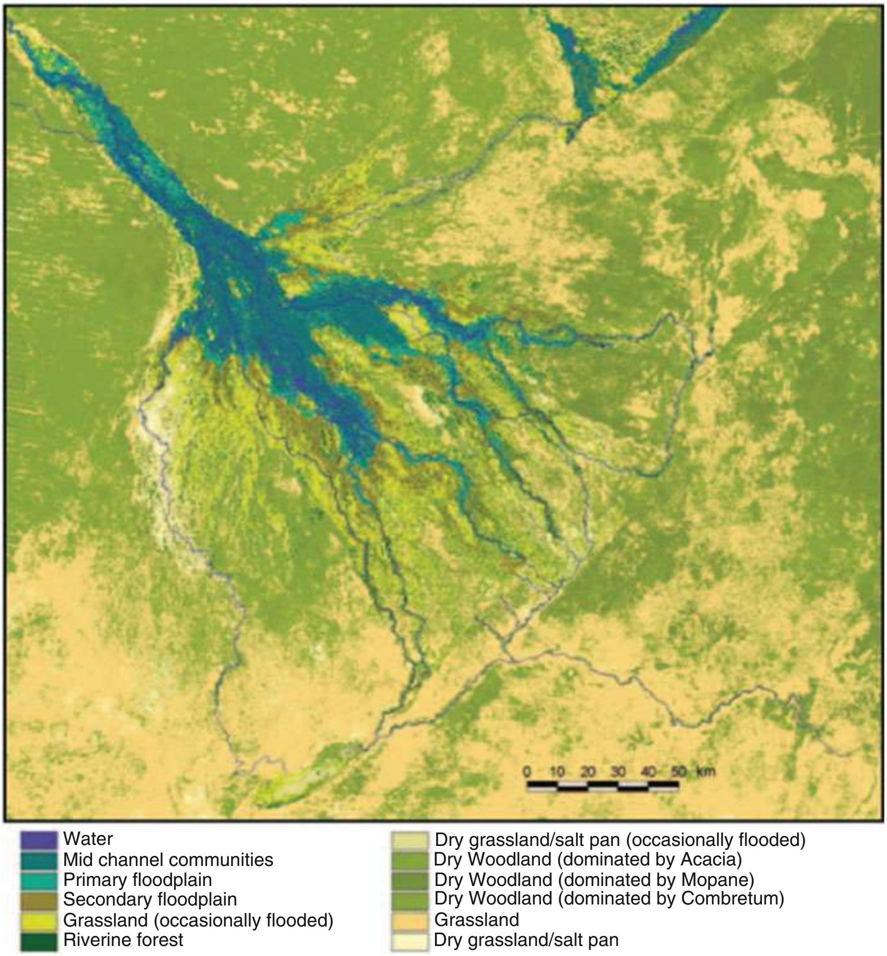

Fig. 1 Ecoregions of the Okavango Delta, Botswana from a combined statistical and contextual rule-based post classification (McCarthy et al. 2005)

submerged vegetation, floating fringe vegetation, seasonally flood and rainfed grasslands, and floodplain woodlands. For this region, time-series of ALOS PALSAR fine beam data were thresholded to determine whether changes in the distribution of open water and inundation vegetation occurred between June 2007 and May 2008. In September, the lowest areas of open water (4,313 km^2) and highest areas of flooded vegetation (27,186 km^2) were observed. In December and January, the area of open water was greater (~10,000 km^2) whilst the area of flooded vegetation was lower (~19,000–22,000 km^2). The total area of wetland on a monthly basis ranged from 22,892 km^2 in June to 32,701 km^2 in January. The total area of wetland (open water or flooded vegetation) over the 5 months of observation was 50,510 km^2 (defined as areas that were either open water of flooded vegetation) with this attributed to both rain and river-fed flooding. The study highlighted the benefits of using SAR data for mapping the extent of both open water and inundated vegetation and understanding flooding regimes. Subsequent studies indicated differences in ALOS PALSAR backscatter between two core aquatic systems dominated by Typha and Papyrus species, with this relating to variations in inundation extent.

The Murray Darling Basin, Australia

The Murray-Darling Basin extends across the Australian state of Queensland through to Victoria and covers an area of over 1.3 million km^2 (Overton et al. 2009). Within this catchment, there are over 200 seasonal and permanent wetland sites; 35 are considered to be significant whilst 10 are listed under the Ramsar Convention. Much of the rainfall that feeds into the Murray-Darling (and other rivers of central Australia) has fallen in the tropical north and the mountains of the east coast. Indeed, water within this catchment takes 3–4 months to flow from the source. In some parts, the water flows into clay pans and remains within these for variable periods of time with evaporation leading to drying up of these waterbodies over time. Much of the vegetation associated with these areas is riparian, with a mix of woody and herbaceous lifeforms. Within this catchment, the primary requirement is for regular and updated information on the extent and duration of inundation, recession patterns, and the changing extent of vegetation and its response to natural and human-induced environmental flows. Such patterns of water inundation are best observed using wide swath data such as those provided by the NOAA AVHRR, MODIS, and Synthetic Aperture Radar (SAR) operating in ScanSAR mode. The benefit of using MODIS data for this purpose is that a wide area of coverage is obtained (Fig. 2), daily observations are provided and both the distribution of open water and vegetation indices (e.g., the Normalized Difference Vegetation Index) and the Enhanced Vegetation Index or EVI) can be calculated; hence, the periodicity of water and the associated response of vegetation can be determined. A benefit of using these data is that the dynamics of the wider catchment can be captured. In saltpan environments, subsurface flows beneath the encrusted cover can also be detected using SAR data. This occurs because the salt is dry at the top, which is penetrated by microwaves allowing detection of the wetter subsurface material. For classifying saltpans and mudflats, Landsat sensor data have often been used (Westbrooke and Miller 1995).

Time Series Data and Combined Techniques

For detailed mapping of wetland areas, the use of time-series datasets and combinations of fine beam dual SAR and optical data is often advantageous. As an example, Milne et al. (2002) used time-series of Radarsat ScanSAR data to track changes in inundated vegetation in Kakadu National Park in Australia's Northern Territory with these used to identify areas of open water, inundated forests, and

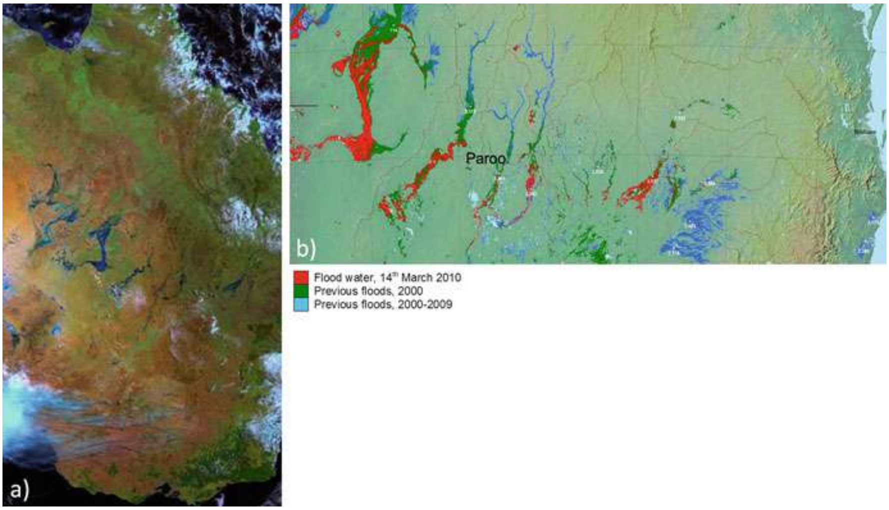

Fig. 2 (**a**) MODIS image of flooding in central Australia on 14th March, 2010. (**b**) Derived maps of the flooding including the Paroo catchment

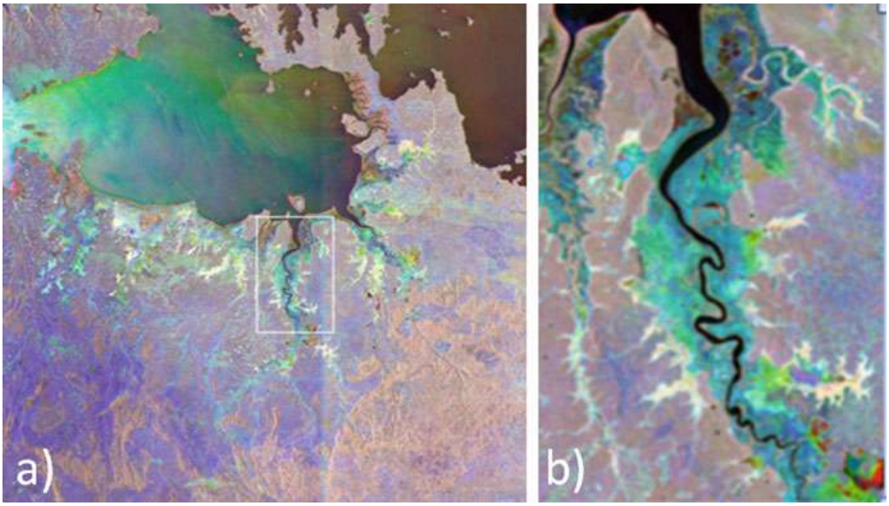

Fig. 3 A color composite of Radarsat data (February (wet), May (early dry), and September (late dry) acquired over Kakadu National Park in Australia's Northern Territory and showing the patterns of flooding and recession. Inundation under vegetation on all three dates displays a *white* coloration (Milne et al. 2008)

macrophytes and track the recession of waters (Fig. 3). ALOS PALSAR and Landsat sensor data have proved beneficial for mapping flooded and nonflooded areas, including swamps, forests, and open water at the Lake Perry in New South Wales, Australia. Further options considered included the use of coherence and

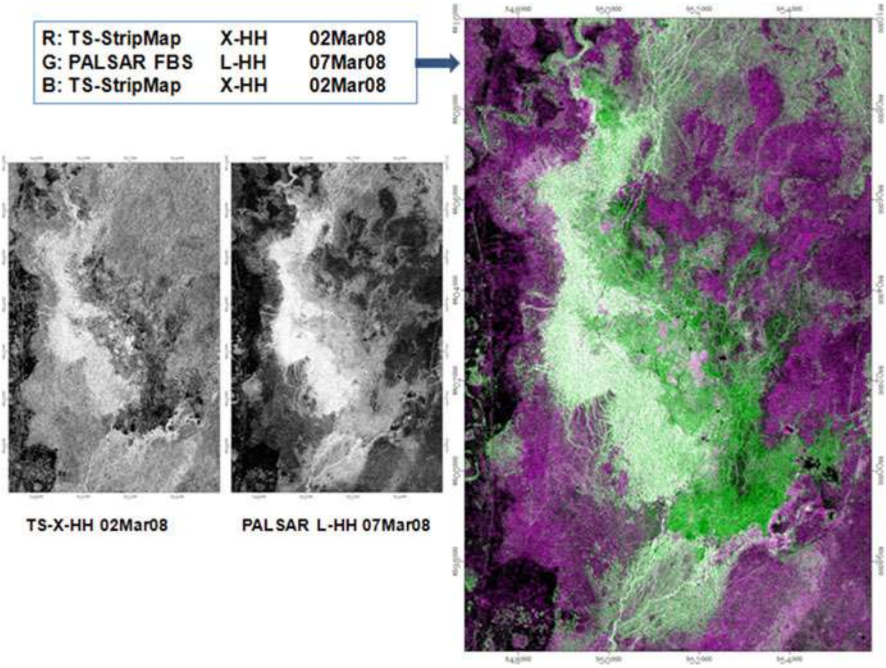

Fig. 4 Different inundation patterns in the Macquarie Marshes reflected in the composite of X-band HH, L-band HH, and X-band HV

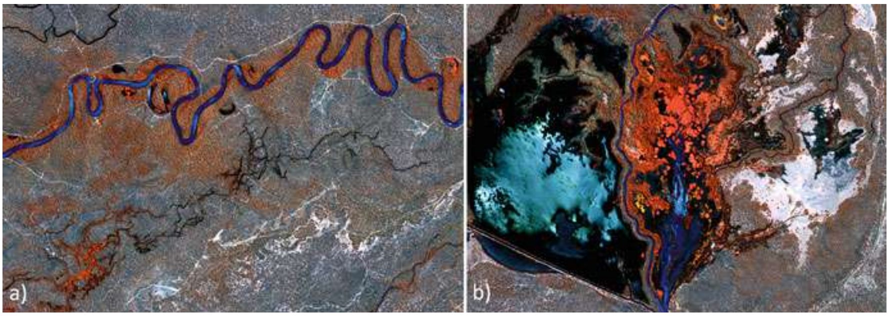

Fig. 5 Open water and inundated vegetation observed using a RapidEye scene of Barmah-Millewa, New South Wales, Australia (© Planet 2017)

combinations of C-band, L-band, and Landsat sensor data. Milne et al. (2008) also used a combination of ALOS PALSAR L-band SAR and TerraSAR-X to discriminate between different wetland surfaces on the Macquarie Marshes in north central New South Wales (Fig. 4). The low shrubs and grasses of the floodplain were associated with volume scattering at X-band and ponded water with aquatic vegetation leading to a strong double bounce interaction. Dark areas at X-band indicated whether the water had overtopped the vegetation in the wetlands, with surface scattering then dominating. The higher returns at L-band were associated with woody vegetation and particularly where this was inundated. More detailed information on wetlands can also be obtained using the higher resolution RapidEye data, including the extent of inundation and detailed mapping of different vegetation communities (Fig. 5).

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Remote Sensing of Wetland Types: Subtropical Wetlands of Southern Hemisphere

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Abstract

The Pantanal is a large cross-border wetland system in South America that supports a diversity of ecosystems. This internationally important wetland is under threat from water diversion but also agricultural development and there is a need for satellite-based observations to characterize, map, and monitor the changing extents and states of the wetlands. To achieve this, a number of connected studies have been undertaken using combination of Synthetic Aperture Radar (SAR) operating at C- and L-band. Different lake types were also classified using an object-oriented approach, with these linked to different levels of salinity, primarily through association with vegetation types.

Keywords

Wetlands · Synthetic Aperture Radar · Lake types · Vegetation

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Introduction

The Pantanal of South America, located in the center of South America, between Brazil, Bolivia, and Paraguay, represents one of the largest wetlands in the southern hemisphere subtropics. Estimates suggest that the inundated area of the Pantanal covers approximately 160,000 km^2 during maximum flooding, and with the entire Pantanal watershed occupying an area of approximately 362,000 km^2 (Junk et al. 2006). The upper Paraguay River and its three tributaries feed the Pantanal wetlands, promoting a strong annual unimodal flood that varies in duration, amplitude, and extent both yearly and spatially (Hamilton et al. 1996). Such characteristic flood dynamics require morphological, anatomical, physiological, and/or ethological adaptations from the local biota. As these seasonal water level variations are the driving force of ecological processes in floodplain systems, identifying the various permanent and semipermanent terrestrial and aquatic habitats in a seasonal flood-pulse ecosystem is critical for understanding the biogeochemical, hydrological, and ecological processes of the ecosystem (Hamilton et al. 1996). The dynamics of inundation in the Pantanal also promote a high diversity of vegetation (Alho 2008), expressed by a unique landscape characterized by different compositions of savanna vegetation, abundant species of aquatic vegetation, and different types of floodplain forests (Alho 2008). In addition to the floristic diversity, a large number of hydrochemically varied lakes, waterways, and other fluvial geomorphological patterns are observed, generating a complex mosaic of wetland habitats (Por 1995; Costa and Telmer 2006; Evans and Costa 2013; Evans et al. 2014).

The vast habitat diversity in the Pantanal is poorly understood, and currently threatened by human development occurring both in the floodplain and on the surrounding plateau. These developments threaten to alter the Pantanal ecosystem in a potentially irreversible manner, mostly through the modification of the natural hydrological cycles of the rivers, and the destruction of natural habitat (Alho 2008). Despite the ecological importance of the Pantanal, and potential consequences resulting from habitat alteration/loss, there is a lack of understanding about the spatiotemporal variability of the landscape units within this wetland ecosystem (Costa 2004). As such, quantification and monitoring of the landscape changes in the Pantanal are critically needed, so that sustainable management practices and effective conservation units can be established. Yet, given the size and relative inaccessibility of the Pantanal system, conventional methods of data gathering are difficult and expensive. Thus, remote sensing technology offers a cost-effective alternative for mapping the spatial variability of the landscape units within this highly heterogeneous wetland ecosystem. In this region, where capturing cloud-free imagery for a large region is difficult, synthetic aperture radar (SAR) imagery is the most viable alternative.

Mapping Wetland Landscape

Costa and Telmer (2006) utilized a combination of C-band (RADARSAT-1) and L-band (JERS-1) imagery to classify the geochemically varied lakes in the Nhecolândia region of the Pantanal, based on the specific types of aquatic vegetation associated with each geochemical condition. Evans et al. (2010) combined temporal coarse resolution SAR (ALOS/PALSAR ScanSAR) and RADARSAT-2 imagery, and more recently Evans et al. (2014) combined 50 m resolution SAR (ALOS/PALSAR, RADARSAT-2, and ENVISAT/ASAR imagery) to map the land cover for the entire Pantanal. Evans and Costa (2013) combine SAR data at improved spatial resolution (12.5 m) and produced a detailed landscape map of Nhecolândia, a sub region of the Pantanal.

SAR Medium Resolution

The first detailed landcover map for the entire Pantanal was generated by Evans et al. (2014) using these SAR imagery acquired over the same seasonal cycle (Fig. 1). The high frequency of SAR data acquisition was only achieved because of the systematic observation strategy associated with the ALOS PALSAR data. The classification uses 50 m spatial resolution, dual-season, HH and HV L-band ALOS/PALSAR, and HH and HV C-band RADARSAT-2 data, as well as a comprehensive set of ground reference points, to map nine classes of the hydrologically variant subregions of the Pantanal by using a hierarchical object-based image analysis approach. This classification was achieved with an overall accuracy of 80% for the entire Pantanal. The relative coarseness of the image spatial resolution and the heterogeneity of landscapes within the region did, however, limit the features that could be resolved and also the accuracy of the classification. The produced habitat spatial distribution maps provide vital information for determining refuge zones for terrestrial species and connectivity of aquatic habitats during the dry season, as well as providing crucial baseline data to aid in monitoring changes in the region, and to help define conservation strategies for habitat in this wetland.

SAR Fine Resolution

The first detailed landcover map for Nhecolândia, a subregion of the Pantanal, was generated by Evans and Costa (2013) using SAR imagery acquired over the same seasonal cycle and an object-oriented classification approach. L-band images from ALOS/PALSAR were acquired for January/February 2008 (12.5 m, HH polarization) coinciding with high water, and for August/September 2008 (12.5 m, HH and HV polarization) coinciding with low water. RADARSAT-2 images were acquired for August 2008 (25 m, HH and HV polarization) coinciding with low water and a field campaign. Additional C-band imagery for high water were acquired in February/March 2010 from ENVISAT/ASAR (12.5 m, HH and HV polarization). The Level 1 classification defined four landscape units with an overall accuracy of 90%. Defined habitats were as follows: (1) forest savanna, which includes deciduous and semideciduous forest, cerradão, cordilheiras, and capãos, and mixed vegetation with shrubs and short scattered trees on a grassy stratum; (2) open grass cover, which includes dominantly grassy/agriculture terrain with some scattered shrubs;

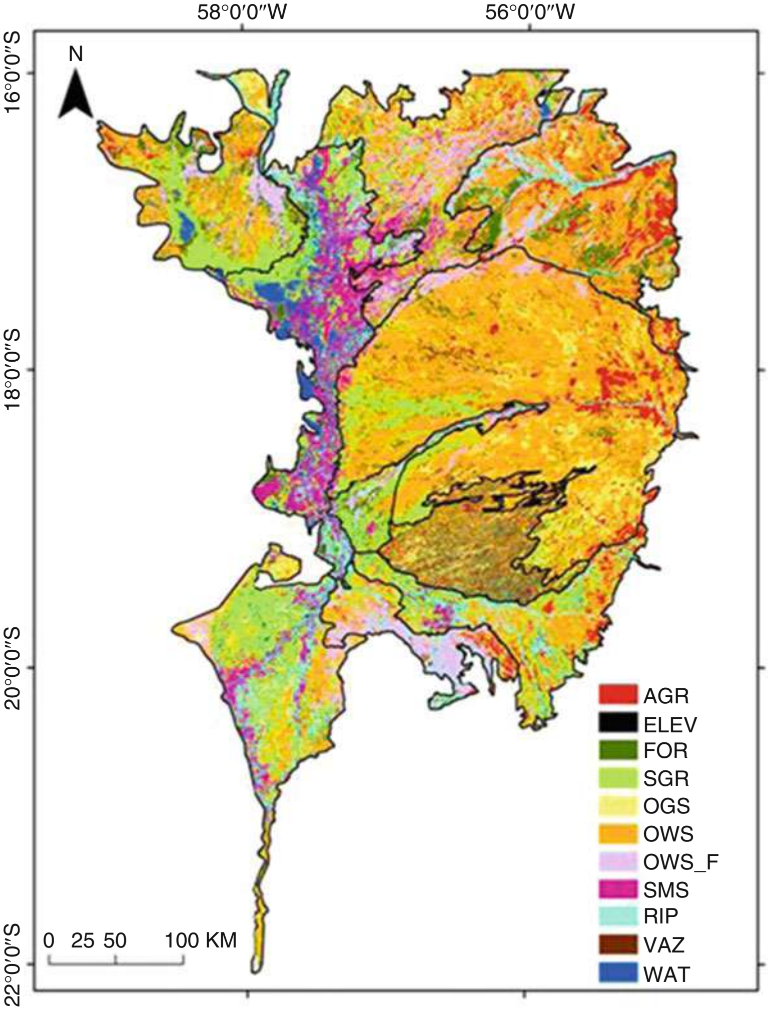

Fig. 1 Classification of land covers within the Pantanal generated using multitemporal ALOS PALSAR and RADARSAT-2 imagery. (*FOR*) forest/woodland, (*RIP*) riparian forest, (*OWS*) open wood savanna, (*OWS_F*) open wood savanna subject to prolonged flooding, (*OGS*) open grass savanna, (*AGR*) agriculture, (*SGR*) swampy grassland, (*SMS*) swampy mixed savanna, (*VAZ*) *Vazantes*, and (*WAT*) water (Evans et al. 2014)

(3) *vazantes*, defined as seasonal drainage channels; and (4) lakes, which includes all types (Fig. 2).

Within this region, numerous small lakes exist and these were classified by Costa and Telmer (2006) using a combination of high resolution RADARSAT-1 and JERS-1 SAR data and refined subsequently by Evans and Costa (2013) The general method to differentiate the lakes relies on the natural differences among these lakes: brackish lakes (locally known as *salinas*) are devoid of any emergent aquatic

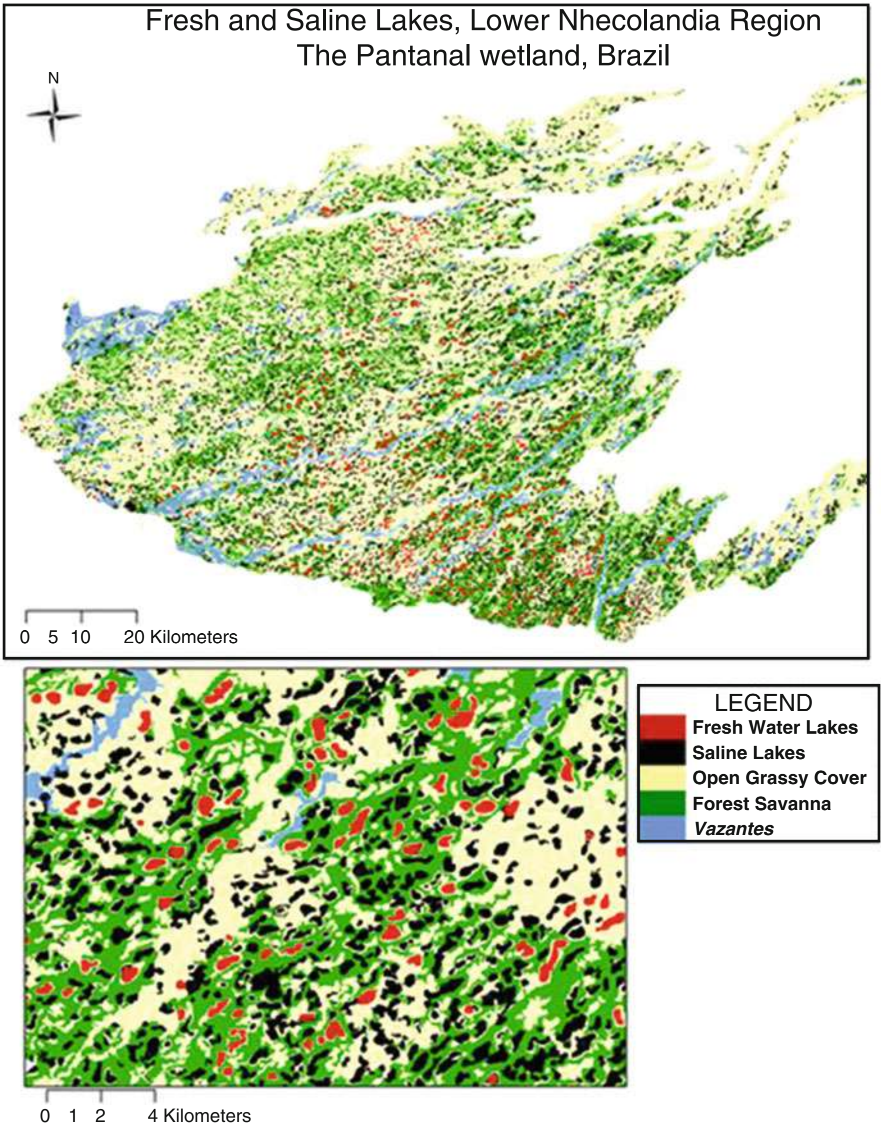

Fig. 2 Landscape classification (Modified from Evans and Costa 2013)

vegetation; fresh water lakes (locally known as *baías*) are populated by a variety of floating/emergent aquatic vegetation ranging in height from 2–100 cm (including *Cyperaceae*, *Pontederiaceae*, *Araceae*, *Salviniaceae*, and *Nymphaeaceae)* and *salobras* can also be colonized by floating and emergent aquatic vegetation, but are distinguished because of the large stands of *Typhaceae*, typically 200–300 cm in height. A relationship has been established between lake water geochemistry and associated vegetation (and therefore, radar backscattering signal), allowing for the classification of lake geochemistry using radar imagery (Costa and Telmer 2006; Evans and Costa 2013). A Level 2 classification used only the “lakes” class from Level 1 and divided it into fresh lakes (*baías*) and saline lakes (*salinas*) with an accuracy of 98% (Fig. 2).

As defined in Evans and Costa (2013), the lakes were classified with a combination of imagery from three SAR satellites and an object oriented classification approach. First, a multiresolution segmentation was performed on the imagery and training objects for the two lake types were defined based on field data and the segmented output. Next, radar backscattering mean and standard deviation values for these training objects were examined to determine separability between the lake types using the individual radar images. Finally, a combination of defined backscattering thresholds (class mean +/− 1SD), user-defined rules, and the utilization of the Feature Space Optimization (FSO) tool (in eCogniton) with a supervised nearest neighbor classification algorithm allowed for the classification of the two Pantanal lake types. The lakes classification map (Fig. 2) shows the broad distribution of fresh and brackish lakes in the region.

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Remote Sensing of Wetland Types: Temperate Bogs, Mires, and Fens

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Richard Lucas

Contents

Abstract

In temperate regions, many bogs, mires, and fens have been subject to anthropogenic disturbance and hence are relatively small in extent and often fragmented. Two examples provided are the active raised upland and lowland bogs of Cors Caron and Cors Fochno in Wales, where the complexity of wetland vegetation and the surrounding landscape necessitates the use of Very High Resolution (VHR) sensor data, such as that provided by the Worldview Sensor. A range of spectral indices can be derived that indicate relative amounts of vegetation with different productivities, dead material, and water, which can also be used to infer vegetation distributions. An approach to classifying temperate wetlands has involved the use of a rule-based classification based that uses these indices as well as other spectral information and the Food and Agricultural Organization's (FAO) Land Cover Classification System (LCCS).

Keywords

Bogs · Mires · Fens · Very high resolution sensor data · Worldview sensor · Rule-based classification

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Introduction

Across much of the temperate zone, bogs, mires, and fens are relatively limited in their extent because of anthropogenic conversion of the habitat itself and/or the surrounding landscape. This is particularly the case in northern Europe where agricultural intensification and urban expansion have reduced the area of wetlands, with many existing as isolated fragments. As a consequence, many remote sensing techniques have focused on using imagery from very high resolution (VHR) sensors.

Example: Cors Caron and Cors Fochno Bogs, Wales

The active raised bogs of Cors Caron (upland) and Cors Fochno (lowland) in Wales have been subject to human-induced pressures for decades. At Cors Caron, remote sensing observations using the Worldview-2 sensor reveal the demise of the primary domes and the efforts that are being placed to retain the water within and around these. The benefit of using the Worldview-2 data is that eight reflectance bands, including the coastal, yellow, and red edge, are available, allowing the calculation of spectral indices including the Normalised Difference Vegetation Index (NDVI), Plant Senescence Reflectance Index (PSRI), and the Water Band Index (WBI). These indices for Cors Caron reveal the extensive areas of agricultural land surrounding the active raised bog, differentiate between senescent grasslands dominated by purple moor grass (*Molinia caerulea*), and allow areas of open water and moist vegetation to be detected (Fig. 1a–c).

The seasonal differences in the active raised bog area at Cors Fochno are revealed by comparing time series of VHR data, such as that acquired by the RapidEye (Fig. 2). Within the cultivated area as well as the margins surrounding the bog, the

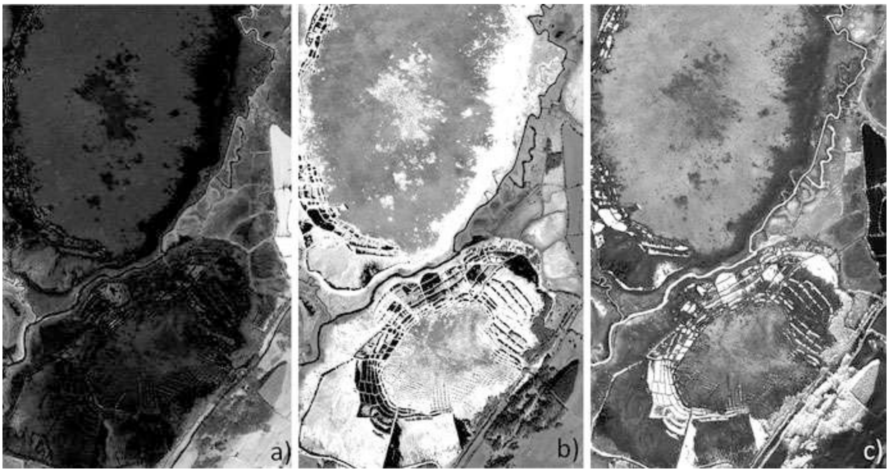

Fig. 1 (**a**) NDVI, (**b**) PSRI, and (**c**) WBIvi derived from Worldview II images of Cors Caron, UK

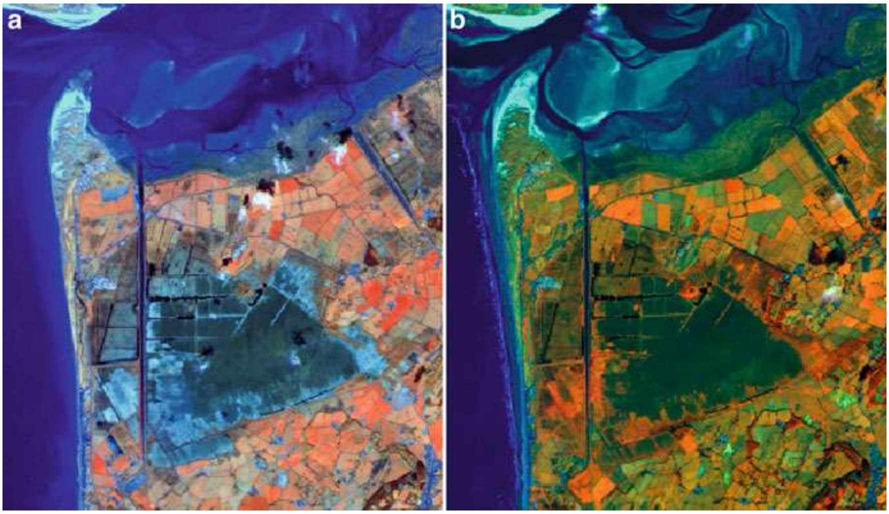

Fig. 2 RapidEye images acquired in (**a**) May and (**b**) July, highlighting the relative stability in the reflectance of the active raised bog but significant changes in the visible and near-infrared reflectance on marginal habitats (e.g., Molinia-dominated grasslands)

vegetation is dominated by *Molinia caerulea*, although *Phragmites australis* is also present but difficult to distinguish. These exhibit a high near-infrared reflectance in the summer months, when they turn highly productive (green) compared to the pre-flush (winter) period. By contrast, the bog surface, which is dominated by woody shrubs (*Myrica gale*, *Calluna vulgaris*, and, to a lesser extent, *Erica tetralix*), grasses (*Eriophorum* species interspersed with *M. caerulea*), and mosses (primarily *Sphagnum* species in the bog pools), remains relatively stable throughout the observation period.

Classification of Raised Bogs

To provide a classification of the active raised bogs and surrounds, Lucas et al. (2014) describe the use of the Food and Agricultural Organisations (FAO) Land Cover Classification Scheme (LCCS) taxonomy in a rule-based approach that utilizes dual season imagery. In this approach, termed the Earth Observation for Dynamic HAbitat Monitoring (EODHAM) system, both a pre- and peak-flush image are required. Initially, spectral rules are applied to discriminate between vegetated and non-vegetated environments that are aquatic from those that are terrestrial. Further differentiation is then made between landscapes and habitats which are cultivated, managed, or artificial or natural or seminatural. To achieve this, reference is made to existing thematic layers (e.g., cadastral information or urban layers) or to large objects formed by segmenting the imagery, with these classified as cultivated, managed, or artificial based on geometry, dimensions, and context. Through cross tabulation, classification to LCCS Level 3 is achieved, which includes the classes of cultivated and seminatural/natural aquatic. To classify beyond Level 3, separate layers are generated which, for aquatic surfaces, represent state (water, ice, or snow), persistence, duration, dynamics, depth, and sediment loads with separate codes assigned to each layer. These sub-codes are then combined to form the full LCCS code with an associated description. Subsequently, these land cover codes are translated to General Habitat Categories

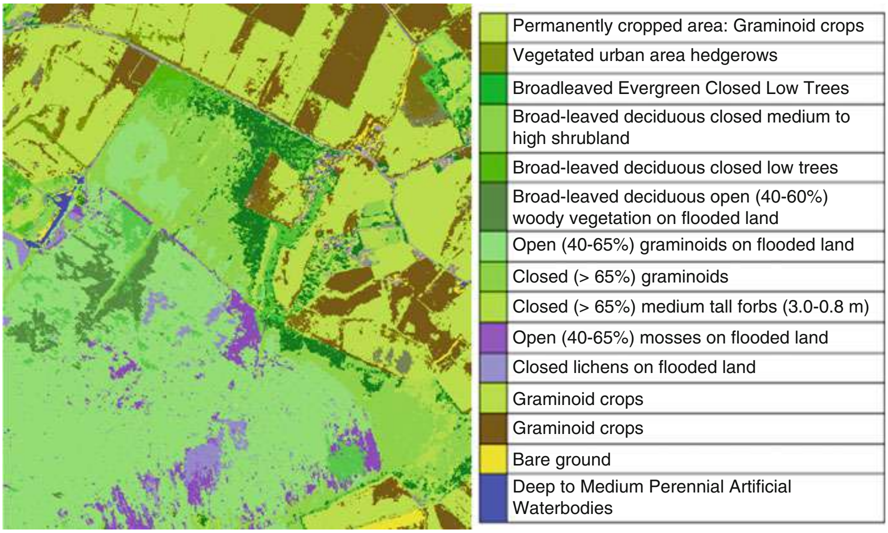

Fig. 3 Classification of wetland habitats, the Kalamas National Park, Greece

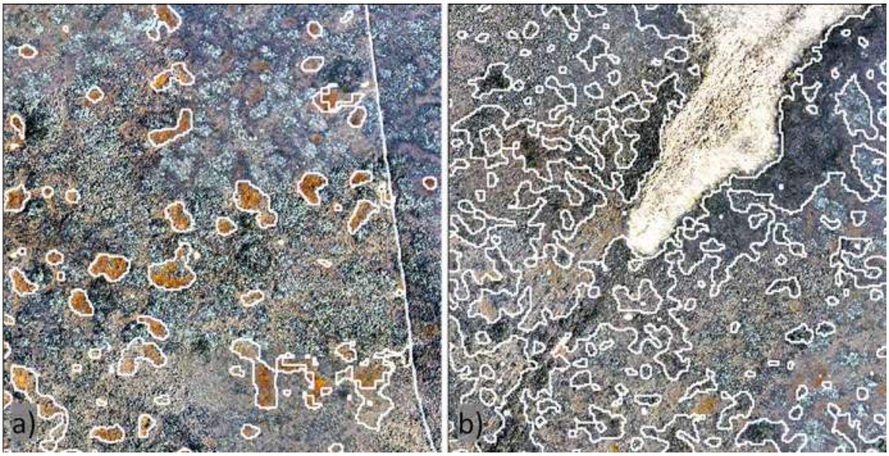

Fig. 4 (**a**) Bog pools containing Sphagnum species and (**b**) vegetated areas dominated by *Calluna vulgaris*, as observed within UAV imagery

(GHCs) and to Annex I habitat descriptions. The accuracy of the classification focuses on each of the layers separately, with the final overall classification being a combination of the errors associated with each layer. Change detection is based on a change in LCCS sub-code or in the spectral indices considered. An example of the classification is presented in Fig. 3.

Other types of remote sensing data can also provide an indication of the nature of the active bog surface. For example, airborne LIDAR at Cors Fochno reveals the location of the pool and hummock sequences indicative of an active bog and also different management regimes, which can then be used to differentiate modified raised bog. Unmanned airborne vehicle (UAV) imagery (Fig. 4) can also provide very detailed mapping of bog surfaces, with bog pools containing *Sphagnum* species being clearly separable from the surrounding vegetation as are areas dominated by *Calluna vulgaris*.

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Remote Sensing of Wetland Types: Tropical Flooded Forests

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Bruce Chapman, and Tony Milne

Contents

Abstract

In many tropical regions, wetlands are extensive and are often associated with forested environments. Notable examples include those occurring in the floodplains of the Amazon and Congo Rivers. In these regions, cloud cover is

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prevalent and wetland mapping approaches have been most successful when using time series of Synthetic Aperture Radar (SAR) data. The description of wetlands in forested environments is particularly enhanced when using temporal L-band HH datasets. The provision of information on these tropical wetlands is leading to a better understanding of their contributions to regional and global climate, particularly given their important role as a repository for carbon.

Keywords
Tropical wetlands · Inundation · Synthetic Aperture Radar (SAR) · Carbon

Introduction

Within tropical regions, many of the wetlands are associated with floodplains with the rivers often flowing through densely forested landscapes, as in the case of the Amazon and Congo Rivers and their tributaries. Characterizing these flood plain environments is important to ensure that the diverse habitats that occur and the forest, game, and fisheries resources that have historically supported local populations are used sustainably. In Brazilian Amazonia, for example, large areas of the flood plain environment (commonly referred to as várzea) have been

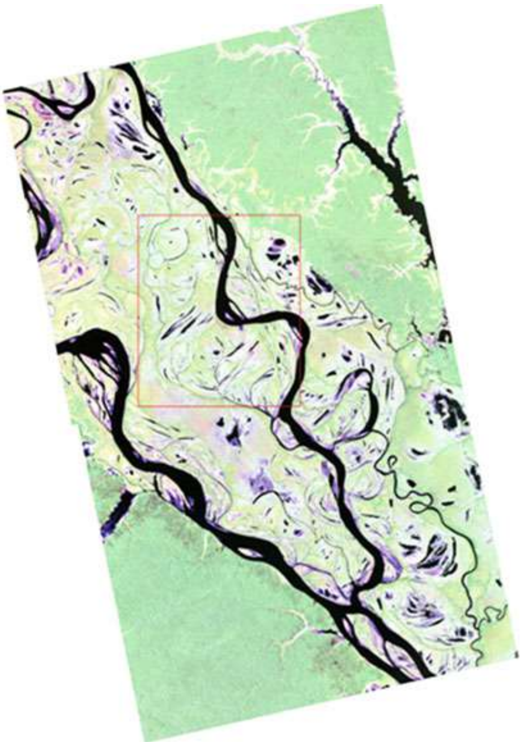
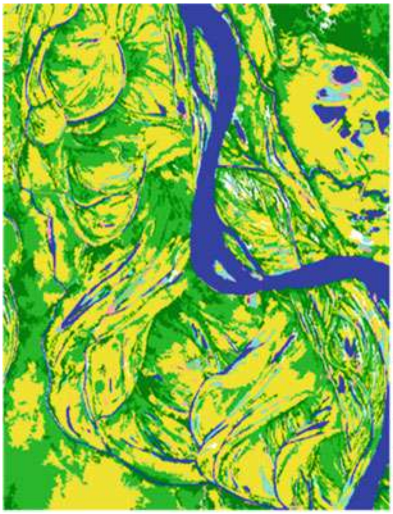


Fig. 1 ALOS PALSAR Fine Beam multitemporal composite, Mamirauá Sustainable Development Reserve, Amazon floodplain (2.8S, 65.0W). *Red*: HH, 14 June 2007; *Green*: HV, 14 June 2007; *Blue*: HH, 30 July 2007. The Mamirauá Reserve is bounded by the Amazon River to the southwest and the Japurá River to the northeast. The red box corresponds to area shown in Fig. 2 (© JAXA METI)

designated as Sustainable Development Reserves. Within these, there is a need to assess and monitor biodiversity and ensure conservation planning is carried out in cooperation with local communities.

Detecting Seasonal Inundation

Within these floodplain environments, the distribution and patterns of vegetation are linked closely to the periods of inundation. Given the prevalence of cloud cover in the tropics, lower frequency (e.g., L-band) SAR data are optimal for detecting seasonal inundation beneath the forest canopies with sensors such as the JERS-1 SAR, ALOS PALSAR, and ALOS-2 PALSAR-2 being particularly relevant. As an example, Fig. 1 illustrates how the diverse environments of the Mamirauá Sustainable Development Reserve, located at the confluence of the Amazon and Japurá Rivers, manifest themselves within an ALOS PALSAR Fine Beam multitemporal composite of L-band HH and HV data. From this image, an object-based classification has been applied allowing discrimination of the primary vegetation and water categories (Fig. 2). Such information provides useful inventory for managing the forests and fisheries but also for planning the conservation of animal species including the giant Amazonian pirarucu fish (*Arapaima gigas*), Amazonian manatee (*Trichechus inunguis*), jaguar (*Panthera onca*), and three caiman (*Melanosuchus*) species.



Structure	Inundation period		Local term
Woody - tall	0-1 months/year		High várzea forest
Woody - tall	1-2 months/year		High várzea forest
Woody - tall	2-6 months/year		Low várzea forest
Woody - tall	6-9 months/year		Chavascal
Woody - short	6-9 months/year		Chavascal
Herbaceous	9-12 months/year		Aquatic macrophyte
Non-vegetated	9-12 months/year		Open water

Fig. 2 Classified image for subset (*red box*) of Fig. 1, using three PALSAR dates as inputs (14 June, 30 July, and 30 October 2007). Between high-water stage in June, and low-water stage in October, the Amazon River fell 9.1 m. In June, nearly the entire floodplain was inundated. Várzea tree species are adapted to tolerate various degrees of flooding – on the lower parts of the floodplain, trees are largely submerged at peak flood stage

The use of time-series of SAR data generally facilitates better classification of inundated wetlands compared to the use of single-date polarimetric or multi-incidence angle SAR data (Furtado et al. 2013). As an example, Silva et al. (2010) used 14 single polarized C-band HH Radarsat-1 images acquired within a single hydrological year to separate woody from herbaceous cover in the lower Amazon floodplain with the distinct temporal responses of the different macrophyte communities allowing discrimination and classification using an object-based approach (with 70–90% accuracies obtained). Even so, confusion occurred between sparse woody vegetation and dense macrophytes although the inclusion of MODIS red and infrared bands reduced confusion between short herbaceous vegetation and bare soil, particularly during periods of low water (Fig. 3).

Using JERS-1 SAR data acquired as part of the Global Rain Forest Mapping (GFRM) project, Rosenqvist et al. (2002) mapped different areas of open water and inundated forest in the Jau River, a tributary of the Rio Negro with the latter exhibiting a distinct double bounce response and high HH backscatter. Mayaux

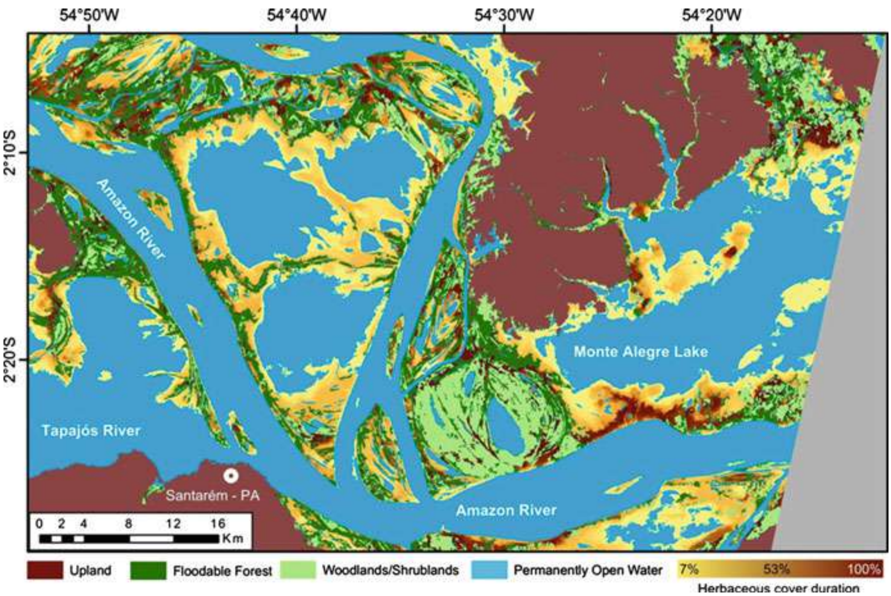


Fig. 3 Land cover classification of the Monte Alegre Lake floodplain region, in the Lower Amazon, based on multitemporal classification of Radarsat 1 C-HH SAR data. A total of 14 scenes acquired during the annual hydrological cycle were used to first derive broad Level 1 classes (upland, floodplain, and permanently open water, and then further separate the spatially static vegetation classes within the floodplain (floodable forest and woodlands/shrublands). The remaining areas were labelled as possible macrophytes, and actual macrophyte presence was mapped individually on each of the 14 scenes. Macrophyte occurrence was then summarized by computing the duration of macrophyte cover along the hydrological year, based on the percentage of the 14 scenes where it was observed

et al. (2002) was similarly able to detect flooded forest in Central Africa using a combination of ERS SAR and JERS-1 SAR data.

The opportunities for mapping seasonal inundation patterns are increased with the use of wide-swath ScanSAR data, as it permits more frequent imaging of inundation. Arnesen et al. (2013) show that flooded forests of the Curuai lake floodplain on the Lower Amazon can be detected with an accuracy better than 80%.

Future Challenges

The ability to accurately monitor changes in inundation extent (e.g., from temporal SAR) supports the study of possible climatological tipping points in regions where extensive inundation is prevalent (Lenton et al. 2008). For example, the floodplains of South America's Amazon River span hundreds of thousands of square kilometers (Hess et al. 2003). Each year the run-off from heavy seasonal rains causes extensive and prolonged flooding of vegetated environments. These seasonally, semipermanently, or permanently inundated areas form biologically diverse habitats but also regulate biogeochemical processes such as the generation of methane and the outgassing of CO_2 (Devol et al. 1990). If annual rainfall rates were to decrease substantially in the next decades, as predicted by some global circulation models (e.g., Huntingford et al. 2008), these would have a substantial effect on greenhouse gas emissions (Bowman et al. 2013).

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Remote Sensing of Wetland Types: Tropical Herbaceous Vegetation

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Tony Milne

Contents

Abstract

The classification of wetland vegetation classes is conditional upon the variety of species or community of species present and seasonal changes in the extent and duration of water and flooding patterns occurring within the landscape. The capability of detecting and mapping water flow under vegetation using synthetic aperture radar (SAR) enables information to be determined about the structure, composition and extent of surface vegetation types.

Fully polarimetric C, L and P –band SAR data was acquired over the Cambodian Tonle Sap Basin in September 2000 and interrogated to map thirteen wetland vegetation ecosystem classes at the western end of the Basin.

In a second example both SAR and optical imagery were used to map wetland vegetation on Lake Chilwa, Malawi.

Keywords

Radar · Acquatic Plants · Classification · River Flow · Discharge Regime

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Introduction

In many tropical wetlands, herbaceous vegetation dominates and may be seasonal in its distribution (e.g., because of different water flows). As an example, in western Cambodia, the flow of water from the Mekong River system into the Tonle Sap Great Lake (TSGL; Fig. 1) during the wet season (July to October) expands the surface area of the lake from 250,000 ha to 1.25 million ha, with this including large areas of adjacent forests and woodlands, macrophytes, and floating grasses. This flow into the TSGL occurs because the distributaries of the lower Mekong downstream from Phnom Penh are unable to cope with the wet season flood discharge flowing into the Gulf of Thailand; the waters back up and flow into the TSGL. With the reversal of water flow back into the Mekong system from November each year, large areas of flooded land on the lakes edge (often with herbaceous vegetation) are successively exposed and used for agriculture. In this fertile zone of migrating waters about 48,000 ha are planted with “receding rice” and another 24,000 ha with other field crops. This water draining back into the Mekong increases downstream flow by an estimated 16% providing much needed water for irrigation in southern Cambodia and Vietnam and reducing the risk of salinity intrusion upstream from the immediate tidal regions.

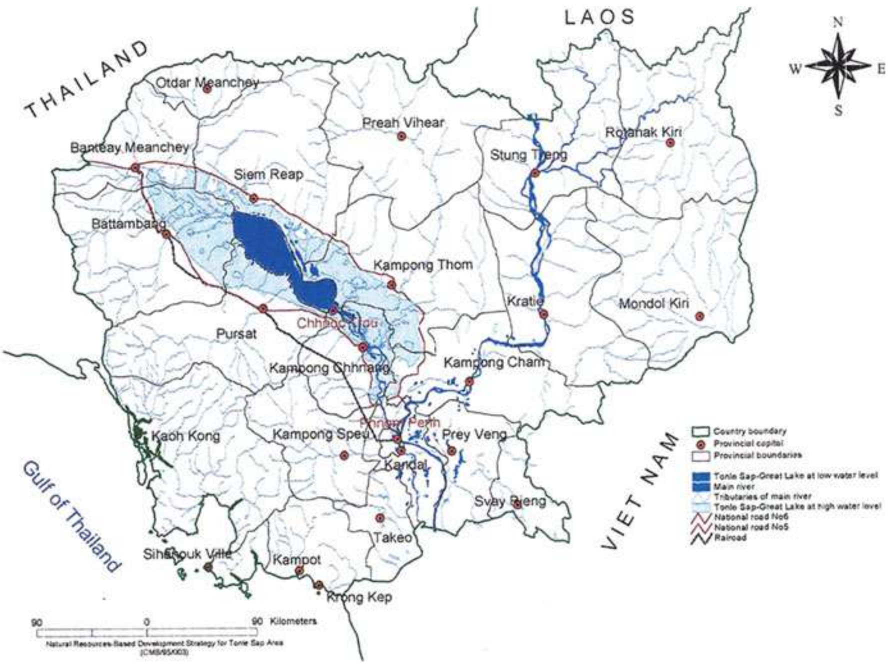

Fig. 1 The location of Lake Tonle Sap, Cambodia

PACRIM-2 Mission

As part of the NASA-Australia sponsor PACRIM-2 mission, airborne Synthetic Aperture Radar (AIRSAR) data were collected over the Tonle Sap and Angkor regions of Cambodia during high water (Fig. 2). The imagery were first segmented

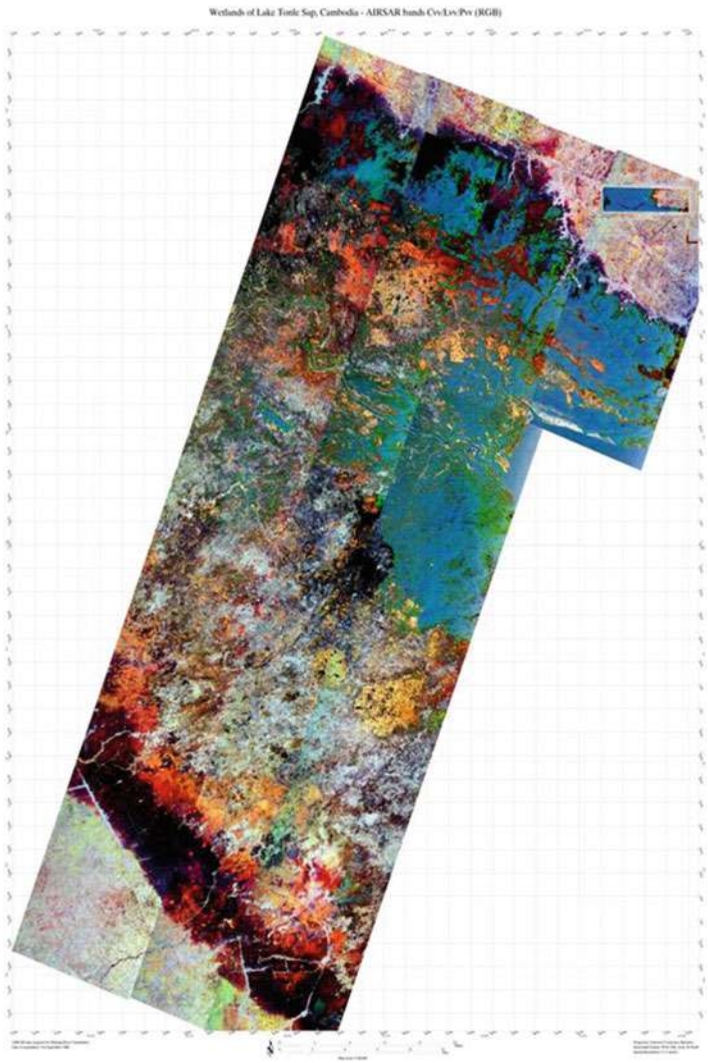

Fig. 2 Mosaic of four AIRSAR flight-lines displayed as an *RGB* colour composite of C-band VV, L-band VV, and P-band VV over the western end of Tonle Sap Great Lake. The data are georeferenced using the WGS 1984 horizontal datum and UTM Zone 48N map projection

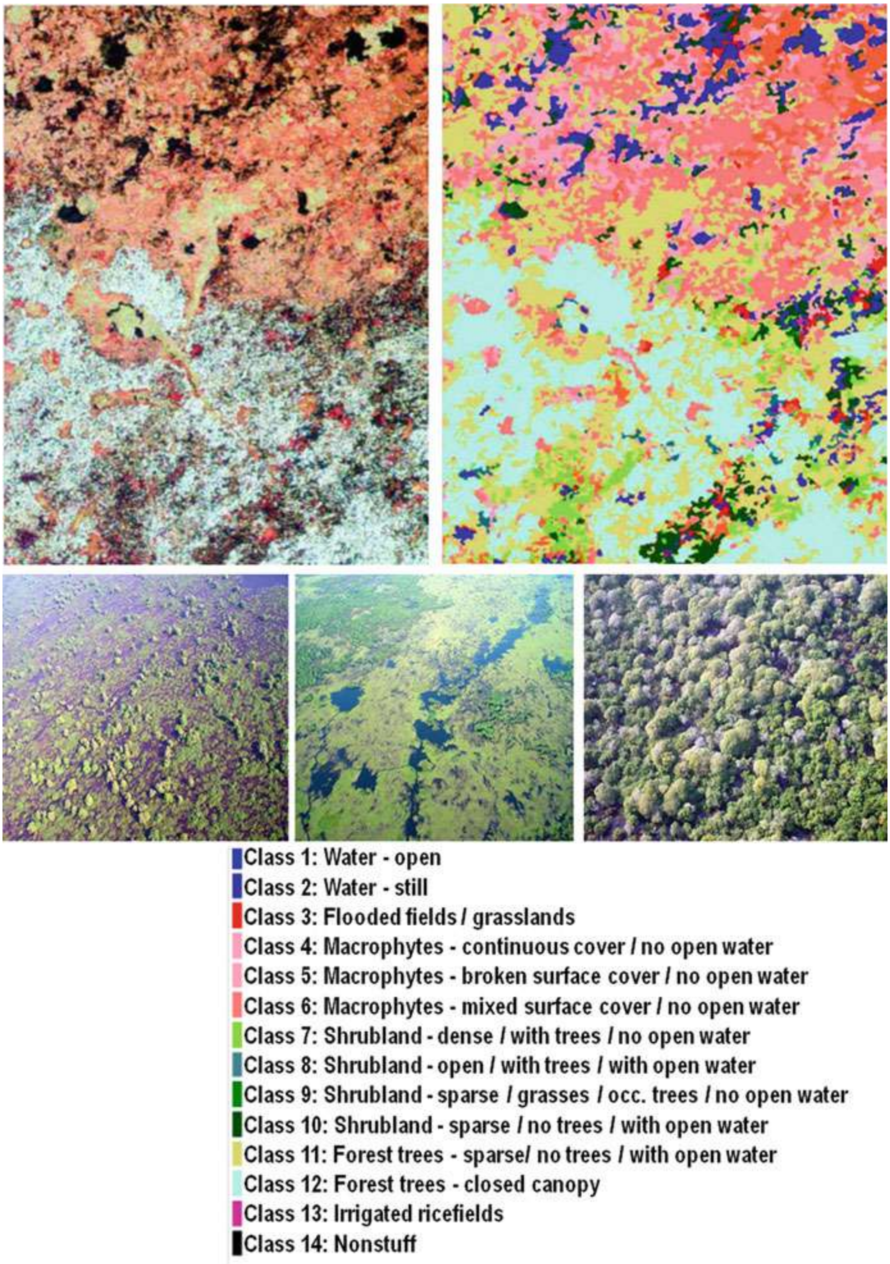

Fig. 3 Classification of wetlands from AIRSAR data

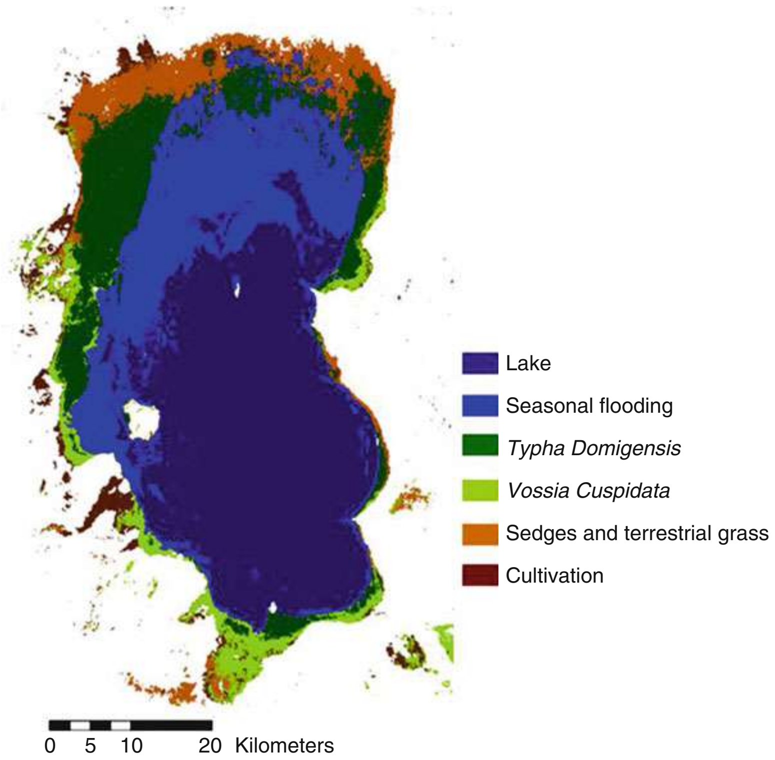

Fig. 4 Classified image of dominant vegetation species, Lake Chilwa, Malawi (© JAXA METI)

using a modified Gaussian Markov Random Filed Model (Dong et al. 2001; Horn and Milne 2002), with each segment associated with attributes such as the mean and standard deviation. An ISODATA classification was then performed on the AIRSAR C-, L-, and P-band imagery to map 13 wetland ecosystems, including macrophytes, floating grasses, shrublands, open and closed forests, mangroves, and open water surfaces (Fig. 3). As well as extent, consideration was given to structural form (e.g., trees, shrubs), growth stage, and condition. The classification scheme was consistent with guidelines listed in the Asian Wetlands Inventory (Wetlands International) for describing wetlands and addressed the needs of the Tonle Sap Biosphere Reserve Act, which requires data and information to assist in formulating management plans for the core, buffer, and transition zones of the Prek Toal Nature Reserve. A limitation of the approach was the fully polarimetric C-, L-, and P-band data were used which are not always available and certainly not from spaceborne platforms. However, further analysis indicated that the three channel combination of C-band HH and L-band VV and HV captured 72.2% of the information (Milne and Tapley 2004).

Both SAR and optical data can be used to map dominant herbaceous wetland vegetation and are often used in combination, as in the case of the Lake Chilwa wetland in Malawi (Fig. 4; Rebelo 2010) and the flooded Várzea forests of the Amazon (Hess et al. 2010). Data from airborne hyperspectral imaging spectrometers has also proved popular as they offer both high spectral and spatial resolution imagery that is ideally suited to monitoring wetland vegetation. Data acquired by hyperspectral sensors such as the Compact Airborne Spectrographic Imager-2 (CASI-2) have also proved useful for mapping the distribution of wetland plant species. For example, Hunter et al. (2010) used these data to map macrophytes in the Upper Thurne region of the Norfolk Broads, UK.

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GlobWetland: ESA Earth Observation Project Series to Support Ramsar Convention

Marc Paganini

Contents

Abstract

Achieving the objectives of the Ramsar Convention requires access to global, up-to-date, and reliable information to complete national wetland inventories and to undertake adequate assessment and monitoring, and establish appropriate management and restoration plans. The use of satellite Earth Observations (EO) with innovative geo-spatial analyses has become a key tool and source of information for such purposes. Remote sensing observations acquired over short- to long-time frames by airborne and more particularly by spaceborne missions are increasingly used to support the implementation of the convention by supporting the efficient management of wetlands through the provision of local to global, up-to-date information. These data also enhance the reporting mechanisms of the convention by facilitating better decision-making through the generation of common datasets and information systems, as well as the harmonization of formats, methods, and procedures for gathering and analyzing information. A number of projects are

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aimed at supporting the Ramsar Convention through the characterization, mapping, and monitoring of wetlands at regional to global levels. Among these are the ESA GlobWetlands Project, the Japan Aerospace Exploration Agency's (JAXA) Kyoto and Carbon (K&C) Initiative, and the NASAMeasures program.

Keywords
Earth observation · Assessment · Monitoring · Mapping · GlobWetlands · Kyoto and carbon initiative · NASA measures

Introduction

The Ramsar Convention addresses the importance of developing and intensifying internationally coordinated actions for the conservation and wise use of wetlands. Achieving the main objectives of the Ramsar convention, which is to maintain the ecological character of wetlands and the ecosystem services they provide, requires having access to global, up-to-date and reliable information to complete national inventories and the ability to perform adequate monitoring activities, carry out assessments and put appropriate management and restoration plans in practice. In recent years, the use of satellite earth observations (EO) with innovative geo-spatial analyses has turned out to be a key tool and unique source of information for the Ramsar Convention (Mackay et al. 2009).

Remote sensing observations acquired over short to long-time frames by airborne and more particularly by spaceborne missions are increasingly used to support the implementation of the Convention by:

- Increasing scientific knowledge and understanding of the physical, biological and chemical components of wetlands (e.g., soil, water, plants and nutrients) and the interactions between these, as well as the changing role of wetlands in the global environment.
- Supporting the efficient management of wetland areas, through provision of local to global, up-to-date and useful information that allows more efficient inventory, assessment and monitoring of wetland sites and catchments, and the development and implementation of restoration or rehabilitation plans.
- Enhancing reporting mechanisms of the Convention, strengthening the ability of the contracting parties to monitor efficiently treaty-relevant behavior, and facilitating better decision-making through the generation of common datasets and information systems, and through harmonization of formats, methods and procedures for gathering and analyzing information.

A number of projects are aimed at supporting the Ramsar Convention through characterization, mapping and monitoring of wetlands at regional to global levels. Amongst these are the ESA GlobWetlands Project, the Japanese Space Exploration Agency (JAXA) Kyoto and Carbon (K&C) Initiative and the NASA Measures program.

Table 1 EO technologies tested during the GlobWetland project

	GlobWetland I (2003–08)	GlobWetland II (2009–13)
Land use/land cover	Development & demonstration	Pre-operational deployment
Change detection	Development & demonstration	Pre-operational deployment
Water cycle regime	Development & demonstration	Pre-operational deployment
Wetland identification and delineation	Feasibility study	Demonstration
Water quality	Feasibility study	Not covered

The ESA GlobWetland Projects

The European Space Agency's (ESA) GlobWetland projects (Fernandez-Prieto et al. 2006; Jones et al. 2009) were designed to demonstrate, across large areas, the unique capabilities offered by EO assets to wetland mapping and monitoring. The GlobWetland projects aimed to facilitate the use of EO technologies by all communities engaged in wetlands conservation, restoration and management in support of the Ramsar Convention. During the GlobWetland projects, different EO applications were studied, from feasibility assessments to research and demonstration and up to pre-operational deployment (Table 1).

GlobWetland I (2003–2008) was launched in collaboration with the Ramsar Secretariat and was aimed at developing and demonstrating products and services based on remote sensing data that could be used to support wetland managers and national authorities in responding to the requirements of the Ramsar Convention and allowing more efficient monitoring of wetlands globally. This initial project involved 52 different wetlands that were distributed in 21 countries worldwide and relied on the direct collaboration of several local, national and regional conservation authorities and wetland managers.

The core products represented the basic set of common geo-information needed for all wetlands and included information on land use and land cover, long-term change analysis and mapping of the water cycle regime. A number of site-specific products were also generated in response to precise requests to better monitor and assess different conditions at the local scale, including water quality parameters (e.g., turbidity, suspended sediments and chlorophyll concentration), coastal wetlands dynamics (erosion and sedimentation), and topography for the delineation of wetlands and catchment areas and location of points of discharge. The GlobWetland I project organized the "GlobWetland Symposium: Looking at wetlands from space" in Frascati, Italy, in 2006, to stimulate discussions between the remote sensing and the wetland communities, review the current and emerging earth observation developments relevant for the inventory, assessment and monitoring of wetlands, and identify key scientific technical and policy-relevant challenges that remained at local to global levels.

Building on the findings of the GlobWetland I project and on the recommendations from the GlobWetland symposium, GlobWetland II (2009–2013; Paganini et al. 2010) aimed to further demonstrate the operational capacity of Earth Observation and geo-information technologies to provide wetland practitioners with effective decision support tools for wetland management and conservation activities. The overarching objective was to contribute to the establishment of a Global Wetlands Observing System (GWOS) in accordance with the Ramsar Strategic Plan and represented a pilot effort by the Ramsar Convention to put in place a regional observation system over the Mediterranean Basin in partnership with the MedWet initiative, the Mediterranean Wetlands Observatory (MWO) and ten countries from the Southern Mediterranean Basin (from Morocco to Jordan). The majority of wetlands were coastal and distributed equally along the southern Mediterranean coast. Time-series of Landsat MSS, TM and ETM+ data were used to produce the base geospatial products for three points in time (1975, 1990 and 2005), and then to derive wetland indicators. The project also developed the GlobWetland toolbox for end-to-end processing of satellite images, and installed the software toolbox at the premises of all partner organisations, with adequate training and capacity building.

The first objective of the GlobWetland II mapping activities was to build on the knowledge gained through its predecessor and produce 1,800 thematic maps, at 1:50,000 to 1:100,000 scale, of land use and land cover (including wetland typologies), change detection (for long term trend analysis) and water cycle regimes (including the estimation of annual variations in water tables) for a total of 200 wetlands.

The Land Use/Land Cover (LULC) maps (Fig. 1) provided a detailed classification of all land parcels within the area of interest, which typically included the wetland site, and the surrounding areas. The Corine Land Cover system (EC 1993) was used for classification, although was adapted to incorporate the Ramsar wetlands classification system. The thematic information provided followed a 5 level nomenclature, with the first distinguishing between water, natural areas and artificial surfaces and the most detailed levels include wetland typologies defined by the Ramsar Convention. Maps were georeferenced to the corresponding national systems and overall thematic accuracies ranged typically from 85% to 95%.

The Change Detection (CD) maps (Fig. 2) provided a historical comparison of land use and cover in the wetland site and its surroundings between two reference dates, thereby providing a synoptic view of the main changes occurring in the catchment areas, whether natural or anthropogenic. From these maps, the evolution of many wetlands could be observed and threats affecting the wetlands and their impacts over time could also be assessed.

The Water Cycle Regime (WCR) maps (Fig. 3) provided an overview of the annual variations of water extent (minimum and maximum) and differentiated permanent and seasonally inundated water bodies and associated vegetation. The

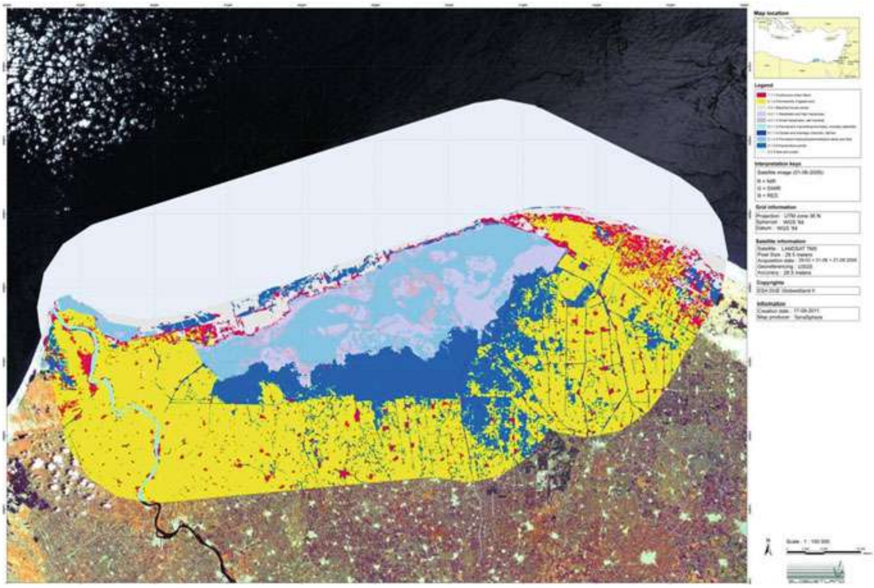

Fig. 1 Land Use Land Cover, Lake Burullus, Egypt, 2005. GlobWetland II project

Fig. 2 Land cover evolution, Lake Ichkheul, Tunisia, 1972–90, GlobWetland II project (Source of information: Landsat TM/ETM, USGS)

Fig. 3 Water cycle regimes (WCR), Merja Zerga, Morocco, 2005, GlobWetland II project (Source of information: Landsat TM/ETM, USGS)

WCR maps provided wetland managers with a unique capacity to monitor and characterize the dynamics of water retention and flow and how these affected the overall ecosystems.

The base maps of LULC, CD and WCR were then further elaborated with geospatial information technologies to derive indicators on wetland status and trends. This served the needs of the Ramsar Convention and of the Mediterranean Wetlands Observatory, in that they provided, for the 200 sites studied, four GlobWetland Indicators relating to the status and trends in wetland ecosystem extent, in wetland threats and in water quantity and extent with these being:

- Changes in wetland area (% wetland loss or gain between two epochs);
- Area of wetland ecosystem inundated (% of seasonally and permanently flooded area);
- Causes of wetland changes (areas lost to agriculture or urban/artificial surfaces between two epochs);
- Status and trends of wetland threats.

These indicators were generated for each wetland site and for each point in time, and also aggregated at an administrative level to derive maps for individual countries and regions.

In addition to the above core products (LULC, CD and WCR), the GlobWetland projects have experimented with EO techniques with promising applications that still need some further developments in order to become operational:

- Maps of wetland ecosystem extent to identify and delineate wetlands as a support to wetland inventories. Such a product serves the needs of national agencies interested in exploring the options to reduce costs associated with large wetland inventory exercises. The production of such maps requires long and dense time series of satellite images to cope with the high spatial and temporal variability;
- Maps of water quality indicators (chlorophyll concentration, total suspended matters, yellow matters). The extraction of water quality parameters in inland waters is still a complex process in particular for shallow waters where the bio-optical properties of the water constituents are contaminated by the reflection of the bottom of the water body. In spite of the still experimental stage of the retrieval methods available, EO satellites can provide today and under certain conditions, accurate information on water quality (Fig. 4).

The GlobWetland Toolbox

To assist the wetland community at large with the processing of satellite images, GlobWetland II developed an easy to use and cost effective toolkit for end-to-end processing of satellite images (Fig. 5), including pre-processing, segmentation and classification, computation of wetland indicators and a Web-GIS system for map distribution (Fig. 6). The toolkit was installed within each of the different organisations of the ten participating countries and user handbooks, targeted training and capacity building, and technical assistance were provided by the project team. These toolboxes have subsequently been integrated within the geographical information systems (GIS) of the national bodies. The Mediterranean Wetlands Observatory decided to continue the mapping undertaken within GlobWetland II and used the GlobWetland Toolbox to produce wetland maps and indicators over the remaining 17 Mediterranean countries of the MedWet initiative, with the objective to publish, in 2014, a special report on the status and trends of wetlands in the Mediterranean basin.

The JAXA K&C Initiative

The Kyoto and Carbon ("K&C") Initiative (Lowry et al. 2009) is a project led by the Japan Aerospace Exploration Agency (JAXA) and supported by an international Science Team consisting of approximately 25 research groups from some 14 countries. The project builds on the experience gained from the JERS-1 Global Rainforest and Boreal Forest Mapping (GRFM/GBFM) projects, which demonstrated the utility

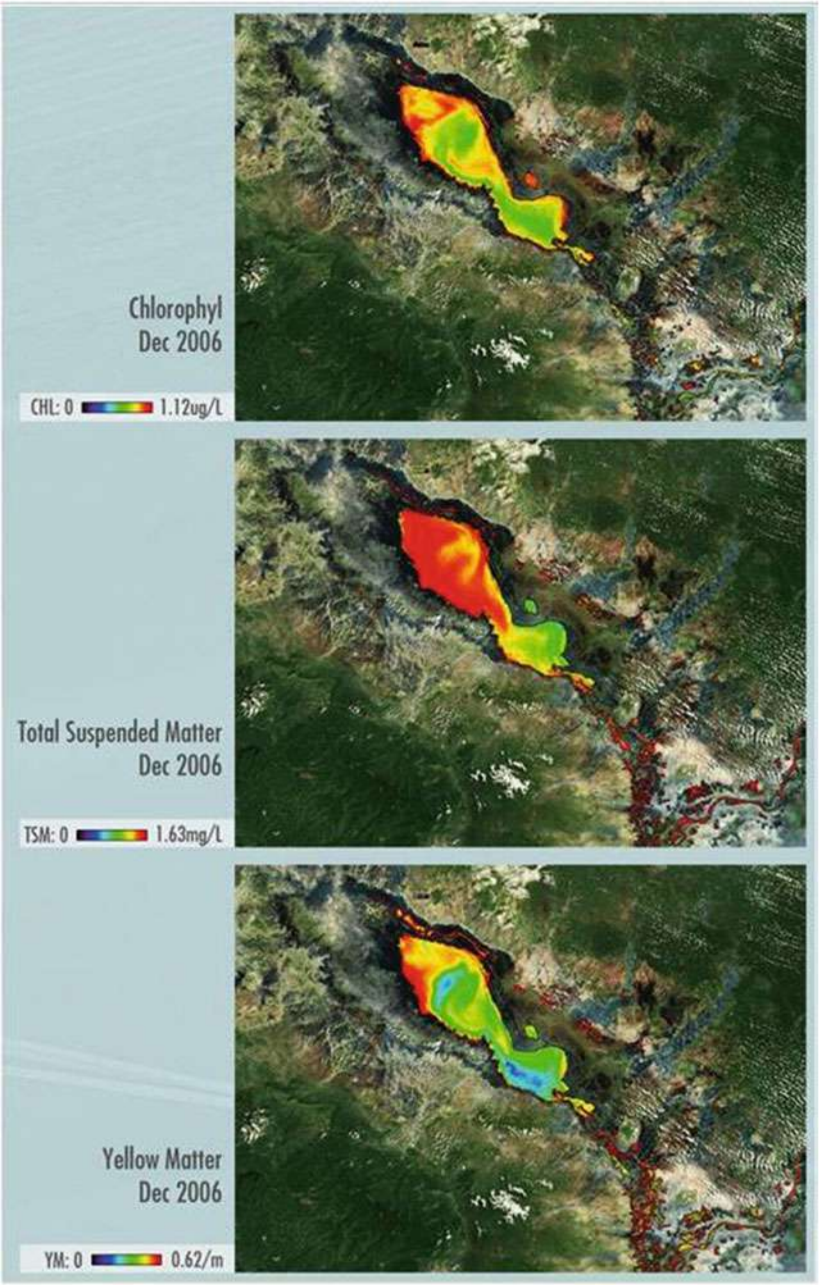

Fig. 4 Water quality parameters (*Chlorophyll concentration, Total Suspended Matters, Yellow matters*) Mekong River Basin, 2006, GlobWetland I project (Source of information: ENVISAT MERIS, ESA)

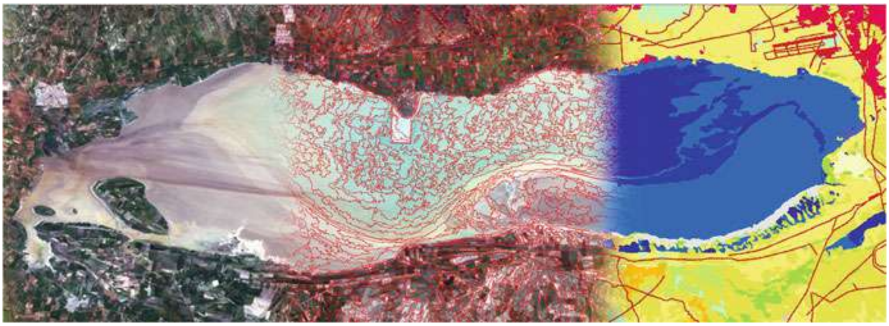

Fig. 5 End to end processing from satellite image geometric and radiometric corrections, to image segmentation and land cover classification, as undertaken within the GlobWetland II project

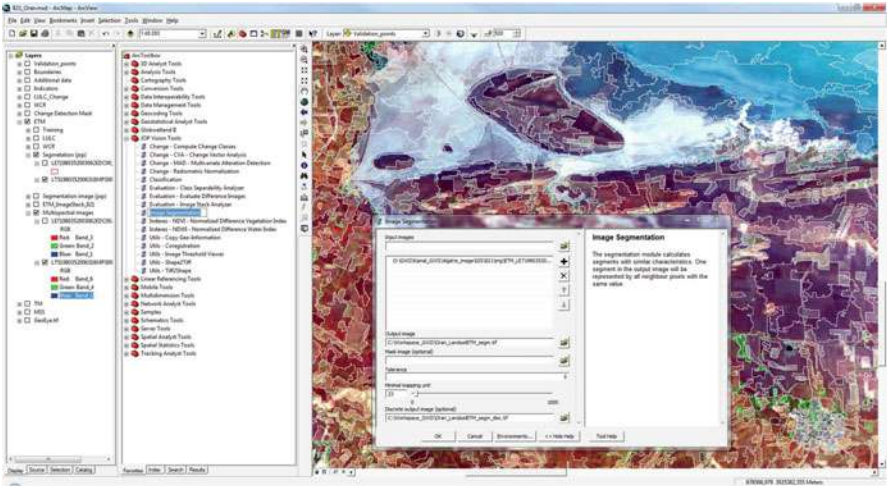

Fig. 6 Snapshot of the GlobWetland II toolbox

of L-band SAR data for mapping and monitoring forest and wetland areas, as well as the importance of providing spatially and temporally consistent satellite acquisitions for regional scale monitoring and surveillance.

The objective of the initiative is to develop regional scale applications and thematic products, derived primarily from data acquired by the Phased Array L-band Synthetic Aperture Radar (PALSAR) instrument on board the Advance Land Observing Satellite (ALOS) satellite, that support the data and information needs raised by international environmental conventions (such as the Ramsar Convention on Wetlands of International Importance), carbon cycle science, and conservation of the environment. The initiative is undertaken within the context of three themes which relate to global biomes: forest, desert and wetlands. Within the

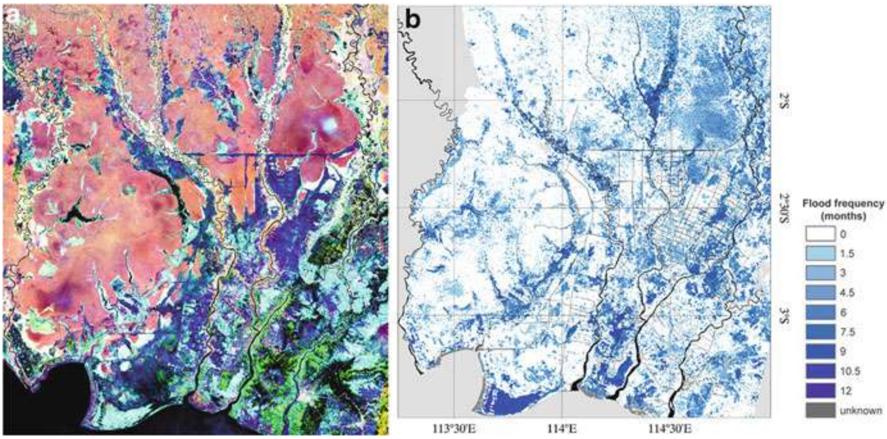

Fig. 7 (**a**) 2007 ALOS PALSAR colour composite image of South Kalimantan, Indonesia, where major conversion of peat swamp forest to rice cultivation is occurring. Canals built to drain water from the swamps are visible as *blue* linear features. (**b**) Map derived from PALSAR ScanSAR time-series showing flooding frequency in 2007 (Hoekman et al. 2010. © JAXA METI)

wetlands theme, the key products that have been identified and are generated from the PALSAR data aim to support global wetland inventory and change analysis. Key outcomes from the K&C Initiative have included mapping of wetlands and flood levels in Indonesia (Fig. 7a, b), with these supporting sustainable peatland management strategies. Flood dynamics have also been quantified in the Pantanal with the resulting maps used to define strategic areas that should be protected for wildlife and help answer questions about how changes in climate and human activities over time affect the biodiversity and people who are dependent upon the wetlands. In Africa, multi-temporal L-band Synthetic Aperture Radar (SAR) datasets have been combined with Landsat Thematic Mapper (TM) and ASTER images, digital elevation models, and vegetation species data to provide information on wetland ecology and hydrology (Fig. 8; Rebelo 2010). The Global Mangrove Watch (GMW) has also been developed with this aimed at monitoring the changing extent of mangroves globally using combinations of JERS-1 SAR, ALOS PALSAR and ALOS-2 PALSAR-2. The K&C research phases leading to the GMW have focused specifically on establishing methods for revising baselines of mangrove extent for the mid 1990s, 2007, 2008, 2009 and 2010 based on JERS-1 SAR and ALOS PALSAR data respectively. Following observation by the ALOS-2 PALSAR-2, revised baselines of mangrove extent are being generated for 2015 and 2016.

The NASA Measures Project

A component of the NASA Measures project is to construct an Inundated Wetlands Earth System Data Record (IW-ESDR). This involves the use of SAR for continental scale mapping of wetland extent, seasonal inundation dynamics and vegetation

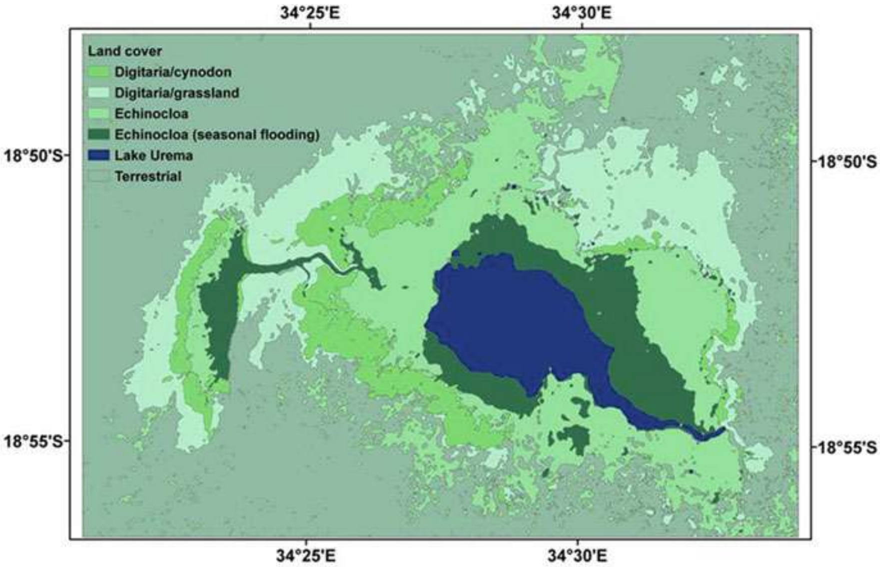

Fig. 8 Dominant vegetation types and flooding regime, Lake Urema, Mozambique, derived from multi-temporal ALOS PALSAR data (© JAXA METI)

(100 m spatial resolution) and multiple satellite observations to generate global monthly maps of inundation extent at ~25 m resolution. The intention is to generate the first global scale dataset of wetland inundation and vegetation, within this then providing a new understanding of how wetlands are functioning at regional to global scales. Such information can be used to quantify, for example, greenhouse gas emissions (primarily methane and CO_2), climate feedbacks and water exchanges.

Conclusion

A number of key projects, including those outlined above, have led to significant advances in the observation, characterization, mapping and monitoring of wetlands. Within these projects, a wide range of remote sensing data have been used over varying time frames, with these ranging from very high resolution (VHR) optical data for localized studies to SAR data acquired consistently and over large regions.

The GlobWetland projects fulfilled their objectives of facilitating the integration of EO data and analysis techniques into the conservation and management of wetlands, serving the needs of wetland conservation practitioners from wetland site managers to stakeholders of the Ramsar Convention. The projects have also led to the use of these data being considered essential tools for the inventory, mapping and monitoring of wetland ecosystems worldwide. As such, the GlobWetland projects have highlighted the cost effectiveness and efficiency of EO data for fulfilling the requirements of the Ramsar Convention, in prioritizing management measures and in policy making, reporting and planning. Indeed, the Scientific and Technical Review Panel (STRP) of the Ramsar Convention has strongly recommended the use of these data at all scales.

Whilst a number of the methods developed for mapping land use and cover, detecting change and quantifying water cycle regimes are sufficiently mature for uptake by the wetland community at large, further development is needed for the production of water quality parameters or wetlands inventory. This is likely to be fulfilled through the third GlobWetland project, GlobWetland Africa (2014–2017), which has the principal objective of addressing wetland conservation and management from a water basin perspective with a focus on the African continent and the use of satellite data from the Sentinel constellations of the Copernicus initiative.

The JAXA K&C Initiative provided L-band SAR data at a regional level, which could be used to support characterization, mapping and monitoring of a range of wetland types, including peat swamps and mangroves. With the launch of the ALOS-2 PALSAR, considerable scope exists for ongoing observations of wetlands at a global scale. Such data can also be used to support the objectives of the NASA IW-ESDR, which is advancing the integration of remote sensing data from a diversity of sources.

The Sentinel 1 and Sentinel 2 satellite missions of the European Copernicus initiative will also provide long-term access to enhanced radar observations and high resolution super-spectral data opening a new scenario for mapping, assessment and monitoring of wetlands worldwide. The C-band imaging radar of the Sentinel 1 mission is providing all-weather day-and-night imagery which will be extremely useful for monitoring wetlands in cloudy conditions. The Sentinel-2 mission will provide systematic optical observations of all terrestrial and coastal zones, at 10 m spatial resolution, with a swath width of 290 km. Together with its twin satellite, it will cover the Earth's surface with a repeat cycle of 5 days at the equator. The impressive footprint of Sentinel 2 along with the short revisit time and its systematic acquisition policy will allows rapid changes in ecosystems to be precisely monitored and is ideally suited to monitor sensitive habitats such as wetlands. It will allow for seasonal and permanent changes in wetlands to be monitored with higher accuracy. The Copernicus Sentinel data policy, with its full and open access for all users worldwide, is an important initiative that will largely facilitate the uptake of these new technologies by the wetland community.

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Section XVI

Wetland Monitoring and Assessment

Charlie J. Stratford

Wetland Assessment: Overview

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Charlie J. Stratford

Contents

Abstract

Wetland assessment is an important part of the wetland policy process and is defined as the identification of the status of, and threats to, wetlands as a basis for the collection of more specific information through monitoring activities. The overall aim of assessment is to answer the question: "what are the values that this wetland provides and how can humans benefit from them?" There is thus a close relationship between wetland assessment and wetland monitoring, with assessment sometimes relying on the results of monitoring, and monitoring being triggered by the results of an assessment. A range of assessment types have been developed, each with its own focus and applicability, ranging from hydrological, biological, functional and integrated assessments to vulnerability assessment. Determining and describing the status, characteristics, and worth of a particular wetland is often done by measuring the current condition of a wetland area within the context of a reference condition. Assessments can make use of existing data or collect up-to-date site data, provided by a combination of desk

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and field-based investigation, often with a combination of expert opinion and scientific knowledge. Depending on its focus, the scale of assessments can range from a broad overview of many functions and services on a regional or watershed scale to very specific investigations into a single wetland site. Future challenges of assessment include developments in assessment technology (e.g., using satellite sensors to assess large areas) and the involvement of the general public in collecting useful scientific information (so-called citizen science).

Keywords
Wetland assessment · Wetland monitoring · Hydrological assessment · Biological assessment · Functional assessment · Integrated assessment

Introduction

Wetland assessment has been carried out by humans for hundreds, possibly thousands, of years. Initially, in an informal and unstructured way, to identify the values or hazards that might benefit or endanger those living nearby. The move to a more structured assessment approach has coincided both with greater awareness of the value of wetland habitats and recognition of the widespread damage that is being done to the natural world. Developing over the last 30–40 years, current assessment methods seek both to provide a greater understanding of the functions and value of wetland habitats and also to meet an ever-increasing need to demonstrate and defend the needs of sensitive areas of habitat in the face of human-induced pressures.

Wetland assessment is now engrained in the laws of many nations through various pieces of legislation and is a necessary preliminary part of much development activity (e.g., Town and Country Planning (Environmental Impact Assessment) Regulations; UK Government 2011). Inventory, assessment and monitoring of wetlands are fundamental tools that provide the basis for successful implementation of the Ramsar Convention on Wetlands (Ramsar Convention Secretariat 2010). There is an increasing onus on countries to provide information on the quantity and quality of their wetland habitats and to take steps to mitigate degradation. Assessment provides the mechanism for doing this (e.g., the EU Habitats Directive; European Commission 1992). Wetland assessment is a rapidly developing area.

This section provides an overview of wetland assessment, including the evolution of the process, the range of processes currently in use, some of the methods employed, and the future challenges facing effective assessment. A more detailed exploration of each method is provided in the subsequent sections.

Relationship Between Assessment and Monitoring

The Ramsar Convention defines wetland inventory, wetland assessment, and wetland monitoring as follows (Ramsar Convention Secretariat 2010):

- Wetland Inventory: the collection and/or collation of core information for wetland management, including the provision of an information base for specific assessment and monitoring activities.
- Wetland Assessment: the identification of the status of, and threats to, wetlands as a basis for the collection of more specific information through monitoring activities.
- Wetland Monitoring: the collection of specific information for management purposes in response to hypotheses derived from assessment activities, and the use of these monitoring results for implementing management. The collection of time-series information that is not hypothesis-driven from wetland assessment is here termed surveillance rather than monitoring.

There is a close relationship between wetland assessment and wetland monitoring, with assessment sometimes relying on the results of monitoring, and monitoring being triggered by the results of an assessment. The two often work together in order to achieve the ultimate goal of establishing current condition, providing the factual underpinning for action, and observing to see whether an action is achieving its objective. When assessments are repeated over time, an established replicable monitoring programme is required in order to provide suitable data. Another important aspect of wetland assessment is understanding vulnerability and the magnitude and duration of impact that may result from a certain pressure. For example, a wetland may respond differently to an acute incident such as a one-off chemical spillage compared to a chronic condition such as sediment input from run-off in the upstream catchment. On-going monitoring can provide the information required to assess various aspects of a wetland.

The Evolution of Assessment Methods

There is evidence of human interaction with wetland areas from much of the world dating back many years. Archaeological discoveries often provide evidence of very early assessment methods, where early settlers chose to live close to wetlands, realising these habitats offered various benefits. Excavations in Japan revealed organic remains of fish traps, ground-level dwellings, and trackways with ages estimated to be around 5,000 years BP. Sites across Europe such as Corlea bog in Ireland, Noyen-sur-Seine in France, and Usvyaty in Russia all show evidence of human settlement. Remains of baskets and harpoon points have been found on the northwest coast of North America, estimated to be between 4,500 and 3,000 years old (Coles 1992).

As sites rich in food, fiber, fuel, and water, living close to a wetland had many advantages. Resources invested in constructing homes near to a wetland and/or putting in the infrastructure necessary to improve access to different parts of the wetland were justified by the material gains that they facilitated. It is likely that this early decision making also considered the possible disadvantages of living near a wetland, such as potentially increased numbers of disease-carrying insects.

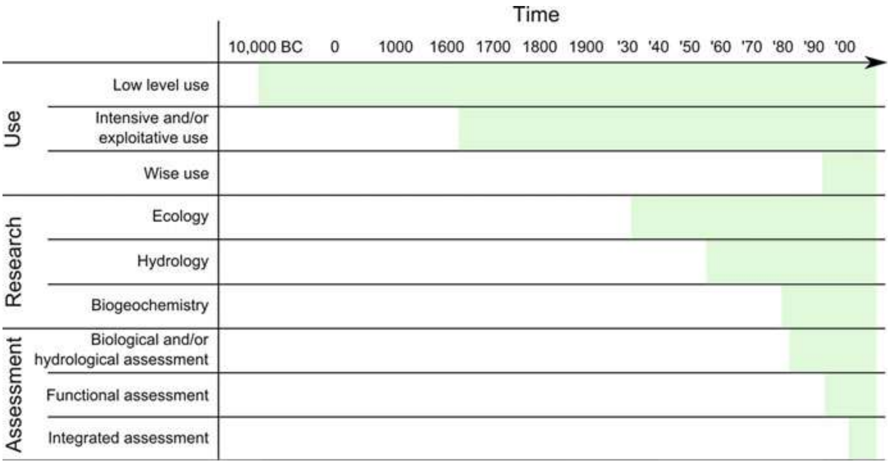

Fig. 1 The evolution of wetland assessment. Time (note nonlinear axis) is shown along the horizontal axis and different aspects of wetland use, research, and assessment are shown on the vertical axis

Although modern methods are more structured and tend to apply a more rigorous set of tests in arriving at an assessment, the overall aim of assessment is in some ways unchanged, i.e., to answer the question: "what are the values that this wetland provides and how can humans benefit from them?" Such a broad question quickly becomes a veritable "can of worms" when challenged, giving rise to subquestions (and a multitude of other, similar questions) such as:

- Can value only be judged from the perspective of "what is the value to humans?"
- Is one value more important that another?
- Is it ok to exploit one value to the detriment of another?

As a result of this complexity a whole range of assessment types has been developed, each with its own focus and applicability. The evolution of wetland assessment reflects the increasing intensity with which humans manipulate landscapes, the growing interest in natural sciences, and most recently a desire to reduce the damage that is being done to habitats of all kinds (Fig. 1).

Early use of wetlands most likely had a minimal impact on the wetlands themselves as the intensity of activity was low. However, as societies and countries "developed," the use of wetlands became more exploitative. Wetlands were modified and in many countries large areas of wetland, too wet for habitation or agriculture, were drained (Biebighauser 2007). Drainage of the Fenland area of the UK became more widespread during the 1600s and with the industrial revolution came the ability to drain much larger areas more effectively (Godwin 1978). The resulting fertile, workable agricultural land was viewed by many as a step forward, meeting the growing need for food supply and reducing flood risk (Baldock et al. 1984). Similarly throughout Europe, large areas of wetland were drained in order to facilitate agriculture. It is estimated that Spain has lost 60% of its inland freshwater wetlands since 1970, Lithuania has lost 70% of its wetlands in the last 30 years, and 67% of France's wetlands have disappeared in the last 100 years (Silva et al. 2007). It is only recently that the idea of "sustainable benefit" has become more prominent through promotion of ideas such as the "wise use" concept (Finlayson 2012).

Types of Assessment

Assessment methods have developed both as society's desire to "use" wetlands for its benefit has increased and as scientific understanding of wetland functioning has improved. The different assessment methods are described here and are summarized in Table 1. Specific assessment frameworks, such as Strategic Environmental Assessment (SEA) and Environmental Impact Assessment (EIA), are described in detail in other sections of this volume.

Assessments can be tightly focused, concentrating on just one aspect of the wetland. Hydrological assessment, for example, seeks to understand and ideally quantify the role of a wetland system in influencing the hydrological regime of the surrounding area. This might be particularly relevant if a wetland is thought to ameliorate or contribute to flood risk. A biological assessment would most likely give an indication of the ecological "health" of the ecosystem or might be applicable if a rare habitat or species is present within the wetland.

Table 1 Examples of different types of wetland assessment

Wetland assessment methods	
Type of assessment	Purpose
Hydrological assessment	To develop a conceptual understanding of hydrological inputs and outputs to and from a wetland system (Acreman and Miller 2007)
Biological assessment	"*To evaluate the health of a waterbody by directly measuring the condition of one or more of its taxonomic assemblages and supporting chemical and physical attributes*" (USEPA 2002)
Functional assessment	*"Wetland functional assessments were developed for the specific purpose of quantifying the levels of function of an existing wetland (impacted site) or the levels of function of a compensatory, mitigation site based on predicted future conditions"* (USDA 2008)
Integrated assessment	*"A set of methods that can be used to investigate the links between biodiversity, economics, and livelihoods in wetlands and to identify and address potential conflicts of interest between conservation and development objectives"* (Springate-Baginski et al. 2009)
Vulnerability assessment	The following example is focused largely on climate change – "*an approach that can provide information and guidance for maintaining the ecological character of wetlands which are subject to adverse change as a consequence of climate change (including sea level rise), while recognizing that climate change will interact with the many other anthropocentric pressures on wetlands"* (Gitay et al. 2011)

A more human-centric view, which seeks to establish what “functions” or “services” a wetland provides either to humans, wildlife, or the environment, is achieved through functional assessment (Maltby 2009). The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) provides a full list of services and examples include carbon sequestration, provision of food and fuel, water storage, and areas of natural beauty. Functional assessment considers the direct and/or indirect benefits provided by wetlands at a range of scales. For example, Acreman et al. (2003) carried out a modeling exercise on the River Cherwell, Oxfordshire, UK, to quantify the impact of embankment removal on downstream river flows. With the floodplain in hydrological connection with the river and therefore able to readily receive flood water, the model results showed a 30% reduction in peak flood flow downstream.

With wetlands increasingly viewed as integral parts of larger systems, not only in terms of biology and hydrology but also in their interaction with humans, a broader integrated assessment approach often makes more sense. Integrated assessment seeks to assess three main aspects of the wetland’s interaction with human society and bring these together in order to assess the interlinkages and connectivity between wetland condition and economic/livelihood status or to express this information in a form and with a focus that can inform and influence real-world conservation and development planning (Springate-Baginski et al. 2009). The three aspects of integrated assessment are:

- The ecosystem and the physical conditions that support it
- The value of the ecosystem services that wetlands provide
- The role of the wetland in supporting the well-being of local people

With increasing focus now on protecting wetland areas, a further assessment method has been developed with the aim of identifying wetlands at risk from one or more threats: wetland vulnerability assessment takes into account the relationship between exposure to a particular risk event, the impacts of that event on a wetland, and the ability of the wetland to cope with the impacts or the efforts needed to minimize the impacts. The resilience and sensitivity of the study area is included in the assessment (Gitay et al. 2011).

An Overview of the Assessment Process

Wetland assessment is the process of determining and describing the status, characteristics, or worth of a particular wetland (Springate-Baginski et al. 2009). This is often done by measuring the current condition of a wetland area and presenting this within the context of a reference condition (Fig. 2). It is then possible to report whether the wetland is in good condition or not. The result will lead to recommendations for activities that either maintain the current condition or seek to mitigate the factors causing the current poor condition. The assessment process therefore

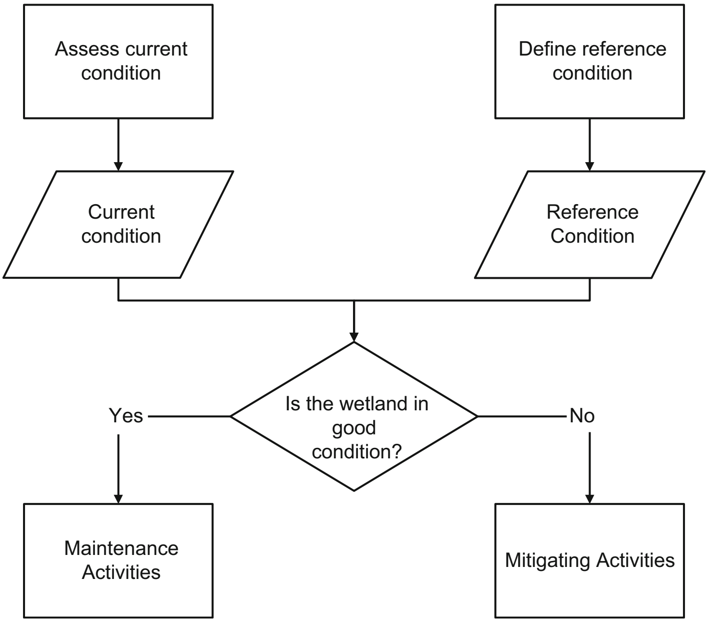

Fig. 2 Basic assessment process and potential outcomes

typically consists of establishing two pieces of information: the current condition of the focus area and the reference condition of the focus area.

Assessing the current condition of a site is likely to make use of existing data as well as require additional up-to-date site data, and this is provided by a combination of desk and field-based investigation. The scope of the data required for this will depend upon the type of assessment being carried out. The definition of a reference condition aims to establish the state a site would be in, in the absence of all or some of the pressures under investigation. A combination of expert opinion, scientific knowledge, and, where possible, identification of an unimpacted comparable site will provide much of the information required. This should take into account the setting (e.g., hydrogeomorphic and eco-regional) and the overall wetland landscape profile, representing the abundance, by class, of wetlands in that occur in the geographical area (USEPA 2006).

Selection of appropriate indicators of condition will be guided by the assessment focus with close reference to existing scientific information. The indicators should be tested at a range of relevant sites and across the range of conditions to be assessed so that their appropriateness and usefulness can be confirmed. For this purpose, it can be useful to establish a network of sites that help establish both the reference conditions and range of assessment conditions; however, this may not always be possible and even when possible could prove to be expensive.

An assessment can take many forms depending on its focus. It could for instance be a broad overview of many functions and services, a continental-scale review of all

Table 2 Three types of wetland assessment method that can be developed to support program objectives (Compiled from data in USEPA 2006)

	Products/applications
Level 1 – Landscape assessment These assessments rely almost entirely on Geographic Information Systems and remote sensing data to obtain information about watershed conditions and the distribution and abundance of wetland types in the watershed. Typical assessment indicators include wetland coverage, land use, and land cover Wetland landscape profiles and landscape development indices (LDI) are used to characterize the lands that surround the assessed wetland. Metrics used in the LDI approach, such as road density, percent forest cover, land use category, and presence of drainage ditches, can provide preliminary information on wetland condition within a watershed This level of assessment can help to target areas for level 2 and level 3 assessments	• Targeting restoration and monitoring • Landscape condition assessment • Status and trends • Example – Wetland extent trends analysis that is conducted by the US Fish and Wildlife Service's National Wetland Inventory is a Level 1 type of assessment (US Fish and Wildlife Service 2014)
Level 2 – Rapid wetland assessment The Convention on Biological Diversity and Ramsar Convention define rapid assessment as a synoptic assessment, which is often undertaken as a matter of urgency, in the shortest timeframe possible to produce reliable and applicable results for its designed purpose (CBD-Ramsar 2006) Rapid assessments use relatively simple metrics for collecting data at specific wetland sites. Their methods should provide a single rating or score that shows where a wetland falls on the continuum ranging from full ecological integrity (or least impacted condition) to highly degraded (poor conditions). Assessment is often based on the characterization of stressors known to limit wetland functions (e.g., road crossings, tile drainage, ditching.). A "rapid" method should take two people no more than four hours of field time, and one half day of office preparation and data analysis to reach a condition score	• Integrated reporting • Watershed planning • Implementation of monitoring of restoration projects • Example – Kotze et al. (2012) carried out rapid assessment of ecological condition in South Africa
Level 3 – Intensive site assessment This is a more rigorous, field-based method that provides higher resolution information on the condition of wetlands within an assessment area, often employing wetland bioassessment procedures or hydrogeomorphic functional assessment methods. It produces quantitative data with known certainty of wetland condition within an assessment area and is used to refine rapid wetland assessment methods and diagnose the causes of wetland degradation. Assessment is typically accomplished using indices of biological integrity or hydrogeomorphic function	• Support the development of water quality standards that are protective of wetlands • Develop design and performance standards for wetland restoration • Verify and refine levels 1 and 2 methods • Integrated reporting • Example – intensive assessment of the Upper Juniata Watershed, Pennsylvania (Hychka et al. 2007)

wetland habitat types, or a very specific investigation into a single wetland site. Table 2 gives examples of three types of assessment ranging from landscape scale to site scale.

Future Challenges

The pressure on the natural environment is likely to increase in the future as population growth and societal development continue. The need for resources of all kinds is going to continue to put pressure on wetland habitats. Safeguarding wetlands will depend more and more on our ability to assess their condition, function, societal importance, and vulnerability. Technological advances have led to improvements in the efficiency and effectiveness with which assessments can be carried out. For example, the modern sensors fitted to some satellites are capable of collecting data from which a wide range of wetland information can be derived. Hydrological parameters such as salinity and soil moisture, topography, and the type and extent of different types of vegetation can be collected for very large areas using these remote sensing techniques (Klemas 2011). There have also been advances in defining the metrics to use in assessments, and target water table regimes and nutrient status now exist for many wetland plant species (e.g., Wheeler et al. 2004; Davy et al. 2010).

Citizen science, which aims to engage the public in collecting useful scientific information, is an emerging area with the potential to provide great benefit to wetland habitats. Not only can the information collected cover a wider scope than would be economically viable through traditional scientific methods, but facilitating public engagement can also promote interest and awareness of the environment. The Watsonville Wetlands Watch is a good example of a citizen science program in operation (http://www.watsonvillewetlandswatch.org).

So although the pressures on wetlands are unlikely to reduce, advances in technology, understanding, and integration will continue to address these pressures in the most effective manner.

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Wetland Assessment Methods: Biological Assessment

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J. Owen Mountford

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Abstract

The central purpose of biological wetland assessment is the quantitative and qualitative description and enumeration of the species, communities, and habitats present. To achieve this, often a basic inventory is needed of the wetland attributes (physical properties, species composition, and structure) and of soil, water chemistry, and hydrological characteristics. Specific aims may be: estimating the nature conservation value at a regional, national, or international scale; determining the need for protective measures; determining the contribution of the wetland to ecosystem services and the scope for exploitation; informing management and restoration strategies; or a comparison to reference conditions. Biological assessment can make use of a wide range of techniques including broad habitat mapping (e.g., terrestrial survey, aerial photography, satellite imagery), detailed community mapping, quadrat and/or transect surveys and phytosociological analysis, population counts of target species (e.g., as indicators of biological integrity or for conservation goals), fixed-point photography, sub-aqua diving

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surveys, remotely controlled cameras, structured grapnel surveys, and integrated survey systems. Nature conservation agencies have developed rapid assessment methods, which express the biological integrity of the wetland in metrics (indicators) that may support management decisions but can also be used to communicate information about the status of wetlands to the general public. A main future challenge is to develop consistency in assessment approaches and to ensure the availability of basic information, which will improve the comparability of assessment results and their application in policy making.

Keywords
Wetland assessment · Biological assessment · Wetland health · Wetland conservation

Introduction

Although methods for the assessment of wetland biota are broadly similar to those generally adopted for other habitats and species, the particular nature and variety of wetlands defined by the Ramsar Convention requires special considerations and approaches (Keddy 2000). The central purpose of biological assessment is the quantitative and qualitative description and enumeration of the species, communities, and habitats present. However, this basic inventory approach is very frequently the precursor to assessment for specific purposes such as Ecological Impact Assessment (Treweek 1999) or measurement of biological integrity (Danielson 2002).

Thus biological assessment could commence with examination of the attributes of a wetland (e.g., its physical properties, species composition, and structure) and identification of the properties of the wetland habitats (Hill et al. 2005). The biological properties of the wetland might comprise the extent of the habitat, the characteristic communities, the species composition and richness, and the presence/absence of typical or indicator species. Structural properties may include vegetation height, location, and extent of scrub invasion and pattern (e.g., the disposition of hummocks, hollows, and pools in bogs). These measures would be complemented by assessment of the soil (pH, nutrients, peat depth, etc.), the hydrological regime, and water chemistry. An example of a programme for assessment and monitoring is provided by Fig. 1 showing the key elements in Common Standards Monitoring of lowland wetlands (after JNCC 2004).

Assessment of water bodies demands measurement of further attributes. Thus, in open water bodies (lakes, pools, etc.), additional biological properties for assessment will include macrophyte abundance and the presence/distribution of emergent vegetation, while the depth and profile of the water body, the substrate type, and detailed assessment of water chemistry will be necessary. The dynamic nature of rivers presents further challenges to assessment, with supplementary biological measures focusing both on the vegetation of the banks and of the watercourse itself and nonbiological properties including river morphology (channel width, substrate type, etc.) and flow regime.

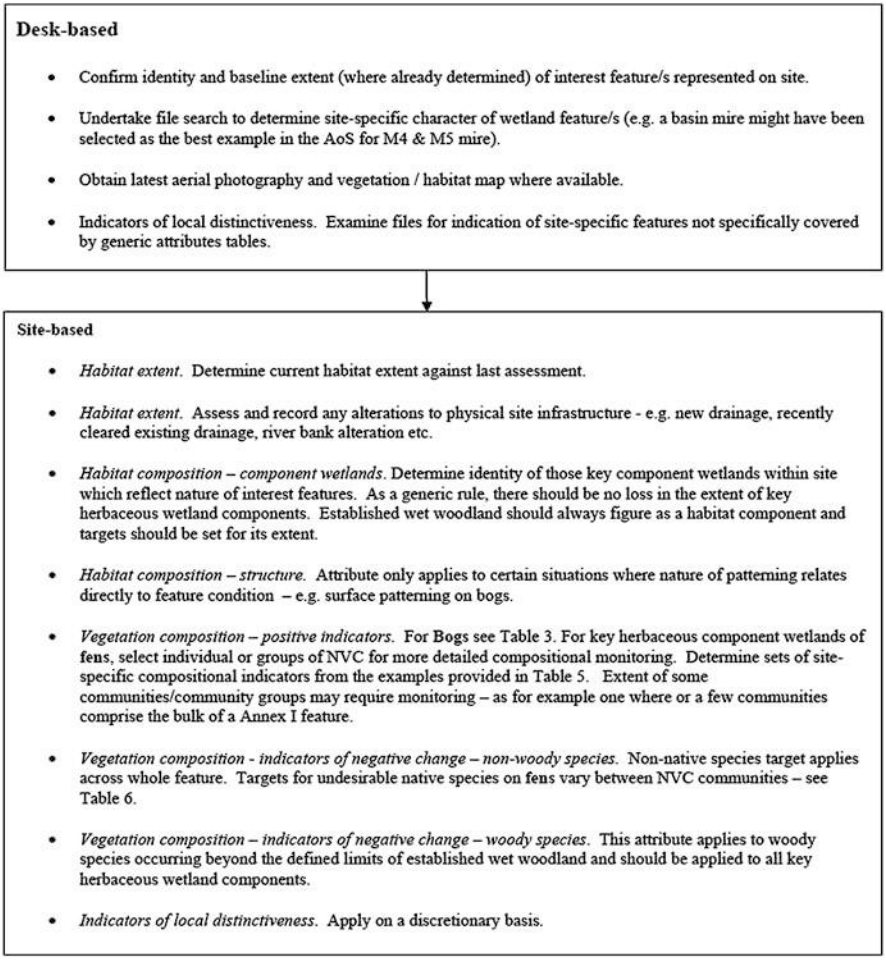

Fig. 1 Key elements in common standards monitoring of lowland wetlands (copyright: JNCC 2004; reproduced with permission)

Assessment Techniques

Assessment specifically of the biological properties of a terrestrial wetland such as a marsh, mire, or swamp can employ a range of techniques, frequently in combination:

- Broad habitat mapping using terrestrial survey, aerial photography, satellite imagery, or a combination of all three
- Detailed community mapping making use of expert terrestrial botanical survey, e.g., phytosociological approaches, UK National Vegetation Classification (NVC: Rodwell 1991–2000) etc.

- Quadrat and/or transect surveys (often with subsequent phytosociological analysis)
- Population counts of target species, these chosen either as indicators of biological integrity or because they have some conservation designation
- Fixed-point photography

In lakes and rivers, further tailored assessments of the biota might include:

- Use of sub-aqua diving surveys or remotely controlled cameras
- Structured grapnel surveys
- Integrated systems such as those used in the UK: River Habitat Survey (NRA 1992) and, for aquatic invertebrates, RIVPACS (Wright et al. 1997).

Detailed descriptions of these techniques and approaches may be found in Hill et al. (2005), Sutherland (1999), and numerous other standard texts.

Assessment Standards

The methods actually used for biological assessment of wetlands will clearly vary depending on the type of wetland being appraised. However, in all cases, the assessment will be made against some standard which allows the results of the assessment to be evaluated. For example, the habitat/community description may be compared with national classification systems (e.g., Rodwell (1991–2000) for the UK and Doniţă et al. (2005) for Romania) or, at a continental scale, e.g., the EUNIS system for Europe (Davies et al. 2004) or the habitats interpretation manual for Natura 2000 in the European Union (European Commission 2013). However, although these frameworks are very useful for appraising the survey data, they should be used with caution, and it is unwise, for example, to employ particular community descriptions as strict goals for habitat management and restoration, especially within a context of global environmental change. Nonetheless, these classifications are based on detailed research and inventory and constitute an invaluable baseline for comparison. Thus, the wetland may be evaluated to:

- Assess its nature conservation value at a regional, national, or international scale.
- Decide whether protective measures are required.
- Assess the scope for exploitation and the contribution of the wetland to ecosystem services.
- Inform the management, especially should restorative approaches be necessary.
- Measure the biological integrity of the site with comparison to reference conditions.

Rapid Assessment and Indicators

Given that detailed ecological appraisal and comparison with reference classifications may be both time-consuming and require particular expertise, nature conservation agencies have developed means of assessment that are more rapid and which employ a limited number of easily understood and measured metrics. Thus, in the UK, Common Standards Monitoring approaches were developed for specific habitats (e.g., JNCC 2004), site condition assessment is applied to designated sites (Natural England 2008), and areas provisionally selected for agri-environment schemes are subjected to a simplified biological assessment within the farm environment plan (FEP; Natural England 2010).

In the context of integrity, biological assessment of wetlands in North America has frequently been taken to mean measurement of the "health and integrity" of the wetland, i.e., "the ability to support and maintain a balanced, integrated and adaptive community of organisms having a species composition, diversity and functional organisation comparable to those of natural habitats within a region" (Karr and Dudley 1981). Many states in the USA have used an Index of Biological Integrity for this approach whereby following an initial classification of the wetland type, the site is subsampled across a gradient of human disturbance, and key wetland biota are recorded and assessed, notably algae, amphibians, birds, fish, macroinvertebrates, and vascular plants. A range of metrics (e.g., species richness in each group) are derived and tested against the gradient of disturbance. The results may be readily summarised graphically and presented to non-ecologists, e.g., managers and the general public (Danielson 2002). Such an approach is grounded in the indicative power of biological species to inform ecologists of the condition of a wetland.

Future Challenges

As with other forms of wetland assessment, the central challenges revolve around the lack of consistency in approach and the frequent absence of basic information, meaning that biological assessment risks proceeding without contextual information and in a manner that is less amenable to comparison. The challenge therefore is to adopt rigorous, practical, and rapid methods that can be directly compared and which provide immediately utilisable outputs for policy makers.

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Functional Assessment of Wetlands 237

Edward Maltby

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Abstract

Decisions affecting wetlands historically have been dominated by sectoral viewpoints, for example, drainage to improve agriculture or wildlife conservation. Government policies supporting wetland maintenance, where they existed, have been based largely on traditional conservation arguments. Such a nature conservation ethic has been insufficient alone to prevent progressive wetland degradation and loss on a global scale. It has commonly failed to compete with the more immediate socio-economic needs of people perceived to result from the alternative uses of wetland areas. This chapter examines the more recent approaches to wetland management which assess their functioning and the delivery of different ecosystem services valuable to people. Functional assessment can help with the more rational decision-making for wetland resources worldwide.

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Keywords
Wetland functioning · FAPS · Wetland assessment · Hydrogeomorphic Units (HGMUs) · Decision-making · Wetland management

Introduction

> Assessment of functioning is a key pre-requisite to making management decisions that affect delivery, by wetlands, of specific or particular combinations of ecosystem services. This knowledge is critical to provision of the evidence necessary to underpin strategic and policy decisions. Effective functional assessment provides an essential tool for those individuals, regulatory bodies and other government or non-government organisations that make informed decisions for appropriate wetland management (Maltby et al. 2009).

Decisions affecting wetlands historically have been dominated by sectoral viewpoints, for example, drainage to improve agriculture, excavation to make marinas or fishponds, or water management to maintain or restore particular environment conditions for conservation of special plants and/or animals. Government policies supporting wetlands, where they existed, were based largely on traditional conservation arguments emphasizing characteristics such as rarity, uniqueness, and representativeness of species and habitats found in wetlands (Maltby et al. 1994). Although generally linked to networks of "protected" areas, the nature conservation ethic has been insufficient alone to prevent progressive wetland degradation and loss on a global scale. It has commonly failed to compete with the more immediate socioeconomic needs of people perceived to result from the alternative uses of wetland areas. This has been particularly true of many developing countries wrestling with the combined problems of poverty, disease, and debt (Maltby 1986).

"A functional approach to wetland assessment is one that acknowledges that wetlands can perform work at a variety of scales in the landscape, which result in significant direct and/or indirect benefits to people, wildlife and the environment" (Maltby 2009). It effectively broadens consideration of wetlands from a view as conservation icons to recognizing their wider utilitarian importance resulting from multiple ecological, biogeochemical, and physical processes and their natural dynamics.

The need for a different perspective on wetlands which recognized the functions they performed (and the values resulting) was recognized initially in the United States where state wetland regulatory laws, starting in 1963, stimulated the first assessment methodologies (Larson 2009). Functional assessment subsequently became central to the US Federal government permitting process which regulates wetlands under Section 404 of the Clean Water Act (33 US Code 1344) and coupled to a "no net loss" policy. Conceptual development of assessment in the USA is described by Brinson (2009), and Smith (2009) provides practical examples in practice. In developing countries the underpinning science-based evidence is much more limited (see examples in Roggeri 2009). With financial support from the European Commission, an empirically based methodology of wetland functional assessment has been developed based on trans-European multidisciplinary research (Maltby 2009).

Wetland Functioning

Physical, chemical, and biological processes, individually or in combination, within the diverse structures of different wetland ecosystems, control different patterns and quantities of functioning such as hydrological (e.g., flood control), biogeochemical (e.g., nutrient retention), and ecological (e.g., habitat provision) functions. Functions result in the provision of different goods and services valuable to people such as flood-risk reduction, pollution reduction, and food chain support. These and other services are now commonly referred to as "ecosystem services," recognizing where and how they impart human benefits (Fig. 1). Examples of the linkages between processes, functions, and services are illustrated in Fig. 2.

Wetlands vary considerably in the range and quantity of functions which they provide. This relates to differences in climate, hydrology, geomorphology, soils, vegetation, and land use. There may be particular variation in the functioning of wetlands depending on position in the landscape. Figure 3 illustrates the sort

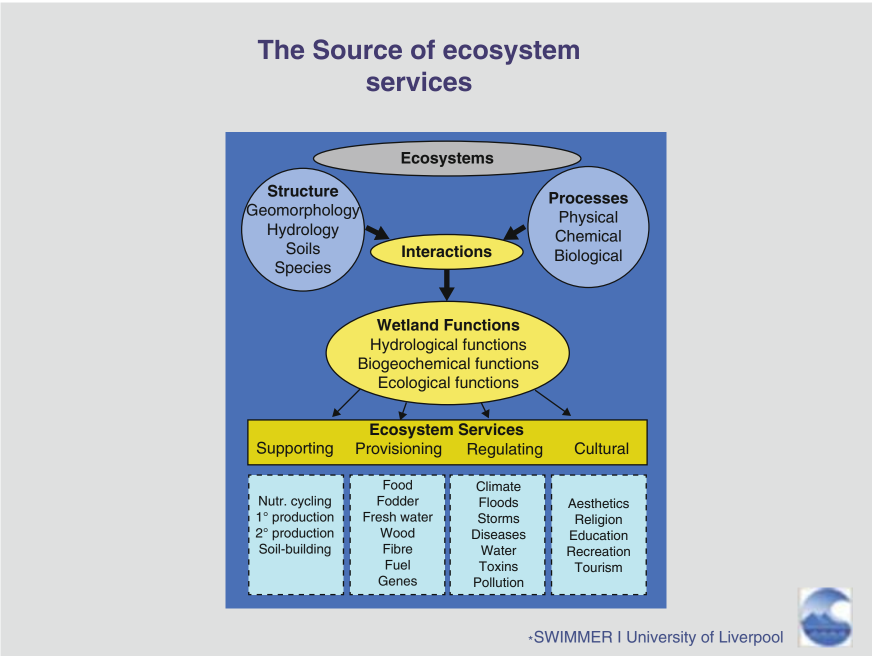


Fig. 1 Ecosystem services framework (From Maltby et al. 2009; copyright Blackwell Publishing Ltd., used with permission from Wiley)

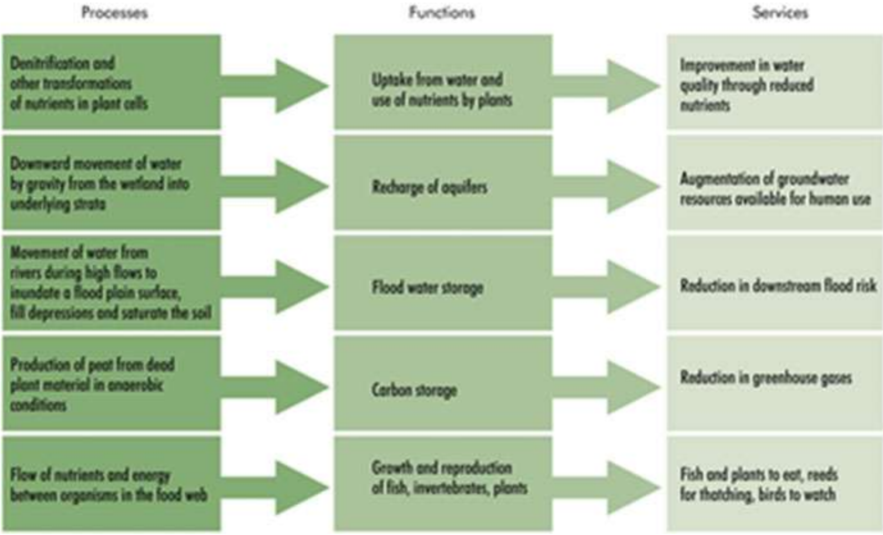


Fig. 2 Relationships between the processes, functions, and ecosystem services of freshwater habitats (Maltby et al. 2011; used with permission from UNEP-WCMC)

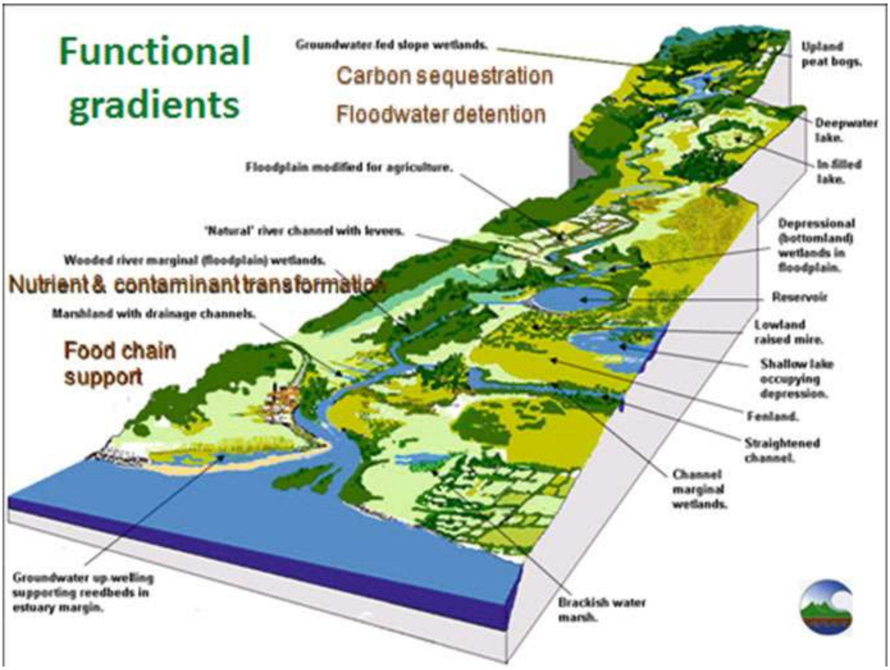


Fig. 3 Functional gradients at the catchment scale (Adapted from A. Aidoud in Maltby (2009), used with permission from Elsevier)

of functional gradient which might exist at the catchment scale. It shows a superimposition of a functional perspective on the more traditional classification of wetland types. Conventional typology gives no direct indication of how wetlands actually work. Functional assessment is required wherever it is important to distinguish which functions are being performed by which wetland or part of a wetland.

Multiple Methodologies in the United States and Canada to Meet Specific Policy and Regulatory Requirements

Bartoldus (1999) identified 40 different wetland assessment procedures for the United States alone, reflecting inter alia the diversity of ecosystems, species targeted, time/effort required, costs, outputs, expertise, and user needs. The rationale has been driven by policy and the need to better inform decision-makers of the public values provided by wetland functioning which may be lost or impaired by development. The inevitable legal disputes involving the rights of individuals and property ownership has led to the concept in the USA of "jurisdictional wetlands" with a need to delineate wetland areas (and their functions) on the basis of both legally as well as scientifically verifiable criteria (Maltby et al. 2009).

Tools for "rapid" assessment have tended to treat a wetland as a single functional unit. The most widely used method is the Wetland Evaluation Technique (WET) developed from the work of Adamus (1983) and Adamus et al. (1987). It has been the basis for training and use by the US Army Corps of Engineers (who together with the US Environmental Protection Agency are responsible for wetland permitting under the Clean Water Act legislation) and other regulatory programs (Brinson 2009).

The HGM (Hydrogeomorphic) Approach has been developed to overcome criticisms of WET (Smith et al. 1995). It was initially designed to estimate change in wetland condition by means of a quantitative comparison of altered/impacted wetlands with those that had not been altered and considered as "reference" wetlands using ecosystem functions as the basis for evaluation. Brinson (2009) provides a critical analysis of the HGM approach and cautions on two major limitations in practice. First, it "does not provide decision-makers with complete information to determine the full consequences of degrading a wetland," and second, "only a rapid level of assessment is provided and the building of a reference system is expensive."

In Canada (which supports 24% of the world's wetlands) the Federal Policy on Wetland Conservation, adopted in 1992, has prompted the development of various assessment methodologies based on the recognition of wetland functioning. They draw heavily on the experience from the United States. A critical review of methods can be found in Hanson et al. (2008). They concluded that no single method is best for all recognized situations in the United States and Canada. "The applicability of any method must be based on wetland types and questions to be addressed. . . .The use of any. . . .assessment method should be accompanied by a brief description of why that methodology was chosen."

Development of Functional Assessment Procedures in Europe

Approaches adopted in North America are generally unsuitable for application in Europe (and elsewhere), partly because of their strong link to regulatory mechanisms but largely because of close associations between wetlands and agriculture, the frequently small size of the majority of European wetlands, and their extreme diversity and intimate role in the culture and socioeconomics of local communities. While the European Water Framework Directive provides for an integrated policy for water management, it does not highlight wetlands as a prime regulatory focus.

Development of a means to assess wetland functioning in the European environment (with potential applications elsewhere in the world) has been based on empirical research findings and hence differs significantly from other approaches. The resulting methodology, Functional Assessment Procedures commonly referred to as the FAPS (Maltby 2009), is built on the identification of relationships between wetland processes and observable site properties resulting in the recognition and mapping of wetland areas in the landscape with particular functional characteristics.

The Hydrogeomorphic Unit (HGMU)

Unlike the scale of application of the HGM approach in the USA, the FAPS recognizes that there is commonly high spatial variability of hydrology and other site characteristics influencing functioning within single wetlands. The FAPS divide a site into the smallest practical identifiable units that exhibit functional homogeneity (Fig. 4). The term "hydrogeomorphic unit" (HGMU) describes these distinct areas which collectively make up the entire wetland site. They provide the spatial physical/landscape template for functional assessment and can be defined at different scales/levels of resolution in the landscape depending on actual wetland type and the purpose of assessment (Maltby 2009).

How Do the FAPS Work?

The basic pathway through the FAPS is outlined in Fig. 5 (Maltby 2009). Determination of processes likely to be occurring is based on the recognition of "controlling variables" which are assessed from field indicators (such as soil properties) and/or secondary data sources (such as information on flooding frequency/extent, fertilizer use, and grazing regime). The determination of function performance is based on the evaluation of one or more component processes underpinning functioning (Fig. 6; Maltby 2009).

The FAPS manual covers 38 hydrological, biogeochemical, and ecological processes contributing to 12 functions that may occur in wetlands. The assessment can

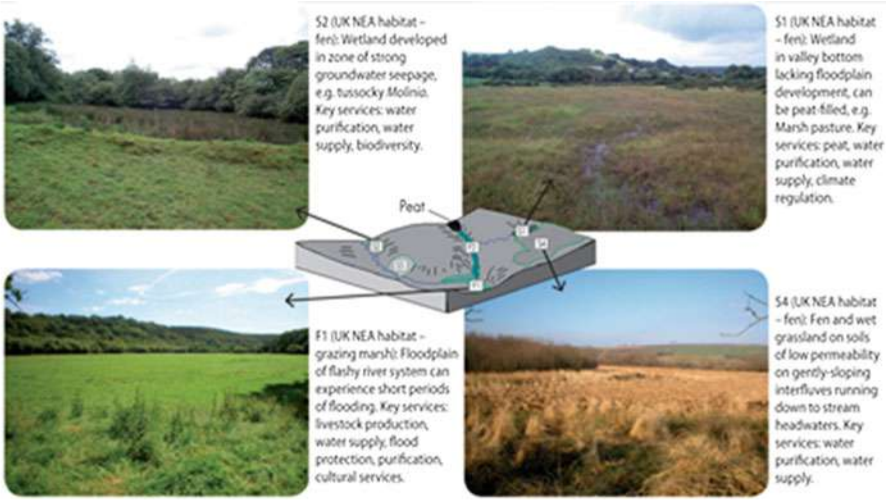


Fig. 4 Examples of functional units in wetlands at the landscape scale from the Tamar catchment (From Maltby et al. 2011; used with permission from UNEP-WCMC)

be carried out manually or by using a CD-based electronic version (Maltby 2009). The outputs can be used ultimately to determine the range of ecosystem services which result from functioning in any part or for all the wetland.

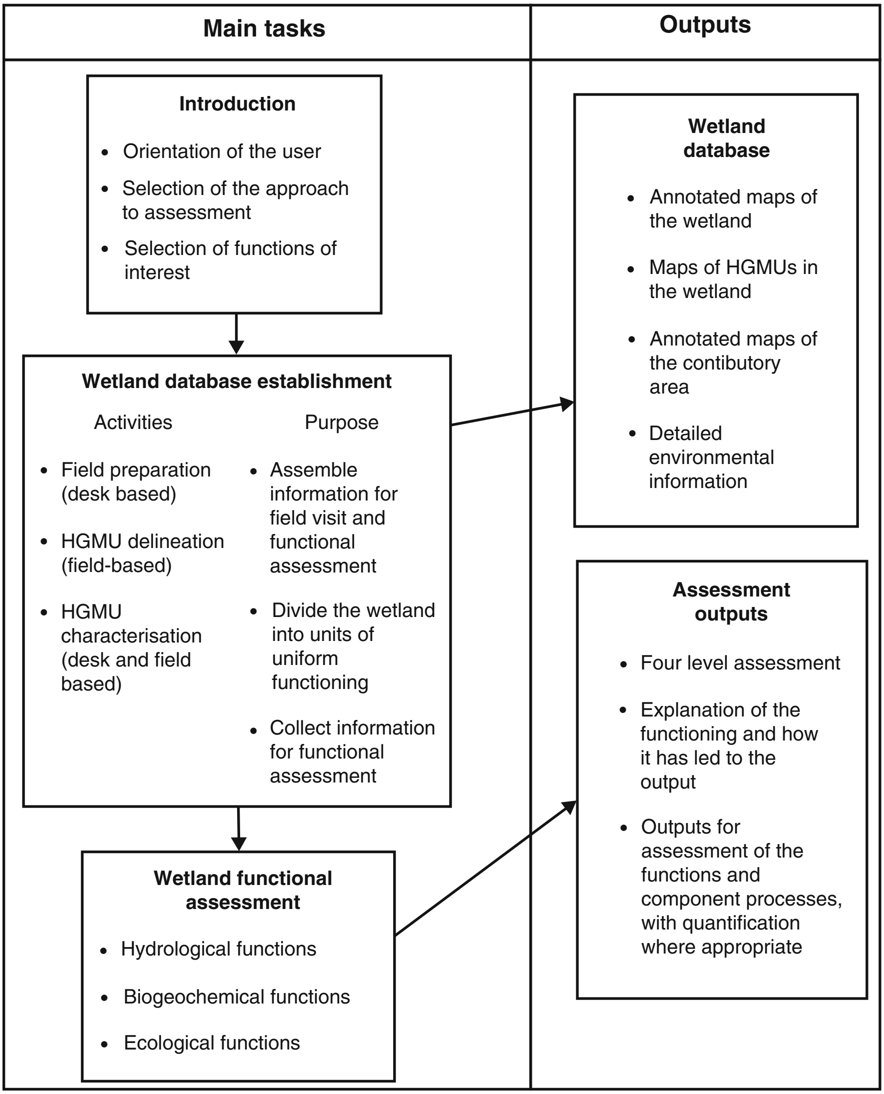
Applications of FAPS

The FAPS are structured to enable:

- Overall assessment of wetland functioning
- Assessment of one or more specific functions for part or the whole wetland
- Assessment of one or more individual processes underpinning functioning
- Evaluation of ecosystem services (and/or values) provided by the wetland

Evaluation of the method at three contrasting sites can be found in McInnes et al. (1998) while the Method Manual (Maltby 2009) itself contains examples at different scales. Rapinel (2012) has explored how the site-based FAPS can be extended to the river catchment scale in France using LIDAR data.

It is possible to manipulate the database entries to explore the possible effects of management and/or environmental changes, providing a tool to assist in scenario analysis and decision-making in relation to possible policy, socioeconomic, and climate changes. A prototype management support tool called the Wetland Evaluation Decision Support System (WEDSS) has been developed (Maltby 2009) but requires further testing for practical application.



Basic pathway through wetland functional assessment procedures.

Fig. 5 Pathway through wetland functional assessment procedures (FAPS) (Source: Maltby 2009; used with permission from Elsevier)

Future Challenges

Functional assessment methods have been developed in recognition of the fact that it is impossible to carry out the research at all wetland sites to determine functions empirically. Not all wetlands perform all functions or particular functions to the same

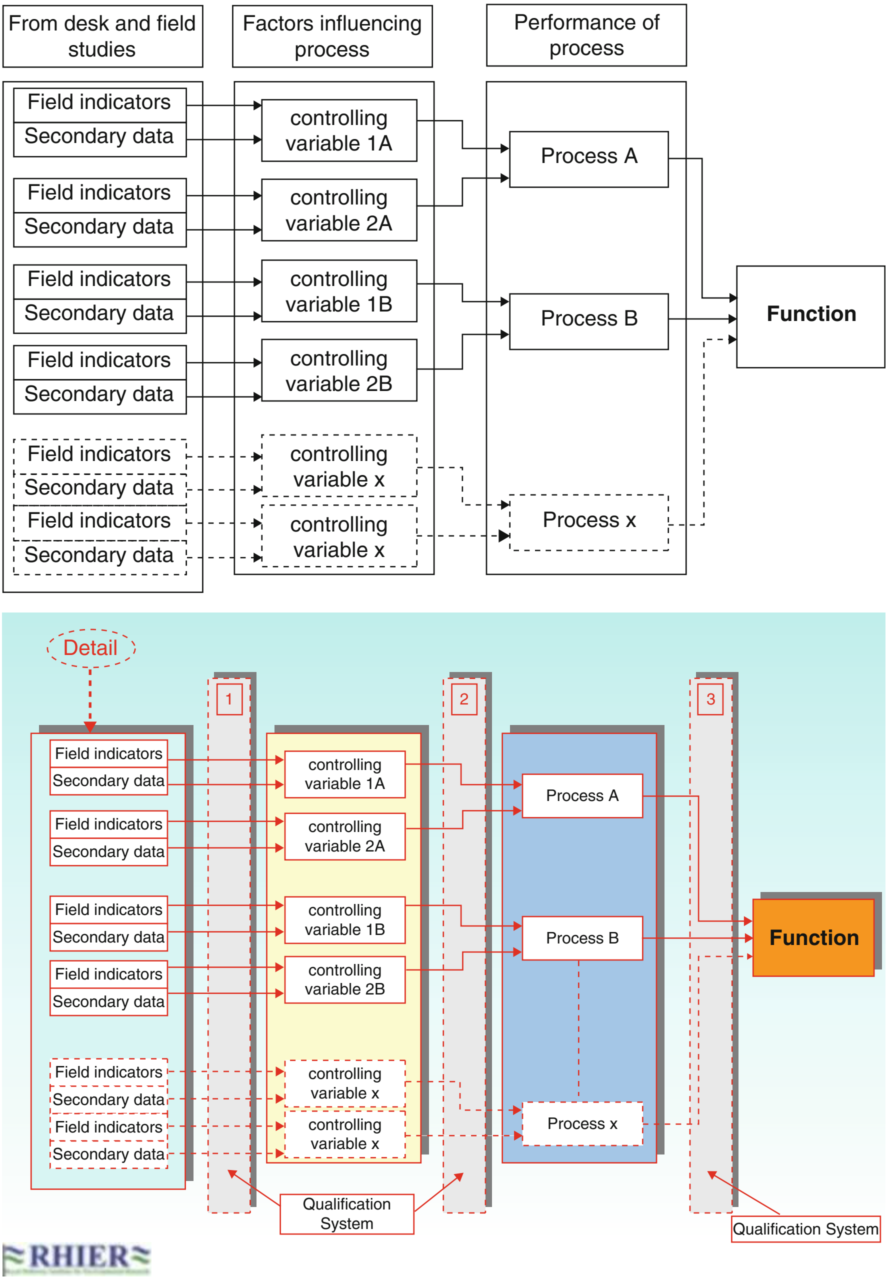


Fig. 6 Procedural steps in functional assessment (Source: Maltby 2009; used with permission from Elsevier)

extent or necessarily in the same landscape position or climate. The reliability of the assessment outcomes is dependent on the strength of the evidence base on which relationships are inferred and on the availability of minimal essential information about the wetland. There are still many gaps in the understanding of relationships between field and/or secondary data indicators, biophysical processes, and functioning. More important is the lack of consistent basic information, especially hydrological regime, for the greater part of the world's wetland resources. Lack of monitoring of wetland sites and very limited new empirical research is likely to be a persistent problem hindering the testing and verification of assessments at more than a limited number of sites. It is, however, essential to continue to test available methodologies to strengthen the confidence in their use and outcomes. Only in this way will it be possible to develop the tools necessary to support the policy shifts which are required to recognize, maintain, and where possible restore the natural capital of wetlands which is so valuable to human society.

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Hydrological Assessment and Monitoring of Wetlands

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Rob Low, Gareth Farr, Derek Clarke, and David Mould

Contents

Abstract

This section introduces ecohydrological conceptual models as a repository for knowledge about the combined ecological and hydrological functioning of a wetland and then provides a starting point (or initial framework) for the

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development of such a model. An ecohydrological conceptual model only needs to include critical elements and mechanisms, in only as much detail as is necessary, of the ecohydrological functioning of the wetland. The model must be recorded, for continuity of knowledge, through maps, diagrams, narrative description, and key data, such as water levels and vegetation surveys. The ecohydrological conceptual model should be continually tested against new data and information and revised and refined as necessary. Also introduced are hydro-environmental supporting conditions (HSCs), defined in terms of, for example, water levels, flow, or water chemistry, that are required to support wetland plant communities. A suite of techniques for ecohydrological investigation and characterization of wetlands are described. These include desk study, walkover surveys and site investigation and monitoring of wetland substrate, water levels, vegetation classification systems, and water chemistry. The results of HSCs can be used to develop and refine the ecohydrological conceptual model.

Keywords
Wetlands monitoring · Hydrology · Hydrogeology

Introduction

The physical and chemical characteristics which favour wetland plant communities, primarily high soil water levels and anaerobic soil chemistry, are related directly to the hydrology/hydrogeology of the wetland and often its surrounding catchment. Appreciation and successful management of a wetland therefore almost always requires an understanding of its hydrological functioning, including the influences on hydrological functioning which often lie beyond the designated boundary of the site.

This section introduces **ecohydrological conceptual models** as a repository for knowledge about the combined ecological and hydrological functioning of a wetland and then provides a starting point (or initial framework) for the development of such a model. Also introduced are **hydro-environmental supporting conditions** (HSCs) that allow us to describe specific hydrological conditions required to support wetland plant communities. A suite of techniques for ecohydrological investigation and characterization of wetlands are described, the results from which can be used to develop and refine the ecohydrological conceptual model.

Ecohydrological Conceptual Models

An important requisite for appreciation and successful management of any system is a sufficiently detailed understanding of the relevant aspects of its form and function. For example, mechanics must use their basic knowledge of the form and function of a car engine in order to diagnose and remedy faults. The same is true in relation to appreciation and management of the hydrological environment, and specifically in this context wetland hydrology, where an understanding of the hydrological form and function of a wetland is called a **conceptual model**. And since the work is at the interface between ecology and hydrology, it is often called an **ecohydrological conceptual model**.

Some of the key characteristics of an ecohydrological conceptual model are:

- It only needs to include critical elements and mechanisms, in only as much detail as is necessary, of the ecohydrological functioning of the wetland; only a sufficient understanding of the complexity of a natural system is required.
- It must be recorded, for continuity of knowledge, through maps, diagrams, narrative description, and key data, such as water levels and vegetation surveys (Fig. 1).
- The ecohydrological conceptual model should be continually tested against new data and information and revised and refined as necessary.

As a starting point for an ecohydrological conceptual model, it is useful to identify the mechanisms of **water supply** to the wetland, **water retention** within the wetland, and **water loss** from the wetland. All wetlands will have at least one mechanism under each of these headings, and this approach offers a useful initial framework for an ecohydrological conceptual model.

There are four primary mechanisms of **water supply** to a wetland:

1. Rainfall. Rainfall is the primary source of water for ombrotrophic systems, i.e., bogs, and is characterized by low pH and a low dissolved mineral content. Atmospheric deposition is the key pathway allowing nutrients such as nitrogen to enter ombrotrophic systems. Temporal variation of rainfall (short term, seasonal, longer term) can be an important overall determinant of wetland water levels.
2. Surface water. Water from streams and rivers, including seasonal surface water flooding, is an important supply for many floodplain wetlands. Flow characteristics and natural water quality are variable depending partly on the environmental factors within the catchment (e.g., topography, geology) and also the anthropogenic pressures (e.g., land use, drainage, and water abstraction). Suspended sediment load and deposition can be an important aspect of surface water supply.
3. Groundwater. Permeable aquifers can be important as a supply to wetlands, but ecohydrologically significant quantities of groundwater can emerge from most rocks or superficial deposits. The character of a groundwater discharge – rate, variability of flow, and water chemistry – is primarily determined by the nature of the rocks and sediments through which the groundwater flows.
4. Surface runoff. Direct surface-borne flow occurring when rainfall exceeds the surface infiltration capacity within the immediate surface water catchment of the wetland. The spatial distribution of surface runoff is dependent on micro-topographic routing on surrounding slopes and can be redirected by boundary drainage systems. In agricultural catchments surface runoff from adjacent fields can often be the source of nutrient-enriched water.

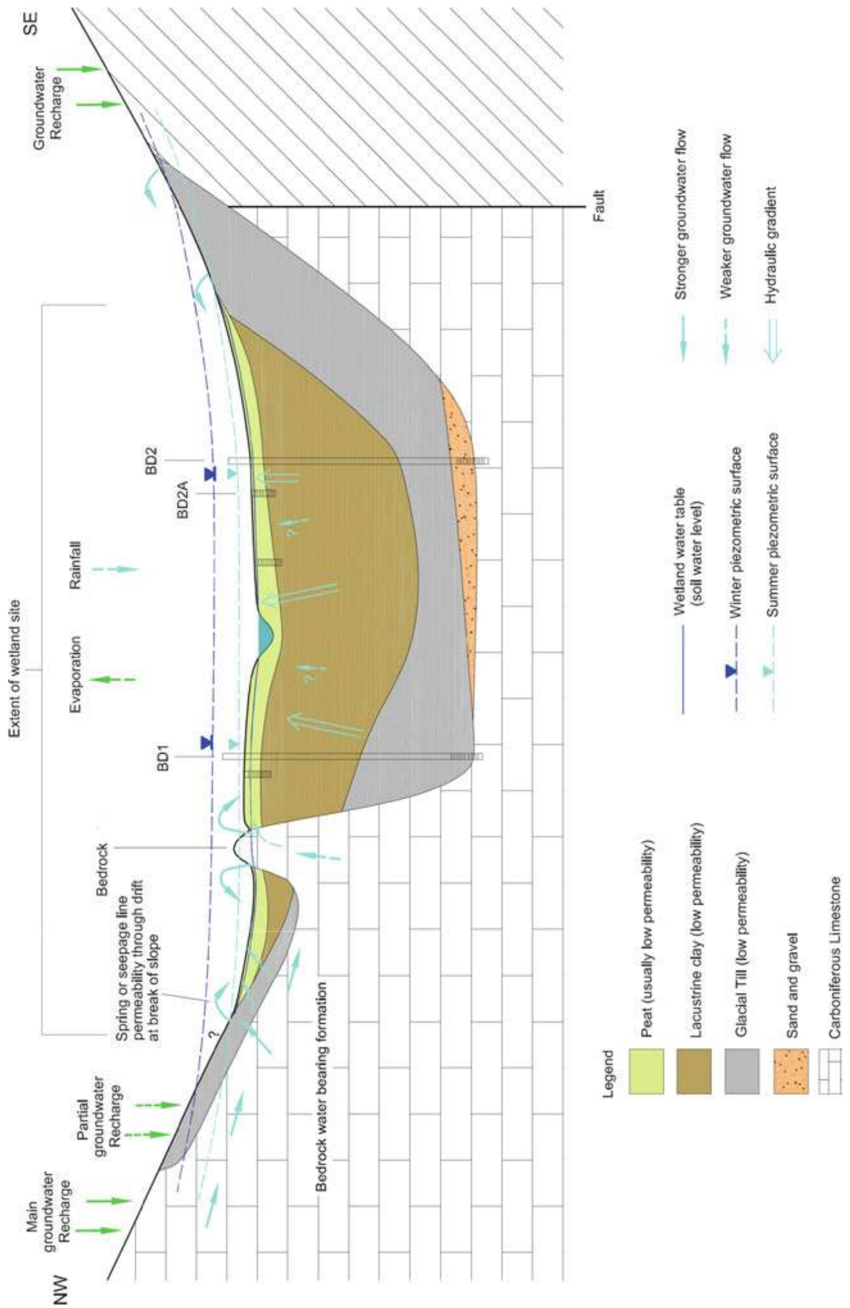

Fig. 1 Diagrammatic example of an ecohydrological conceptual model from Cors Bodeilio, part of the Anglesey and Llyn Fens NNR, Wales, UK (Schlumberger Water Services 2010)

Water retention within wetlands is caused primarily by the combination of low topographic (and hydraulic) gradients; the presence of poorly permeable deposits such as silt, clay, and peat; and the presence of water-retaining vegetation (e.g., *Sphagnum* mosses). Wetlands on steeper slopes also depend on a continuous supply of water to maintain wetness (soligenous systems). The mechanisms of water retention often only become apparent when they are compromised, for example, by the presence and effects of ditches that bypass the water retention mechanisms and accelerate water flow through a site.

There are three primary mechanisms of **water loss** from a wetland:

1. Evapotranspiration. The combination of direct evaporation from open water and soil surfaces and transpiration from plants. Evapotranspiration is often the predominant cause of lowered wetland soil water levels in the UK during warmer months.
2. Surface water. Discharge to streams and rivers flowing from or through wetlands.
3. Groundwater. In a natural condition, water loss to groundwater is relatively rare because of the landscape position and poorly permeable basal deposits associated with many wetlands.

Hydro-environmental Supporting Conditions

It is useful here to introduce the concept of hydro-environmental supporting conditions (HSCs). The term "hydro-environmental," rather than just "hydrological," is used to acknowledge the vital interaction of water with other environmental factors, such as geology, wetland substrate, and micro-topography, in producing favorable conditions. HSCs are the specific hydrological conditions, defined in terms of, for example, water levels, flow, or water chemistry, which are required to support a wetland plant community. At a basic level, HSCs are obvious – near-surface water levels are a requisite for peat-forming wetlands, and a base-rich groundwater supply is required for alkaline and calcareous fens. At a more detailed level, information on HSCs for many wetland plant communities can be obtained for, for example, lowland wetland communities (Environment Agency 2010), wet grassland communities (Gowing et al. 2002; see Box 1), and wet woodland communities (Barsoum et al. 2005). Since the recognition and application of HSCs is a relatively new and complex subject, the information in these sources is often incomplete and/or uncertain, and judgment based on experience is often required to determine and use HSCs.

Box 1 Hydro-environmental Supporting Conditions for Wet Grassland Communities (Gowing et al. 2002)

A very good example of the determination of HSCs for wetland plant communities is provided by the work of Professor David Gowing and others on wet grassland communities – it is based on extensive botanical and hydrological data collection at 18 sites throughout England. Two metrics were chosen to describe a wet grassland water level regime – sum exceedance value (SEV)

(continued)

Box 1 Hydro-environmental Supporting Conditions for Wet Grassland Communities (Gowing et al. 2002) (continued)
for soil drying and SEV for soil waterlogging. The method relies on threshold water levels being specified, one defining the level at which the zone of densest rooting begins to become waterlogged and the other defining the level at which drying of the surface soil becomes detectable by plants. For each threshold, the SEV is the depth-time integration when the water table is above or below the threshold value, with waterlogging only being integrated between March and September, during the period of active grass growth – the concept is illustrated in the graph Fig. 6.

When data for the two SEVs (5-year means) were plotted against each other (Fig. 7), it was found that the water level regimes for wet grassland communities were distinct, suggesting that the water level regime is an important, perhaps the most important, determinant of plant community composition. The water level regime information is therefore very useful for management of wet grassland sites.

An important criterion for recognition of successful hydrological management of a wetland is therefore the presence of favorable spatial and temporal distributions of HSCs for the target wetland plant communities. It follows that an ecohydrological conceptual model should include an understanding of the processes which combine to produce these HSCs at critical times and places within the site; these processes are likely to act at a variety of scales, both within and outside the wetland. An appropriately detailed understanding of these processes should aid identification of causes of hydrological problems within a wetland that could result in unfavorable wetland condition.

Understanding the hydrological functioning of most wetlands is easier when equipped with a basic understanding of groundwater flow theory, including principles such as hydraulic head, hydraulic gradient, hydraulic conductivity, and Darcy's law, as described in any basic hydrogeology textbook (e.g., Price 1996; Hiscock 2006). It is also very useful to have an understanding of the wider-scale environmental water cycle because hydrological conditions within a wetland are often significantly influenced by processes operating beyond the site boundary.

Developing and Refining the Ecohydrological Conceptual Model

Desk Study

Many investigations will start with the collation and review of existing information, often in the form of a short written report. This is called a "desk study" and is an important first step toward gathering the information needed to develop a wetland ecohydrological conceptual model. The desk study should be undertaken

Table 1 Desk study sources of data and potential uses to inform a hydroecological conceptual model

Source of data	Example of data and use
Site managers and local experts	The range and depth of information that can be gained from site managers and local experts can be an important starting point for the desk study
Published literature	Peer-reviewed literature
Grey literature	Site descriptions and reports and notes on the ecology, hydrology, management, and historic and current pressures
Vegetation survey	Vegetation maps (e.g., those produced using the National Vegetation Classification or NVC in Britain) will provide a baseline from which to monitor change. For example, certain communities that are more groundwater dependent maybe used to indicate areas where groundwater is an important HSC
Geological maps	Bedrock and superficial geology can be used to characterize the wetland, and if detailed enough, a geological cross section may be produced
Borehole archives	Stratigraphical data that can be used to create geological cross sections of the wetland (as above) and to understand the depositional history of the wetland
Soil maps	Soil map and properties
Water chemistry	Nutrient levels, e.g., nitrate and phosphate, ions, and physical parameters such as pH, dissolved oxygen, and electrical conductivity are all part of characterizing HSC at any wetland
Rainfall	Rainfall data from an on-site or local monitoring point may also include other climatic variables such as wind speed, moisture, and sunshine
Groundwater level and chemistry data	Existing boreholes installed with hydrometric monitoring could provide information on the local or wider supporting aquifer/s
Groundwater models or maps	Groundwater flow direction and catchment-scale conceptual model
Aerial photographs	Historical and current hard copy or digital photographs ideal for seeing land use and vegetation changes
Air pollution information	Modeled or measured deposition of atmospheric nutrient loading

in advance of any new information being collected or before fieldwork or a walkover survey is undertaken. Table 1 lays out a step-by-step list of sources of information to be included within the desk study phase. This information will be vital to support the ecohydrological conceptual model and to help identify potential HSCs.

Walkover Survey

There is a limit to the conceptual understanding that any desk-based assessment can provide, and following the desk study, it is almost always important that a site visit or

site walkover survey is undertaken. As the realm of ecohydrological investigations and the identification of HSCs are still relatively modern, it is advisable to involve both an ecologist and a hydro(geo)logist in the walkover survey; this cross-disciplinary approach is a theme that runs throughout the entire ecohydrological conceptual modeling process.

The walkover survey facilitates collaboration between the ecologist and the hydro (geo)logist, and basic data such as observations on the presence, levels, and flows of water in relation to key wetland vegetation communities can be recorded. The survey provides a platform for the discussion of ideas, thoughts, and theories that can support the understanding of HSCs, can underpin detailed site investigation, and ultimately can enable further development of the ecohydrological conceptual model.

Before engagement with the site at a detailed level, the position of the wetland within the wider landscape should be considered. What generic type of wetland is under consideration (bog? fen?), what are the related wetland plant communities, and what are the likely HSCs? Is the site likely to be supported by rainfall, groundwater, surface water, or a combination thereon? What are the main water retention and loss mechanisms? Does the immediate catchment or landscape setting suggest potential anthropogenic pressures such as agriculture, urban development, or industry?

During the walkover survey, the hydro(geo)logist might consider questions such as: What are the main water supplies to the site? How is water retained within the site, and how can water be lost from the site? Looking for evidence of groundwater inflow to the site (springs, seepages, etc.), estimates of flow can be made, and basic field parameters (pH, electrical conductivity, and dissolved oxygen) can be collected. Hand augers can provide an affordable, quick, and easy way to characterize the near subsurface in both geological and hydrogeological terms, informing potential locations for dipwells to monitor groundwater levels and chemistry.

An ecologist will often be able to provide surrogate hydrological evidence whereby the presence and condition of certain plant communities or species can be used to infer the existence of certain hydrological mechanisms, regimes, and conditions (HSCs). For example, the presence of ombrotrophic vegetation in central areas of a fen will suggest that rainfall is the predominant source of water in those areas, which in turn has implications for the interpretation of water flow through the site.

Site Investigation and Monitoring

Information requirements from site investigations will vary between wetland sites and will be influenced by the results of the cross-disciplinary desk study and the walkover survey. Figure 2 represents a generic wetland with some common pressures on the right-hand side (e.g., groundwater abstraction, nutrient enrichment, and drainage) with the left-hand side illustrating a selection of the more common wetland monitoring techniques.

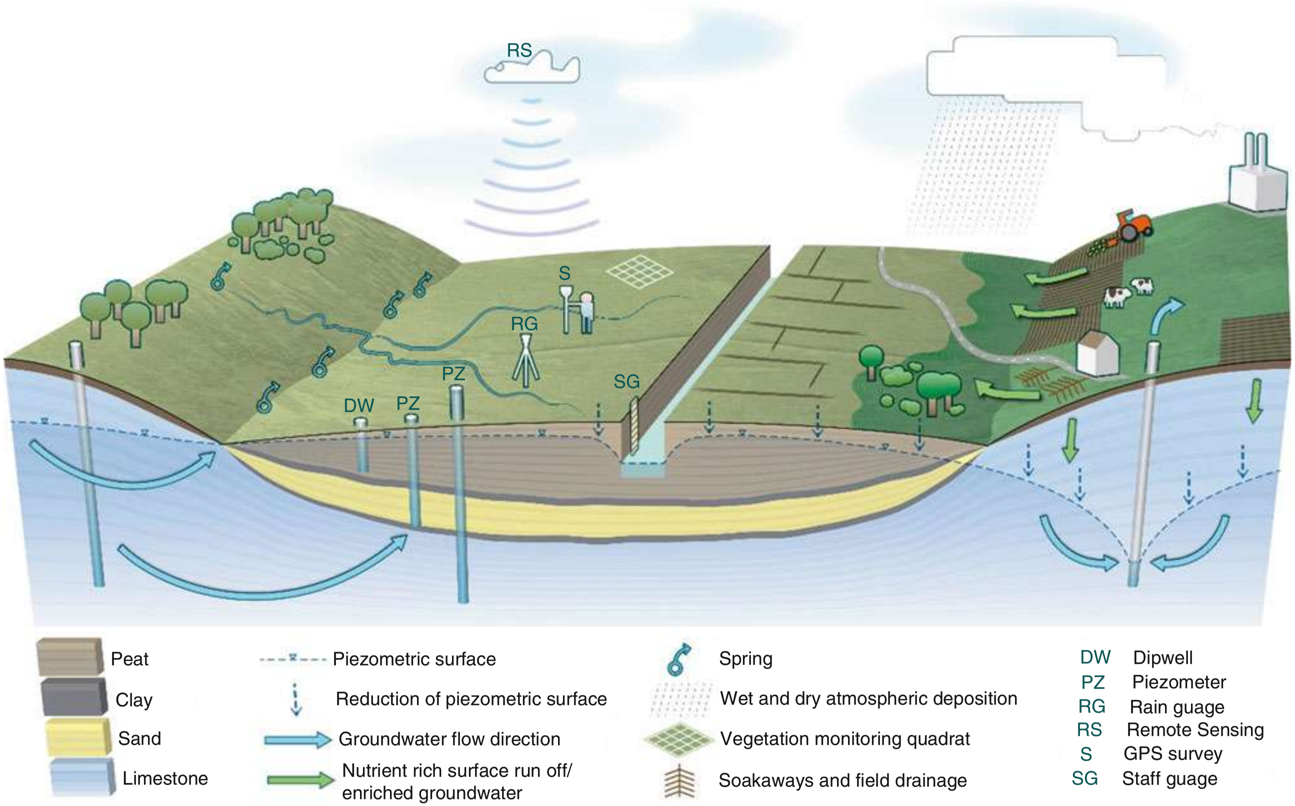

Fig. 2 Generic pressures (*right-hand side*) and common site investigation techniques for wetlands (*left-hand side*) (Image: BGS© NERC)

Wetland Substrate

It is important to survey the wetland substrate and to compare it with any published geological and soil maps; the distribution and nature of substrate types are important determinants of hydrological conditions within a wetland.

Peat probes are an inexpensive tool for determining the thickness of peat deposits. If you suspect there is a thickness of peat then a peat probe – basically a long thin rod – can be pushed through until a more resistive material, such as gravel or clay, is encountered. Peat probes can be used safely to prove peat deposits up to around 6 m in thickness and sometimes more. Repeat measurements across a site can quickly result in an understanding of the thickness and lateral continuity of a peat deposit and also the general shape of the underlying mineral surface, which in turn often allows the history of peat accumulation to be inferred.

Hand augers are another low-cost method to characterize near-surface superficial deposits. The operational depth of a hand auger depends upon the material and the strength of the user, but retrieval of material from greater than 2 m depth can often be challenging! The auger head will retrieve about 20 cm of material for each insertion, allowing the user to create a geological log of the near-surface deposits.

Drill rigs are a more expensive option; however, they can offer deeper investigation, retrieval of sediment/bedrock cores, and the option of installation of monitoring wells. The most common issue with drilling at wetland sites will be arranging safe access for the drill rig and avoiding causing damage to any interest features.

Geophysical methods such as ground-penetrating radar and electrical resistivity can be used as nonintrusive methods to characterize large areas of the wetland substrate. Geophysics can be expensive but can also help inform suitable areas for the installation of monitoring wells. Geophysical data can also be collected from airborne surveys, although the cost of this is significant and airborne surveys are often used to look at landscape-scale rather than site-scale detail. Beamish and Farr (2013) show that airborne geophysics can be useful to help characterize wetlands on a landscape scale, potentially helping to guide ground investigations. The attenuation of airborne radiometric data can identify areas of water saturation near the surface, while conductivity data appears capable of mapping the occurrence of clay concealed beneath peat.

Vegetation Classification Systems

In Britain, a common standard of vegetation classification, called the National Vegetation Classification (NVC), is used (see Rodwell 2006). The NVC was the product of a commissioned research project in 1975 funded by the Nature Conservancy Council (NCC), designed to be used by all the conservation bodies in Britain, allowing comparable datasets to be gathered and compared for similar plant communities. NVC data will not exist for all wetlands; it can be timely and costly to collect over entire wetland sites. Where however it does exist, it can offer useful information with respect to development of the ecohydrological conceptual model. The presence of many NVC communities can be used to infer the presence of specific HSCs, as noted above.

A large number of countries have a similar vegetation classification system which defines vegetation associations based variously on floristic, ecological, and physiognomic criteria, for example, the Canadian and US vegetation classification systems; mapping of wetlands according to these types of systems will provide similar useful information in relation to development of conceptual models.

Water Levels

Water level with respect to the ground surface is often a key parameter describing HSCs for wetlands and will form an important part of any ecohydrological site investigation and conceptual model. It is worth giving careful consideration as to when and where water level data will be collected. Firstly, information gathered from the desk study should be consulted and used in conjunction with on-site ground condition and vegetation data. Discussion between the hydrologist and ecologist should be undertaken to ensure the water level monitoring data informs both the HSCs and also the hydrology near or within key vegetation areas. Siting of water level monitoring points next to important areas of vegetation or where repeat vegetation surveys occur will only increase the value of both of these datasets. The period of water level monitoring is also an important consideration if, for example, hydrological extremes such as drought and flood are to be recorded or if changes in vegetation linked to changing HSCs are to be identified. Data from monitoring periods of less than 1 year are often limited in their usefulness, and long-term monitoring periods of several years or longer may be required to produce meaningful datasets.

The data set shown above (Fig. 3) is the monthly water table variation over a 40 year period in an undisturbed coastal sand dune system in the UK. Over the whole period, there is no long-term upward trend (which might be due to sea level rise) or decline (which might be due to higher temperatures/climatic change). However there are significant inter-decadal changes. Take, for example, the period 1972–1982, which shows a definite and continuous increase in water table levels. Such a 10 year data set will produce a statistically significant positive upward trend. However the next 10 years (1983–1993) shows the direct opposite – a statistically significant downward trend. A similar sequence is also apparent after 2000, when increased awareness of climate change might make us state that this is definite proof of climate change, if we did not have the proceeding 30 years of data.

Short-term sudden changes in water level may be relatively easy to identify, and their cause may be readily found, such as increased well pumping or raised reservoir water levels. Changes in groundwater level over several months or years are more difficult to explain – is there a slow but gradual change in the rainfall pattern, is the land use (hence evapotranspiration) changing, or are there slow long-term mechanisms such as sea level rise in play?

In reality, a calibrated aquifer recharge model exists for the system presented in Fig. 3 (Clarke and Sanitwong 2010). The model shows that interannual variability of rainfall is the main driver of these changes and no definite climate change signal is apparent. The lesson learned here is that changes in the medium term (5–10 years) should not be used to prove or demonstrate the influence of a single driver of change.

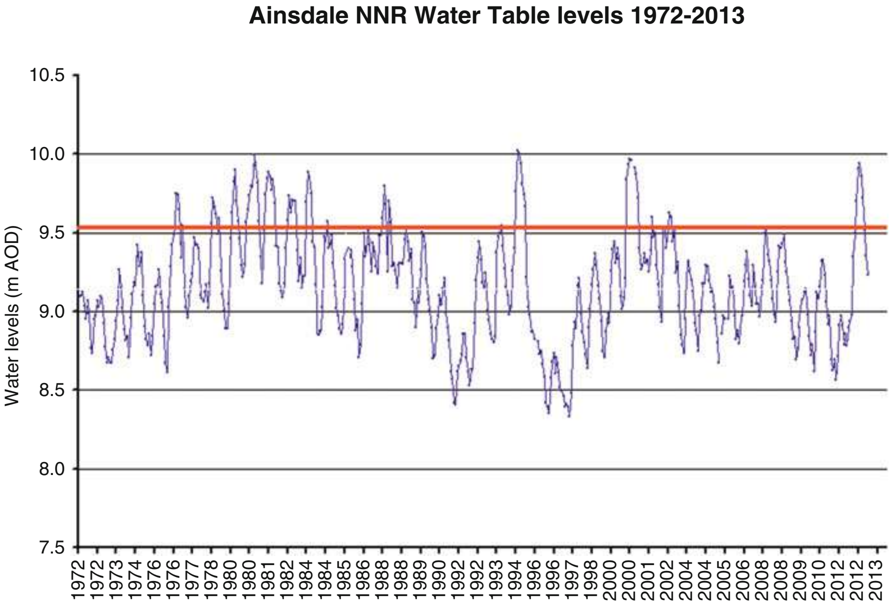

Fig. 3 Long-term groundwater level record from well 11 at Ainsdale Sand Dunes National Nature Reserve, Merseyside, UK

Climate change is a slow and incremental process and is usually described within time steps of 30+ years, and during this period, natural variability may be an order or magnitude or more than any climate change influence.

When monitoring water levels at a wetland, it is most likely that you will also want to monitor surface water, such as a ditch, pond or pool, or soil or groundwater. Measurements can be made manually or electronically with an in situ water pressure data logger. Surface water levels can be monitored using similar techniques to groundwater levels.

Techniques for monitoring groundwater levels include (Fig. 4):

Dipwells are inexpensive plastic tubes, ranging in diameter from 12 to 50 mm, with holes or slots to allow water ingress. They can be installed manually using a hand auger, usually to between 1 and 3 m depth. Dipwells can come with a variety of geotextile membrane covers which can be selected based upon the sediment into which they are being installed. Finer geotextile membrane covers are used where there are fine-grained sediments such as silts or fine sands to limit the ingress of this material into the dipwell as much as possible. When dipwells are installed in peat, they are sometimes attached to a tube driven into competent underlying material such as a basal clay or bedrock, to allow the vertical position of the dipwell to be maintained. Water levels and soil water levels can be measured in dipwells by using an electronic water level dip tape or by installation of an electronic water pressure data logger.

Fig. 4 Dipwells can be manually installed (*left*), deeper dipwells and piezometers can be installed into more competent material using portable hand-powered drills (*middle*), and larger percussion drills are used to install deeper boreholes and piezometers into bedrock

Boreholes are drilled using a large drill rig, of which there are various types; the scale of effort, cost, and the installations themselves are much larger than for dipwells. In the context of wetland investigations, boreholes are normally drilled to a maximum of 10 m, and depending on the drilling technique, the materials underlying the wetland can be examined and recorded with reasonable precision. A borehole can be completed either with slotted tube throughout its depth, allowing water ingress at all levels, or with slotted tube at a specific depth (usually the base), allowing water ingress and pressure measurement at that depth. The latter are called piezometers because they measure subsurface water pressure (piezometric pressure). Combinations of dipwells and piezometers can be used to help to characterize vertical hydraulic gradients, which indicate the potential for vertical flows of water, e.g., upwelling into wetland sites.

Survey and construction data should be recorded for each monitoring well. All dipwells, piezometers and boreholes should be surveyed to a common datum to allow the comparison of data from one well to another. This datum can be an arbitrary fixed point within or close to the site (a local benchmark) or if possible ordnance datum (OD) or sea level. It is vital that a borehole log is made for each well and should include the type and thickness of strata encountered, if possible recorded in line with an international standard or description. Borehole logs should also include survey elevation data, notes on the decisions made to install them in any given location, and notes on the vegetation or habitat they are associated with.

Instrumentation and Frequency of Data Recording

Regular manual water level recording is recommended at all wells to correct any data collected from in situ electronic pressure transducers. There are many proprietary pressure transducer systems on the market, and one should be selected based upon the water column (pressure) range, accuracy, and resolution that is required. For simplicity, it is recommended that all loggers should be set to record coincidentally and that they are set to run on standard time (e.g., GMT in the UK). The frequency of data measurement and recording should be decided according to the purpose of monitoring; a higher frequency (e.g., 15 or 30 min interval) yields data which will provide information about the short-term dynamic functioning of the system, whereas a lower frequency can be used for background monitoring.

Figures 5, 6, and 7 show an example of the increasing information obtained from higher-frequency sampling. This shows the shallow groundwater response to rainfall at a wetland site with constant groundwater recharge and a diurnal water table fluctuation driven by evapotranspiration. Note that the monthly and weekly sample rates do not pick up the rainfall events and that sub-daily (in this case hourly) sampling is needed to detect the diurnal pattern. Monitoring at a suitably high temporal resolution can allow estimation of evaporative loss (e.g., Gilman 1994; Mould et al. 2010) during periods of zero rainfall, when lateral flow is constant and evaporative loss is enough to drive a diurnal oscillation in water levels (assuming constant lateral or upward shallow groundwater flow). However producing unnecessarily large datasets can be problematic when information storage and analysis are considered. So a monitoring program should consider the cost versus benefit of monitoring frequency.

Water Chemistry

Water chemistry or quality is often, in conjunction with water levels, a key HSC for many wetland plant communities, e.g., alkaline and calcareous fens both depend on specific groundwater chemistries. An understanding of baseline chemistry and variation in chemistry through time are key to identifying risks (such as nutrient enrichment) and to underpin successful management of wetlands and wider catchments. For all sampling, a repeatable and defensible **methodology** should be implemented following best practice procedures. A **comparative analysis suite** should be used for all wetland investigations with agreed lower limits of detection for nutrients and sufficient ions to characterize groundwater facies or types. Table 2 shows the minimum analysis suite that has been agreed by the Water Framework Directive UK Technical Advisory Group (WFDUKTAG) for wetlands, the aim of which is to make results, especially for nutrients, directly comparable within the UK (WFDUKTAG 2004).

Where possible each sample point must be put "**in context**" which means it should be associated with a specific vegetation type or habitat, and a reason for its inclusion should be recorded. This information may already exist on the borehole log as described in the previous section.

In any wetland investigation, water chemistry samples may be obtained from surface water (e.g., ditches, ponds, runnels) and groundwater (e.g., seepages,

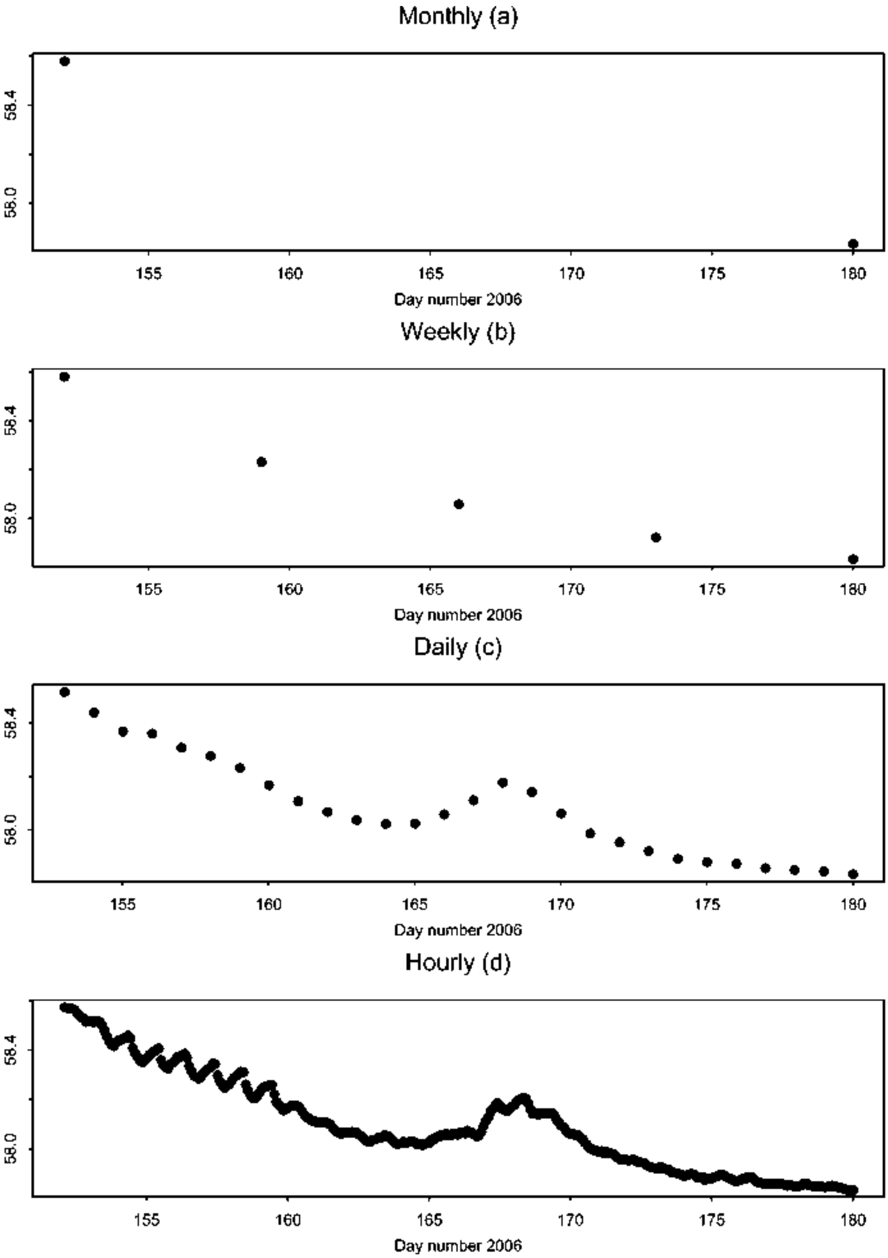

Fig. 5 Water level monitoring data from Otmoor, UK. The same data set is shown at four distinct sampling frequencies, with detail increasing as frequency increases

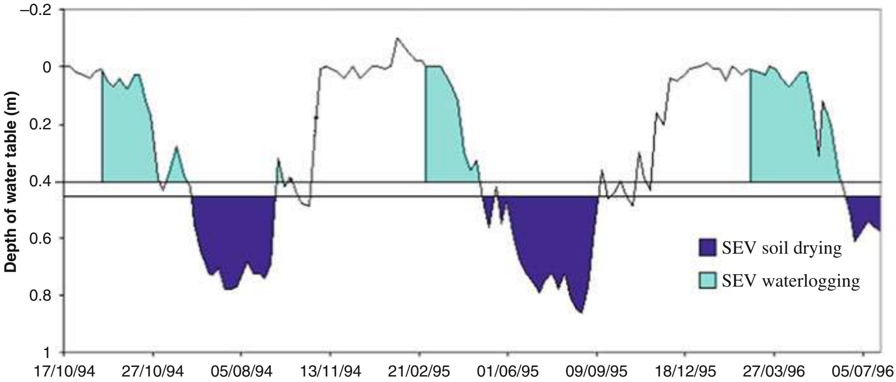

Fig. 6 Time series water levels relative to the ground surface. The *shaded areas* demonstrate how the SEV areas for soil drying and wetting are defined

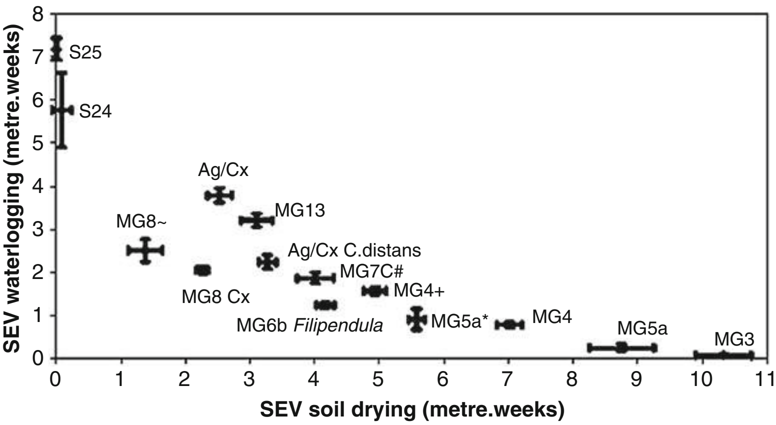

Fig. 7 SEV soil drying versus SEV soil waterlogging. The labeled points within the graph space represent separate wet grassland NVC communities

springs, dipwells), each of which can pose their own difficulties. Low or diffuse flows in many wetland areas can be problematic but not prohibitive to sample. Simple tools such as a stainless steel jug (and some patience) will often allow samples to be collected from even the smallest of runnels or seepages. Portable low voltage submersible pumps can abstract water from dipwells, and syringes can be used to sample water from small ponded areas or seepages. Field readings for pH, temperature, dissolved oxygen, and electrical conductivity need to be recorded on site using appropriate methods. Wherever possible, measurements should be taken of flowing water, with time allowed for instrument stabilization.

Table 2 Wetland water chemistry analysis suite (WFDUKTAG 2004)

Determinant	Units	Suggested minimum limit of detection
Alkalinity pH 4.5 – $CaCO_3$	mg/l	5 mg/l
Ammonia – N	mg/l	0.03 mg/l
Bicarbonate – HCO_3	mg/l	n/a
Calcium – Ca	mg/l	1 mg/l
Chloride ion – Cl	mg/l	1 mg/l
Conductivity at 25 °C	uS/cm	n/a
Hardness total – $CaCO_3$	mg/l	n/a
Iron – Fe	ug/l	30 μg/l
Magnesium – Mg	mg/l	0.3 mg/l
Manganese – Mn	ug/l	10 μg/l
Nitrate – N	mg/l	n/a
Nitrite – N	mg/l	0.004 mg/l
Nitrogen total oxidized – N	mg/l	0.2 mg/l
Orthophosphate – P	mg/l	0.02 mg/l
Oxygen dissolved – field measurement	mg/l	n/a
Oxygen dissolved – field measurement	%	n/a
pH – field measurement	pH	n/a
Phosphate	mg/l	0.02 mg/l
Potassium – K	mg/l	0.1 mg/l
Sodium – Na	mg/l	2 mg/l
Sulphate – SO_4	mg/l	10 mg/l
Temperature – field measurement	CEL	n/a
Redox potential – field measurement	Mv	n/a
Iron dissolved	ug/l	n/a
Manganese dissolved	ug/l	n/a

Novel Techniques

Novel groundwater analysis can help to characterize the HSCs at wetlands and to improve the ecohydrological conceptual model. It is possible to understand the recharge age of groundwater using several dating techniques. One technique that is applicable to wetlands, where waters are often less than 50 years old, is the dating of **chlorofluorocarbon (CFC) and sulphur hexafluoride (SF_6)** aerosols. This analysis can also help to infer groundwater mixing and likely groundwater flow mechanisms (Gooddy et al. 2006).

When a wetland is faced with problems of enrichment by nitrogen, then it is possible to use **nitrogen and oxygen stable isotopes**, often in conjunction with other analysis, to determine the source of nitrogen dissolved in groundwater (Saccon et al. 2013). The method works by comparing the ratios of the respective isotopes, ^{15}N to that of air ($\delta\ ^{15}N$ ‰) and ^{18}O relative to Vienna Standard Mean Ocean Water ($\delta\ ^{18}O$ ‰). The analysis can help to "fingerprint" various sources of nitrogen, including soil organic matter, inorganic fertilizers, and atmospheric deposition.

Future Challenges

Ecohydrology is an expanding subject in the UK, and it is an example of a subject where a truly bi- or multidisciplinary approach can pay significant dividends and is in fact essential. There are few people who have the complementary skill sets and knowledge to work alone and effectively in this field, and positive collaborations are therefore required. Appropriate education and training to provide ecohydrologists is encouraged.

More widespread monitoring and collation of ecohydrological data for wetlands, according to the guidelines above and more detailed sources, will allow the hydrological functioning of wetlands to be understood more clearly at both site-specific and generic levels and will also give information for characterization of HSCs. In turn, this will allow better wetland hydrological management.

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Wetland Assessment Methods: Integrated Assessment

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Abstract

It is recognised that wetland habitats have the potential to provide a wide range of useful functions that support human wellbeing. In order to meet the combined needs of habitat conservation and ecosystem service delivery, it is necessary to consider wetlands in a broad context. Integrated assessment aims to develop and address multidisciplinary questions through coordinated surveys of biodiversity, livelihoods and economic valuation. Interactions between aspects are identified and assessment reporting is integrated where possible.

Keywords

Integrated assessment · Management · Environmental indicators

Introduction

Wetland habitats are capable of carrying out a wide range of biophysical functions which can in turn provide benefits to people, supporting communities and their livelihoods. Integrated assessment aims to consider all aspects of a wetland and can provide the information and evidence necessary to enhance and maximise benefits and support the effective inclusion of wetlands in regional policy and planning. By comparison, assessments that focus on single aspects of a wetland may not capture the relationships between physical processes and community value and therefore fail to identify important cause and effect relationships. A wetland integrated assessment should be carried out at a spatial and temporal scale appropriate to the interactions between a wetland and its surroundings.

Integrated Assessment: The Concept

The concept of integrated assessment recognises that humans are part of ecosystems and that human activities, needs and desires must be considered in order for the management of natural resources to be practical and sustainable (Brooks et al. 2006). The broad range of ecosystem services that wetland systems provide is well documented (e.g. Millennium Ecosystem Assessment, 2005). Understanding how society is affected by these services and how they respond to natural and anthropogenic drivers across space and time is a fundamental aspect of integrated assessment.

Rotmans et al. (1996) define integrated assessment as “an interdisciplinary process of combining, interpreting and communicating knowledge from diverse scientific disciplines in such a way that the whole cause-effect chain of a problem can be evaluated from a synoptic perspective with two characteristics:

- Integrated assessment should have value-added compared to disciplinary oriented assessment.
- Integrated assessment should provide useful information to decision makers.

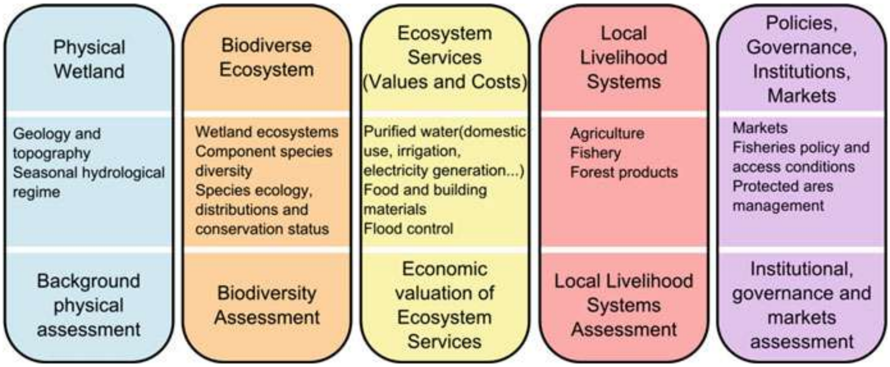

Fig. 1 The interlinked aspects of a wetland landscape, example properties, and corresponding assessment for each (Figure created based on a concept in Springate-Baginksi et al. 2009)

Types of Integrated Assessment

Although formal integrated assessment has specific meaning as an assessment that includes all relevant aspects of a wetland landscape (Fig. 1), the term has also been used to describe assessments that integrate a subset of those aspects. An example is the Index of Biotic Integrity (IBI) that integrates a set of biological indicator variables and assigns them a set of numerical scores based on comparison with an undisturbed reference site. The goal of IBI is to measure the ecological integrity (condition) of a wetland, with integrity being defined as deviation or lack thereof from regional reference (least affected) conditions (Mack et al. 2004). The resulting assessment indicates whether the species present are typical of a disturbed or undisturbed site and is useful in reporting condition and diagnosing the types of stresses that a system is under.

Another example method is Ecosystem Health Assessment, which may be defined as the capacity for maintaining biological structure on the one hand and the ability to achieve reasonable and sustainable human goals on the other. From this perspective, ecosystem health is as much about sustaining human communities, economic opportunity, and human and animal health, as it is about sustaining the biological functions of ecosystems (Rapport et al. 2001). The three attributes central to ecosystem health, vigor, organization, and resilience, are probably best understood from an ecological perspective. However, they apply equally to the human aspects of assessment. Ecosystem health assessment is a useful tool for evaluating the ecological and social outcomes of collaborative management (Muñoz-Erickson et al. 2007).

Although the term integrated assessment is used to describe a range of methodologies it is now accepted as referring to a formal process in which a multidisciplinary team work together to assess a wetland in terms of its combined biodiversity, economic and livelihood values. The process is described below.

The Integrated Assessment Process

An integrated assessment should consider all relevant aspects of a wetland landscape (Fig. 1) and will ideally do this in a single coordinated program of activity. The International Union for Conservation of Nature (IUCN) has developed an Integrated Wetland Assessment Process, and this recommends that an integrated approach is adopted during all phases of the assessment process.

The outline process is as follows (Fig. 2):

- Initially, all parties in the process would jointly define a set of research and/or management questions on the basis of the management objectives.
- The questions are addressed though a multi-disciplinary field survey, which combines biodiversity survey, livelihoods survey and economic valuation.
- The results of the survey are reported in a single integrated output.
- The report leads to recommendations and management advice which form the basis of a management plan.

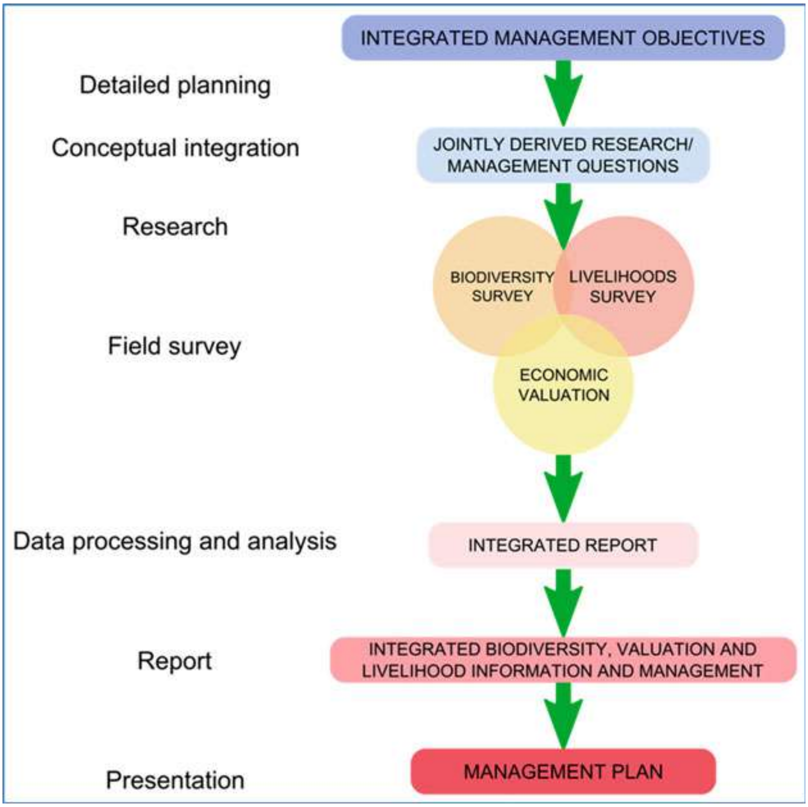

Fig. 2 The integrated assessment process as employed by an integrated survey team (Figure created based on a concept in Springate-Baginksi et al. 2009)

Initially, all parties in the process would jointly define a set of research and/or management questions on the basis of the management objectives. The questions are addressed through a multidisciplinary field survey, which combines biodiversity survey, livelihoods assessment, and economic valuation, and the results of the survey are reported in a single integrated output. The report leads to recommendations and management advice which then form the basis of a management plan.

A benefit of this approach is that information can be exchanged between disciplines at any stage in the process and hence ensures that work carried out by different disciplines is compatible. This model helps wetland conservation and development stakeholders to move away from a situation where they are making decisions on the basis of a series of biodiversity assessments, economic valuations, and social development reports that have been carried out by different groups of people, who were commissioned separately by program or project partners, did not consult one another, worked in different places at different times to each other, using different methods, analytical tools, and scales of working, and were each able to provide only a part of the information required leaving gaps which had to be filled by information derived from guesswork, inapplicable generalizations, or vested interests (Springate-Baginski et al. 2009).

Integrated assessments are not always straightforward to carry out. The planning and human resource needs required to carry out an assessment in this way can be considerable. The participants each have to get used to new methods, unfamiliar terminology, and to different ways of thinking. This acclimatization takes time but is a necessary part of forming a team capable of carrying out an effective assessment. However, all parties also benefit from the wider scope and sharing of information as the scientific community receive the necessary guidance on prioritizing their research activities, and policy makers better understand how to design effective and scientifically sound policies in order to end up with an assessment that will deliver useful information to decision makers (Brouwer et al. 2003).

Indicators for Integrated Assessment

Environmental indicators are generally understood as quantifiable variables which provide information about changes in environmental conditions (Brouwer et al. 2003). Indicators can be quantitative or qualitative depending on the nature of the condition they are describing. For example regulators may wish to enforce an environmental standard on the basis of quantitative criteria, whilst a qualitative description of community structure may be a more appropriate indicator of ecological disruption (Noss 1990).

In addition to selecting indicators that show a consistent and measurable response to environmental change, integrated assessment will require the set of indicators selected are capable of being brought together for cross-discipline assessment.

An Example of Integrated Assessment: The South Lincolnshire Fens and River Glen Catchment. Lincolnshire, UK

The River Glen Catchment Management Study was commissioned in the context of a growing consciousness of the need to conserve and restore habitats whilst supporting the livelihoods and considering the impacts of future climate change (Stratford et al. 2014). It sets out to integrate the planning of water resources and land use for public water supply, natural environment, agriculture, flood risk management, navigation, and leisure. A key outcome of the study was to propose new pilot wetland restoration areas which could support flood risk management works, investigate the combined use of water storage for environment and agriculture (e.g., winter-filled wetlands which support trickle irrigation projects in early summer), contribute to the local economy, and support local EU Water Framework Directive (WFD) shortfalls and objectives.

A partnership approach was developed with consultation of relevant stakeholders. The initial review phase revealed the following:

- A desire to address the decrease in area of wetland habitat through restoration of wetland features; however, without additional hydrological storage it is unlikely that the water demands of these features will be supported through the summer months.
- Agricultural activity in the area is predicted to become limited as water availability declines in the future. There is a need to support sustainable agriculture into the future and both better use of resources and access to additional resources could address this.
- A combination of natural processes and instream management has had a detrimental effect on natural geomorphological functioning and catchment connectivity. The impact of these two pressures needs to be understood further. Instream habitats are currently impacted by poor water quality and variable water quantity.
- The future prosperity of the study area can benefit from improved features to attract tourism including making the most of current resources such as wildlife reserves, improving amenity features such as cycle paths, and opening the area up to new opportunities (e.g., development of navigable inland waterways).

Four work steams were proposed, which will collectively address many of the pressures identified and will constitute a significant positive step towards achieving the issues identified through the integrated assessment.

- **Creation of wetland features**. Four areas for wetland restoration and/or creation have been identified, featuring reedbed and open water areas, wetland habitat mosaics, and raised water level areas.
- **Modification of instream and riparian habitat**. Six sites have been identified based on flow regime, water quality pressures, and instream macrophyte and

invertebrate habitat for a combination of solutions including reedbed filters, riparian restoration, and other instream improvements.
- **Targeted stakeholder engagement**. The study has identified the following areas needing further stakeholder engagement: the farming community, navigable waterways, water companies, flood management practitioners, mineral extraction companies, and the tourism industry.
- **Targeted research**. There remain some as yet unanswered questions and targeted research projects are recommended to address these.

In the event of no action, both demand for water and availability of water are likely to continue to diverge. The change in seasonal availability of water is likely to increase risks associated with both flooding and drought. It is probable that the requirements of many stakeholders (agriculture, human consumption, natural environment) will increasingly compete for limited resources, and in an increasing number of years the lack of resources will have a detrimental effect. Regardless of any climate change impact, the projected population increase in this region will lead to increased demands for water. This range of trends and pressures are common to many catchments in the developed world and the integrated approach outlined here may potentially have application elsewhere.

Future Challenges

There is currently a lack of understanding and consensus on which indicators best represent particularly the less tangible aspects of integrated assessment. It is recommended that a detailed analysis should aim to identify indicators for integrated natural resource management and link these to potential indicators for ecosystem services (de Groot et al. 2008).

Economic pressures may limit some aspects of an assessment and this may ultimately affect the robustness of any conclusions. It is recommended that, as a minimum, the following core aspects are maintained (Brouwer et al. 2003):

- Adoption of the catchment scale for analysis
- A good understanding of the hydrological functions provided by the wetland
- A framework within which to capture and map cause and effect relationships
- Use of recognized economic principles for valuation of water-related services

Efforts should also be made to simplify and harmonize the language used in assessment for delivery of clear and consistent messages. For example, there is often confusion between the terms ecosystem (or environmental) “goods,” “services,” “benefits,” “functions,” and/or “values.” Future assessments should, at the outset, aim to establish an agreed terminology and set of definitions, making clear in any reporting what these are and what they mean.

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Monitoring of Wetlands: Overview

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Charlie J. Stratford

Contents

Abstract

Wetland monitoring provides the key to understanding how wetlands function and how they respond to environmental change. Monitoring programmes can be designed for a range of purposes such as ensuring that wetlands are managed in accordance with best practice or quantifying how effective a habitat management strategy has been. Proper planning is required to support robust statistical data analysis and to see that monitoring activities complement existing work. Technological advances are improving the temporal and spatial scales over which monitoring can be carried out and schemes to involve volunteers in making observations are both generating useful data and raising awareness of the importance of wetland habitats.

Keywords

Data collection · Long-term · Large Scale · Citizen Science · Volunteer

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Introduction

Wetland assessment and wetland monitoring are closely related. While assessment is a single observation or "snapshot", monitoring provides information about the wetland over a period of time. The benefits of monitoring are far-reaching, from underpinning and validating scientific understanding to providing the data and information necessary to check if a regulation is being adhered to. From a scientific perspective, monitoring is the key to understanding environmental change processes, whether on a large scale, such as the response of global peatlands to climate change, or on a small scale such as nutrient cycling within the root zone of a wetland plant. From a management perspective, monitoring can both generate the critical information that is essential for providing sound stewardship of the environment and provide a basis upon which to hold individuals and organisations accountable for their environmental stewardship (Office of the Auditor General of Canada 2014). For the Ramsar Convention on wetlands, assessment and monitoring are important pillars under the implementation of the Convention to achieve "wise use" of wetlands (Finlayson 2012).

Assessment and monitoring activities include establishing the location and ecological characteristics of wetlands (baseline inventory); assessing the status, trends, and threats to wetlands (assessment); monitoring status, trends, and reductions in existing threats or the appearance of new threats (monitoring); and taking management actions to redress the causes of change in ecological character of wetlands (Ramsar Convention Secretariat 2010).

Reasons to Monitor

Maintaining healthy wetlands is vital if they are to provide the range of benefits (ecosystem services) that have been identified in recent years through reports such as the Millennium Ecosystem Assessment (2005). Wetland assessment aims to establish the current condition of a site and compare it to its previous condition and/or potential condition. The assessment process is likely to to lead to a set of recommendations, which may include some form of monitoring, e.g., to observe whether a certain policy or standard is being adhered to (compliance monitoring), to see whether a planned activity has been properly carried out (implementation monitoring), to establish whether an activity has been effective in achieving its goal (effectiveness monitoring), or to test if a conceptual model or understanding is correct (validation monitoring; see Table 1). Monitoring is the mechanism that enables us to do this and an effective monitoring programme will support:

- Understanding of natural processes
- Evaluation of remedial and restoration activities
- Awareness of changes in threats to wetlands or of new threats
- Protection of wetland resources
- Better catchment management

Table 1 Summary of different types of monitoring and what each aims to do (Compiled from data in Government of British Columbia (2014) and Sydney Coastal Councils Group Inc. (2014))

	What question does the assessment or monitoring aim to answer?	Details
Snapshot assessment	What is the current condition of the wetland?	This is often the trigger for subsequent monitoring programmes
Compliance monitoring	Are the legal requirements, e.g., for pollution prevention or water abstraction, being met?	This can be a useful regulatory device however it does not evaluate effectiveness and fails to address two essential management questions: Are we going in the right direction? How do we change if the direction is wrong?
Implementation monitoring	Were the proposed steps in the management plan/policy carried out correctly?	This is carried out to assess the rate of progress towards a specific goal. It is useful for management reporting and frequently used in industry
Effectiveness monitoring	Are the desired outcomes of the management plan/policy being achieved?	This is used to determine whether the plans or practices implemented are meeting the anticipated outcomes. It provides limited opportunity to learn new information and refine future activities
Validation monitoring	Is our hypothesis about the functioning of the wetland ecosystem correct?	This seeks to verify and validate the conceptual understanding of the study area. Extending validation monitoring through experimental treatments can provide learning and research opportunities

The level of effort required typically increases from compliance monitoring, through implementation and effectiveness monitoring, to validation monitoring (Fig. 1). Routine or extensive data collection, perhaps involving simple yes/no lists, is normally sufficient to monitor compliance. Implementation monitoring may in addition require quantitative data collection and analysis in order to assess whether the activity in question has been properly carried out. The level of effort required to monitor effectiveness can vary depending on the nature of the outcome being assessed, and can in some cases require a considerable programme of activity (e.g., has a certain land management practice been effective in increasing the abundance of a particular wetland species). Validation monitoring, by which a hypothesis is tested, is likely to require the greatest level of effort in order to carry out a scientifically sound experiment. Detailed information will probably be collected before, during and after the event or intervention of interest and subsequent analysis will take a similarly robust approach.

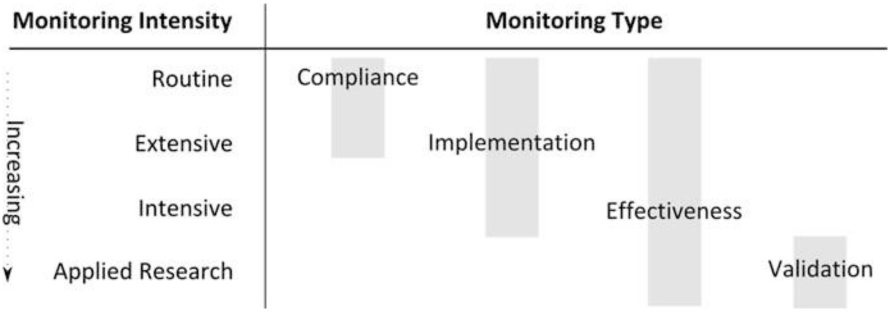

Fig. 1 The relationship between monitoring type and monitoring intensity for the four types of monitoring outlined in Table 1 (Figure created based on concept in SMC (2014))

The Basic Requirements of an Effective Monitoring Programme

An effective monitoring programme should be based on a clearly defined question, and a good understanding of that question will help to ensure that the full range of aspects to monitor is identified and considered in the programme. To provide maximum benefit, the monitoring programme should consider, and possibly sit within, a broader set of objectives so that complimentary activities are coordinated (Department of Natural Resources and Water 2007). Time spent at the outset liaising with stakeholders will greatly increase the chances of developing and carrying out a successful monitoring programme. Environmental monitoring is likely to be most effective when there is:

- Sound planning from the outset, and coordination with other relevant programmes of activity
- Collaboration with the right partners so that all relevant aspects are considered
- Efficient use of resources with a view to making the monitoring programme sustainable
- Appropriate reporting of results, and effective incorporation of data into decision making

As an example of a high-level coordinated programme, Table 2 details the ten recommended elements that should be considered when designing a monitoring and assessment strategy (USEPA 2003, 2006). The ten steps cover the entire process and provide a useful template and checklist upon which to base any monitoring campaign.

It is also useful to think ahead as to how the data generated by the monitoring will be analysed, making sure that the requirements of relevant statistical approaches are taken into account. Sites should be selected that capture the variability of the population in question. Three approaches to sampling design are widely used, depending on the purpose of the study, and these are applicable at a range of scales. A stratified random sample design looks at the whole population and takes a random sample from it. This

Table 2 Details of the ten recommended elements of a wetland monitoring and assessment programme (Compiled from data in USEPA (2003) and USEPA (2006))

	Element	Description
1	Monitoring program strategy	The wetland strategy should be included within the wider environmental monitoring strategy. Such integration fosters coordination and prioritization of monitoring activities across waterbodies. It should be comprehensive in scope and identify any impediments to an adequate monitoring program. The timeline for implementation should not exceed 10 years.
2	Monitoring objectives	The monitoring program should be efficient and effective in generating data that serve management decision needs. For example, supporting establishment of a baseline of wetland condition and refine or create wetland specific water quality standards, evaluating the cumulative effects of wetland loss and/or restoration, or the environmental consequences of a federal action or group of actions including the effectiveness of compensatory wetland mitigation.
3	Monitoring design	The monitoring design should specify the approach, which will likely be one or more of: • Examining every unit in the population of interest – this approach identifies significant wetlands in need of specific regulatory attention. • Probability sampling – studies based on statistical samples rather than complete coverage are referred to as sample surveys. • Best professional judgement to target sampling within specific wetlands for purposes of comparison. The strategy should identify the type of wetland classification system and mapping system they intend to use as part of their sampling design.
4	Core and supplemental indicators and methods	Core indicators are selected to represent the high-level characteristics that should be considered at all sites in the programme, whilst additional supplemental indicators would be used according to specific decision criteria. For example, a set of core indicators can be used to characterize wetland condition in terms of ecological integrity with supplemental indicators then used to characterize a wetland's special significance as critical or outstanding wildlife habitat.
5	Quality assurance	Quality management and assurance plans should be established, maintained, and peer reviewed in accordance with the relevant policy to ensure scientific validity of monitoring and laboratory activities and to ensure that government reporting requirements are met.
6	Data management	Modern data storage systems are helping to make data rapidly and widely available. Partnerships will help improve data sharing and reduce the cost of wetland monitoring by minimizing duplication. For example, by selecting data from comparable ecoregions with similar wetland classes, it may be able to use existing stored data to assess wetland reference condition.

(*continued*)

Table 2 (continued)

	Element	Description
7	Data analysis/assessment	Common methods for analysis help to ensure that the rigor of wetland sampling and analysis is conducted in a manner that is commensurate with that needed for a particular type of decision making.
8	Reporting	Reporting should be appropriate to the target audience. The intended user group, format, style, and peer review requirements of project reports should be identified in the initial phases.
9	Programmatic evaluation	The use of information produced should be documented to determine how well the program serves its decision needs. The review will help identify contingencies that will allow monitoring to continue in the event of an operational issue (e.g., a funding shortfall).
10	General support and infrastructure planning	Current and future resource needs should be captured. Programmes often commence at geographical locations where there are wetlands at risk, discretionary dollars, interested people, and existing data. The actual costs should be documented in terms of both money and time, taking into account the logistics and budget resource needs relative to project staffing, training, field operations, laboratory needs, and office needs. Such budget documentation forms the basis for future funding requests and project plans.

can be very effective in gaining a high-level characterization of the population, but may poorly represent rare and possibly valuable components. Targeted sampling aims to identify sites experiencing specific issues and focus activities at those sites. Reference sites are also monitored in order to establish the range of conditions along a gradient from impacted to nonimpacted. This approach is effective in dealing with a particular issue as long as the issue has already been identified. The third design is Before-After, Control-Impact (BACI), and this compares the response of a site experiencing an event to a reference site not experiencing the event, covering a time period from before the event to after the event (USEPA 2002).

Spatial and temporal variability needs to be taken into account at all times and it is particularly important to properly understand the natural range of conditions when establishing a reference. If there is prior knowledge of the sample population it may be possible to identify the spatial and temporal monitoring resolution required, however without this there is likely to be a degree of iteration as the preliminary dataset is collected and analysed. The duration of monitoring should be sufficient to capture the range of processes operating on a seasonal, annual, and interannual timescale. The benefits of long-term monitoring should not be underestimated (see Clarke, *The importance of long-term groundwater level monitoring*). USEPA 2002 suggests that “Rather than monitor intensively for one year before and one year after an activity, it may be more cost-effective to carry out less intensive monitoring but over a longer period such as at least three years before and after." Once the temporal variability has been established, it may be possible to reduce the sampling frequency to just those times when a pressure is more likely to occur (e.g., during low or high flow conditions) however this is dependent this is dependent on having developed a robust understanding robust understanding of the temporal variability.

What to Monitor?

In order to understand what to monitor, it is helpful to have helpful to have an initial conceptual understanding conceptual understanding of the wetland system. The aim of this is to make an initial assessment of the key processes and identify the likely links between cause and effect. This may be achieved by consulting existing data sets, such as topographic, geological and habitat maps, and expert knowledge, which can include both specialists in a relevant scientific discipline and people with a specific knowledge of the site(s) under investigation. The range of aspects to be included in the monitoring programme will also be influenced by the type of monitoring carried out. Taking as an example the broad question "Are our restoration activities achieving their targets?" this could be investigated using any one of the four monitoring types listed in Table 1. The aspects required for each would be quite different. Compliance monitoring would need to include appropriate measures of the restoration targets (e.g., quantity of restored habitat), implementation monitoring would need to include measures of the process and how well it was carried out, effectiveness monitoring would include aspects of both the restoration activities and their outcomes, and validation monitoring would include measurements necessary to establish why a certain activity did or didn't have a certain outcome.

The availability of and access to relevant information has improved greatly over the past 10–20 years with many datasets being made publically available. The development of tools such as Geographic Information Systems (GIS) has made the process of interrogating these datasets more straightforward and they are widely used in identifying and analyzing wetland habitats and understanding the spatial relationships between them and their surroundings. This information can be used to form an initial conceptual understanding, which can then be developed by adding increasing detail and resolution as necessary to support the monitoring objective.

To illustrate this, Fig. 2 shows a conceptual hydrological model of a wetland, which could have been constructed using readily available information. Building upon this conceptual model, it is now desirable to move from a qualitative to a quantitative description of the processes identified. Each process identified (e.g., precipitation, runoff, evaporation) can be monitored using a range of techniques and at a spatial temporal scale appropriate to the study. For example, precipitation (P) could be measured using a single point rain gauge recorded once per day, or from a network of rain gauges logged automatically every 5 min. The movement of water between the wetland and aquifer beneath could be estimated by measuring the relative water levels hydraulic properties of each layer, or calculated from measurements of water temperature through the wetland/aquifer profile.

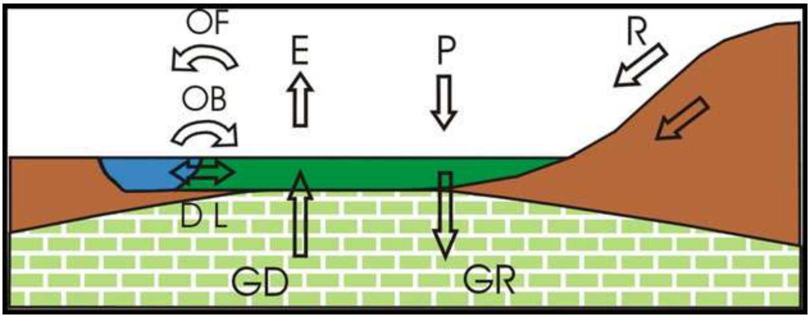

Fig. 2 Conceptual hydrology of a combined groundwater and surface-water fed wetland system. The wetland (*shaded green*) is adjacent to a watercourse (*shaded blue*), and sits on top of an aquifer (*green brick* shading) and at the bottom of a hillslope (*shaded brown*). *P* precipitation, *R* runoff, *E* evaporation, *D* drainage, *L* lateral inflow, *OF* overland outflow, *OB* overbank inflow, *GD* groundwater discharge, *GR* groundwater recharge

When selecting where and what to monitor, try to ensure that the following points are considered:

- Is it representative?
  - Does it answer the question?
  - If required, can it cope changes over space and time?
- Is it reliable?
  - Does it adequately reflect a response to the impact concerned?
  - Is it scientifically sound?
- Is it feasible?
  - Will it work within the confines of the project in question?

Monitoring Techniques

A wide range of techniques exist for monitoring different aspects of a wetland habitat, making use of various hydrological and ecological field sampling and laboratory analysis methods. Over the years, technological advances have provided increasingly efficient and effective monitoring tools, and recent developments in geophysical and remote sensing techniques are increasing the efficiency with which monitoring can be carried out. For example, measurement of soil moisture has progressed from costly point-scale destructive lab measurement (sample volume $\sim 10^{-5}$ m^3), to nondestructive sensor-based point measurement (using instruments such as the Neutron Probe and Capacitance Probe), to nondestructive field or landscape-scale measurements using instruments such as the land-based COSMOS sensor or satellite-based detectors (sample volume $\sim 10^8$ m^3) (Fig. 3, Robinson et al. 2008).

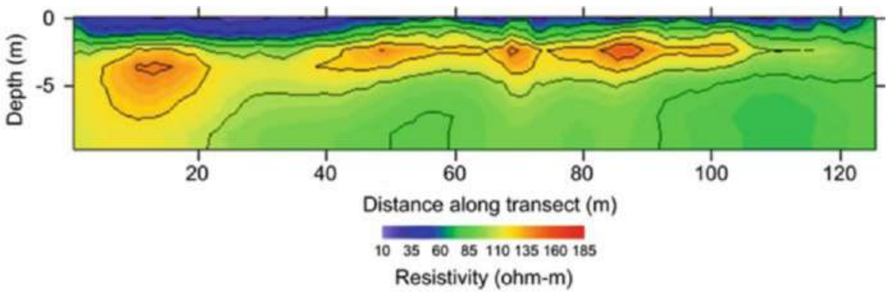

Fig. 3 Example of field-scale resistivity data collected using electrical resistivity tomography at Boxford water meadows, UK. These data can be used to estimate soil moisture content (Source Musgrave 2006; Reproduced with permission)

Satellite-based sensors provide data on a range of wetland properties. High resolution multispectral mappers are available for mapping small patchy upstream wetlands. Thermal infrared scanners can map coastal water temperatures, while microwave radiometers can measure water salinity, soil moisture, and other hydrological parameters. Synthetic Aperture Radars (SAR) help distinguish forested wetlands from upland forests. Airborne LIght Detection And Ranging (LIDAR) systems can be used to map wetland topography and produce beach profiles and bathymetric maps (Klemas 2011; Purkis and Klemas 2011). At the other end of the spatial scale, wireless automatic hydrological loggers can be deployed on the ground to send either continuous or event-triggered information about water level, flow, rainfall, or other processes at particular locations of interest.

Large-Scale Monitoring

The monitoring discussed up to now has largely been conducted in response to a specific issue. Although potentially more difficult to justify financially, monitoring purely to observe the environment has provided some fundamentally important datasets (e.g., long term carbon dioxide measurements at the Mauna Loa Observatory in Hawaii). Whilst it is accepted that it is not viable (either scientifically or economically) to measure everything everywhere, establishing multidisciplinary long-term monitoring networks of sites is seen as increasingly important in order to understand change and its impact. The benefits of long-term monitoring are widely recognised, for example providing the context and background for more focussed observations. This context can help differentiate between short-term variability and long-term trend and this is an essential in order to understand environmental change. Drivers of change can have variability occurring over timescales from fractions of a second to tens of (or more) years, and trends often cannot be judged in just a few years. Long-term monitoring provides the baseline against which the present and future state of the environment can be assessed in a reliable context.

An example of a Large scale monitoring network is the English Long-Term Monitoring Network (LTMN) (Natural England 2014). At the end of 2013, the network consisted of 27 operational sites covering a range of habitat types, geographically distributed across England. At each site a range of hydro-climatological, atmospheric and ecological data are collected.

- Weather (at least hourly)
- Air pollution – diffuse ammonia and wet (precipitation) deposition (monthly)
- Butterflies (weekly during the flying season)
- Birds (twice a year)
- Vegetation (every 4 years)
- Soils (every 6 years)
- Land management activities (as they arise)

The data collected through this project is managed centrally and is made publically available through the Environmental Change Network website (www.ecn.ac.uk) where users can view and produce customized data summaries through interactive web pages or request raw data for analysis.

Volunteer Monitoring

The cost of carrying out wide-ranging long-term monitoring is potentially very high. In some cases, volunteer monitoring programmes can provide a low(er) cost alternative to traditional monitoring traditional monitoring methods whilst also increasing awareness of environmental issues. One of the challenges in making volunteer monitoring work effectively is designing a scientifically robust programme, capable of providing useful scientific information, but which can be conducted by non-specialists. Volunteer monitoring can help provide data for wetlands that may otherwise be uneconomical, increase the amount of data available to decision makers at all levels of government, build awareness of wetland issues, and help prevent damage to wetlands. The USEPA Office of Water encourages all citizens to learn about their water resources and supports volunteer monitoring because of its many benefits. In the USA, national conferences bring together volunteer organizers and agency representatives in order to provide support, promote consistency, and share experiences. Similarly, the UK Environmental Observation Framework (www.ukeof.org.uk/) provides support and advice for setting up citizen science projects, as well as links to UK environmental datasets.

Future Challenges

The need for information and data that accurately reflects the complex processes in the natural environment is increasing. For example, collecting the data required to quantify wetland ecosystem services is important in understanding and protecting wetland systems, but is difficult to do in practice. Long-term and high-frequency monitoring is leading to larger datasets and there is a need to develop methods, tools, and techniques that can collect, store, and analyze this information in a cost-effective manner.

Challenges also exist in developing data collection methods that are robust and reproducible. Experimental tests of the differences in mapping between trained habitat surveyors, who all surveyed the same landscape of heterogenous vegetation, showed differences in the size, shape, location, and community allocation of polygon boundaries between surveyors (Hearn et al. 2011). The variability inherent in mapping the same piece of land was as much as 30% between surveyors. If surveys are carried out to characterize changes in vegetation over time, then this could distort the reported change in the character of vegetation, even if underlying vegetation assemblages remain broadly similar (Stratford et al. 2014).

In addition there is the challenge of demonstrating the importance of maintaining monitoring as economic pressures increase, and non-essential activities are scrutinized with a view to cutbacks and money-saving measures. Ongoing efforts should continue to make clear the importance of monitoring in providing useful data, whilst at the same time using a range of techniques to ensure that monitoring is carried out as efficiently as possible.

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Ecological Monitoring of Wetlands

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Tom Dahl

Contents

Abstract

Wetland resource planning, management, modeling, and policy formulation rely on scientifically sound information regarding the extent, type, and condition of wetlands on the landscape. This requires characterization of wetland resources as well as developing an understanding of how these systems respond to environmental change. Wetland monitoring is defined as the systemic observation and recording of current and changing conditions and provides these information needs. The choice of techniques is dependent on the programmatic objectives as well as the physical landscape and geographic extent of the monitoring effort. Mapping, inventories, surveys, and statistical sampling are all methods of data collection that serve as a basis for monitoring. Short-term monitoring is useful for project-level planning and compliance, mitigation, or remediation and for assessing the immediate impacts of specific events such as hazardous spills and other risk assessment studies. Longer-term monitoring is needed to assess ecosystems or landscape-level changes in hydrology, habitat fragmentation, climate

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change, responses of vegetation to stressors, species utilization of wetlands, and other ecological trends resulting from cumulative impacts. Monitoring for change in wetland extent provides crucial information for the development of national policy and legislation, governance of financial and technical assistance, tax reform, wetlands management strategies, research, and strategic planning efforts involving wetlands. Combination with monitoring of wetland condition (vegetation, soils, hydrology, water quality, or algae) has allowed the development of comprehensive wetland monitoring programs.

Keywords
Wetland monitoring · Wetland assessment · Spatial assessment · Wetland indicators

Introduction

Wetland monitoring is defined as the systemic observation and recording of current and changing conditions (USEPA 2013). Wetland resource planning, management, modeling, and policy formulation rely on scientifically sound information regarding the extent, type, and condition of wetlands on the landscape. This requires characterization of wetland resources as well as developing an understanding of how these systems respond to environmental change. Wetland monitoring programs provide these information needs.

Since the early 1970s, researchers have mapped and monitored wetlands in an effort to further understand these ecosystem intricacies and interactions (Ozesmi and Bauer 2002). Worldwide there are many efforts currently ongoing to monitor wetlands. Some are site specific and involve on-the-ground observations. Other projects are regional or landscape-level in scope and involve analysis of remotely sensed information, geospatial modeling and utilize geographic information systems to process large datasets (Lopez et al. 2013). Ecological monitoring of wetland sites contributes information to support international treaties, conventions, or agreements (Finlayson 1994; Backhaus and Beule 2005) and help conserve wetland resources.

Strategic elements for monitoring wetlands share some common objectives that include the ability to establish baseline data; make assessments of wetland extent (location, size, and type) and/or condition; monitor changes over time; provide the data needed to guide management, research, and policy direction; and provide a quantifiable element to gauge progress. Finlayson (1994) has presented a framework and guidelines for wetland monitoring that enumerate important concepts.

The transitional nature of wetlands and the role they play in landscape ecology makes interpreting the results of monitoring projects challenging. Spellerberg (2005) points out that because of their complexities and interactions, monitoring ecosystems can be difficult, logistically demanding, and costly. However, the melding of wetland science and landscape ecology is producing new applications for wetland inventory and monitoring, and there is renewed emphasis on the ability to monitor ecological conditions over variable spatial and temporal extents (Miller and Rogan 2007).

Strategies for Monitoring Wetlands

Monitoring of ecological change in wetlands can be undertaken at several levels of intensity and by using an array of approaches. The choice of techniques is dependent on the programmatic objectives as well as the physical landscape and geographic extent of the monitoring effort. For example, short-term wetland monitoring is useful for project-level planning and compliance, mitigation, or remediation and for assessing the immediate impacts of specific events such as hazardous spills and other risk assessment studies. Longer-term monitoring is needed to assess ecosystems or landscape-level changes in hydrology, habitat fragmentation, climate change, responses of vegetation to stressors, species utilization of wetlands and other ecological trends resulting from cumulative impacts (Fig. 1).

Mapping, inventories, surveys, and statistical sampling are all methods of data collection that serve as a basis for monitoring (Spellerberg 2005). A wetland inventory has been defined as a measurement, catalogue, or estimate of the extent of the wetland resource for a defined geographic area (Dahl and Watmough 2007). Wetland inventories are often associated with comprehensive assessment of wetland status at a given point in time and contribute substantially to the knowledge base about wetland types and extent. Inventories should produce quantitative spatial data that are needed to track wetland losses and gains as well as ecological changes to wetlands (i.e., migration, succession, etc.). Wetland restoration and

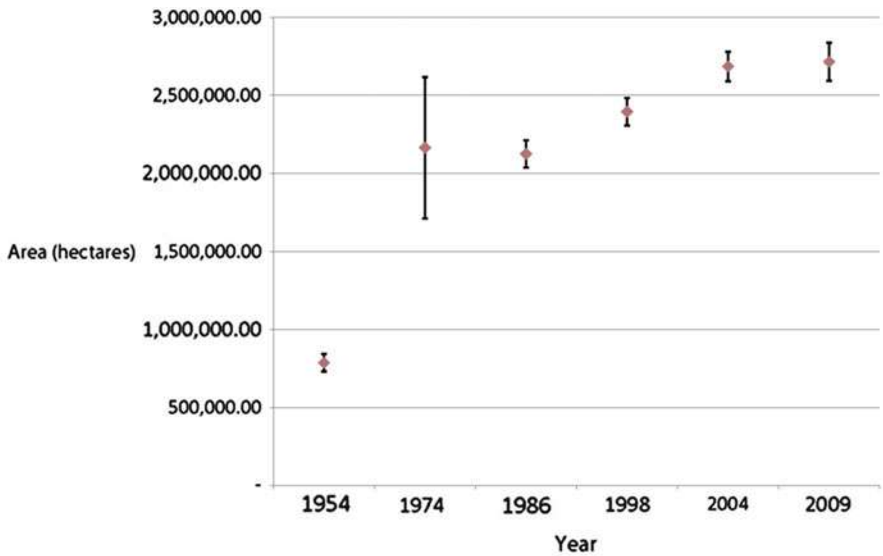

Fig. 1 Long-term trends of freshwater pond area in the conterminous USA provide baseline information about wetland extent and type. Shallow freshwater ponds have increased steadily throughout the last 50 years, with current area more than three times what it was in the 1954. However, the quality of these wetlands has not been assessed (Figure based on data in various reports by US Fisheries and Wildlife Service)

creation can also be measured as a way to gauge progress in achieving wetland conservation goals.

A number of regional and national wetland inventories have been conducted using various techniques. Finlayson and Davidson (1999) have summarized these efforts in describing the global extent of wetland inventory information. Wetland inventories are a powerful tool for conservation planners; however, comprehensive, detailed mapping projects are not easily updated, maintained, or enhanced to meet the changing needs of resource managers and address evolving resource issues. Newer technologies in the fields of remote sensing, mapping, and landscape characterization are providing improved inventory data with higher spatial resolution and more finite categorization of wetland types (Halabisky et al. 2013). However, direct comparison of these modern datasets with older wetland inventory map information can confound analysis of wetland area changes (Genet and Olsen 2008). As the need to update wetland inventories often out-distances current budgets and program capabilities, there continue to be pressures on the wetland resources that require effective monitoring at temporal and spatial scales useful for directing wetland conservation efforts. In these instances, a different approach for monitoring wetland resources may involve a scientifically based process based on probabilistic sampling to periodically measure wetland area extent (status) and change rates (trends).

Examples of Wetlands Monitoring Approaches

There are various examples of wetland monitoring efforts that have been undertaken to broaden our understanding of the geographic distribution and ecological importance of wetlands. Some of these studies include determining the global distribution of mangrove wetlands (FAO 2007), the status of global peatlands and atmospheric carbon emissions (Joosten 2009), a series of in-depth reports on monitoring coastal environmental condition including coastal wetlands in the USA (USEPA 2011), and an effort currently underway to develop and electronic global saltmarsh atlas by the Marine Assessment and Decision Support Program of UNEP World Conservation Monitoring Center.

Monitoring Wetland Quantity

In the USA, federal policy goals pertaining to monitoring wetlands have historically been based on area extent and the ability to provide a quantitative measure of wetland area as a means to assess progress toward achieving the national goal of "no net loss of wetland." Modifications to this policy have been made with the recognition that comprehensive monitoring of wetlands includes both a quantity and quality component (Council on Environmental Quality 2005). Geospatial assessments to ascertain progress toward an overall increase in the area of wetland provide important information on wetland habitat types (Fig. 2), land use trends, and scientifically defensible

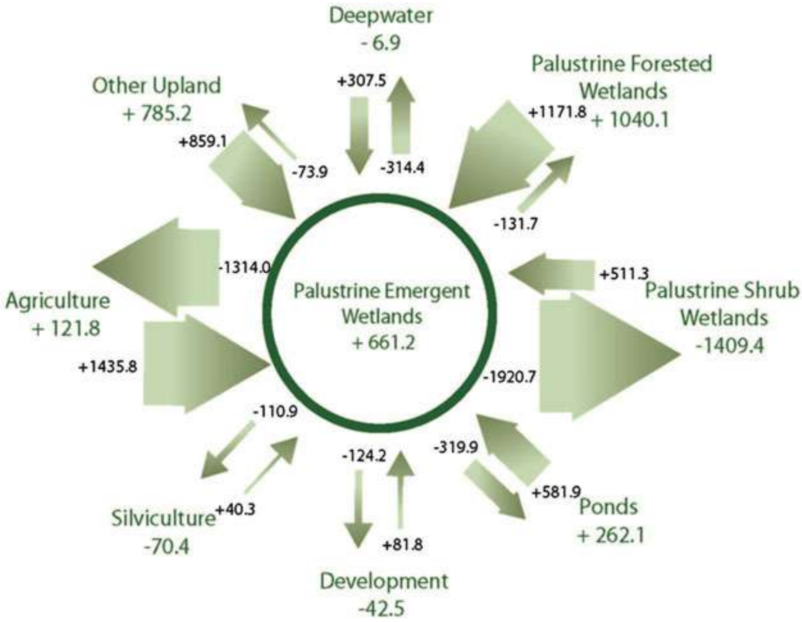

Fig. 2 Area immigration and emigration of freshwater emergent wetland in the conterminous USA (all numbers are in thousands of hectares). *Arrows* indicate the estimated area lost and gained between upland land use, deepwater lakes and rivers, and other wetland types. The interchanges shown represent changes in wetland area that occurred between 2004 and 2009 (Source: Dahl 2011). Monitoring wetland extent and type can yield this type of ecological information

indicators for achieving national or regional policy goals (Dahl and Watmough 2007). The USA developed a program specifically to monitor wetland area using a single, consistent definition and study protocol and has produced a series of wetland studies that document spatial changes on the landscape (Dahl 2011). The information generated has been crucial in the development of national legislation, conservation provisions, governance of financial and technical assistance, tax reform, federal and state policy development, wetlands management strategies, research, and strategic planning efforts involving wetlands. The most recent information on coastal wetland trends in the USA is being used to assess coastal wetland conservation efforts and help formulate national policy (Dahl and Stedman 2013).

There are several limitations to these methods to measure changes in wetland area including the availability of high quality and timely imagery for use in change detection. Additionally, wetland size (small wetlands necessitate the use of appropriate imagery to detect saturation or ponding) and variability in hydrologic conditions influencing wetland classification are factors (Fig. 3). Some land use practices can also affect wetland change detection. Disturbed sites or lands in transition from one land use type to another often result in ambiguous remote sensing indicators, making wetland classification and determination more difficult (Dahl 2004).

Fig. 3 Spring flood waters cover both wetlands and uplands along the Lemonweir River, Wisconsin, USA. Extreme climatic conditions can negate the value of early spring (leaf-off) imagery intended to aid in the inventory and monitoring of wetland habitats (Photo by T. Dahl)

Monitoring Wetland Quality

While the data on wetland area trends have provided important information on land use changes and wetland extent, recent development of wetland condition indicators (i.e., vegetation, soils, hydrology, water quality or algae) to aid in the determination of the ecological integrity of wetlands has made it possible to integrate quantity and quality data into comprehensive wetland monitoring programs (Genet and Olsen 2008). For example, algae and aquatic vegetation have been used to indicate physiochemical conditions and assess biological integrity in wetlands in the USA (USEPA 2002).

Assessing wetland quality requires a suite of methods or levels of measurements. Base level measurements are designed to be landscape-scale parameters using geospatial information and remote sensing techniques to assess large areas or numbers of wetlands. A second level of assessments uses rapid qualitative methods based on simple observational metrics specific to individual wetlands. The highest level involves intensive site assessment based on quantitative field sampling. All three levels provide data that in aggregate can be used to make determinations about wetland quality (MN Pollution Control Agency 2006). In the USA, the existing monitoring framework used to determine wetland extent has become a complimentary component for conducting a baseline assessment of wetland quality, and the two monitoring efforts will provide scientifically defensible information documenting the current status of wetland quantity and quality (Scozzafava et al. 2007).

Future Challenges

There are considerable challenges associated with monitoring wetland changes that can occur both spatially and ecologically over time. Wetland monitoring information is needed not only to assess past alterations and update status but also to anticipate emerging issues such as climate change, increasing demands on natural resources and landscape-level changes. Considerations should also be given to the comparability of datasets and the ability to integrate monitoring data across networks.

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Compliance Monitoring of Wetlands

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Emma Goodyer and Johan Schutten

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Abstract

Protection of valuable wetland resources is commonly achieved through a combination of direct protection of high-value wetland habitats as "protected sites," indirect protection of wetland habitats through a protection of the hydrological functionality of the landscape, and limiting environmental pressures on wetland habitats, for example, through licensing of potentially damaging activities. Both the protection of hydrological functionality and the limiting of pressures require regulatory controls. Compliance monitoring is the tool to ensure conformance with planning or licensing conditions and thus ascertains if the hydrological functionality is maintained or the environmental pressure is within acceptable levels. The purpose of compliance monitoring is typically to demonstrate that the quality and integrity of a

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wetland are maintained while carrying out, and in some cases for a period before and after, a regulated activity. The chapter provides guidelines for the design of a compliance monitoring scheme, including baseline monitoring, monitoring during the regulated activity, and post-activity monitoring. Compliance monitoring can be carried out by the environmental regulating authority or by the operator of the activity. Where the operator carries out the monitoring, the regulator is responsible for spot checks to verify accurate reporting or to support enforcement action.

Keywords
Compliance · License · Monitoring · Environmental standards · Hydrological model · Environmental risk

Introduction

Wetlands provide a range of ecosystem services (Maltby and Ormerod 2011); however, there has been an accelerated loss of wetlands in the last 100 years (Silva et al. 2007). Protection of the valuable wetland resource is commonly achieved through a combination of (a) direct protection of high-value wetland habitats as “protected sites,” (b) indirect protection of wetland habitats through a protection of the hydrological functionality of the landscape (such as the European Water Framework Directive, WFD; European Commission 2013a), and (c) limiting environmental pressures on wetland habitats, for example, through licensing of potentially damaging activities. Both the protection of the hydrological functionality of the wetland and the limiting of pressures require regulatory controls. Compliance monitoring is the tool to ensure conformance with planning or licensing conditions and thus ascertains if the hydrological functionality is maintained or the environmental pressure is within acceptable levels. The purpose of compliance monitoring is typically to demonstrate that the quality and integrity of a wetland are maintained while carrying out, and in some cases for a period before and after, a regulated activity.

In the context of this chapter, the term “wetlands” is used to refer to marginal and terrestrialized wetland habitats in a temperate-Atlantic setting (e.g., the UK) and does not encompass the wider Ramsar definition of wetlands (Ramsar 2013) to include bodies of water. Increasing public interest in wetlands and the ecosystem services which they provide has led to the creation of protective legislation. In Europe the Water Framework Directive (WFD) aims to protect the hydrological functionality of the landscape and thus the water environment. Other legislation such as the Habitats Directive (European Commission 2013b) aims to directly protect high-value wetland habitats and other biodiversity interests. The WFD brought about the requirement for functional and pressure-related wetland assessment methodologies which have been adapted by member states to deal with specific issues faced in different countries. An example is the translation of the WFD into Scots Law to form The Water Environment and Water Services (Scotland) Act 2003 (WEWS; The Scottish Government 2013). Statutory, regulatory organizations, such as the environment agencies, use these methodologies to set limits to hydrological change or environmental pressure of proposed activities and thus reducing the risk of environmental damage to wetlands to acceptable levels. Compliance monitoring is used to ascertain whether the activity is carried out in accordance with these specific standards and thus has resulted in an acceptable environmental pressure and ultimately protected the wetland habitat and its ecosystem services.

The Application of Compliance Controls

Developing a compliance monitoring scheme is challenging, and one of the initial difficulties, to both the environmental regulator and the applicant, is identifying where the wetlands are that could be affected by the proposed activity. In many cases this relies on expensive vegetation surveys although projects such as the Scottish Wetland Inventory (SEPA 2013a) are being established to aid national and local development plans. The requirements laid down by the regulator are tailored to the level of risk which the activity poses to a wetland.

Where wetlands are present and a proposed activity is deemed likely to pose a risk to their function or state, compliance controls are introduced to safeguard the wetland. This may involve monitoring to demonstrate that the level of impact resulting from the activity is acceptable. These controls are then included in a legally binding agreement between the environmental regulator and the party carrying out the proposed activity, which forms part of the license to carry out that activity. Compliance with the conditions within the agreement is assessed through compliance monitoring.

Monitoring of compliance is targeted to measure the level of environmental pressure and the function or functions of the wetland which are likely to be impacted. In the UK a *source* -> *pathway* -> *receptor* model framework is used for assessing whether an environmental pressure could act on a wetland receptor and how the pressure changes during the migration between pressure source and wetland receptor. For example, a groundwater abstraction lowers the water table in the aquifer (pressure), which transmits to the wetland by causing a reduced groundwater input. The change in available water results in an impact upon the wetland receptor, and this may be evidenced, for example, by a change in vegetation or chemical parameters through monitoring.

Designing a Compliance Monitoring Scheme

Compliance monitoring should be designed to ascertain if the hydrological quality or quantity standards in the license are met or if the wetland that is at risk is meeting relevant quality criteria during and after the activity. The aim is not to measure every possible variable and effect but to target the sampling effort and scale of monitoring to the variables which are likely to be impacted or that are included in the license as quality standards. The monitoring design should be specifically targeted to assess the agreed quality parameters or determine the change of hydro-ecological function and resulting wetland quality which may be impacted by the activity. It should cover an appropriate time scale to ensure that sufficient evidence is collected to clearly demonstrate whether any changes in wetland condition are related directly to the activity or due to unrelated factors. Analysis of the data should be scientific, and where possible statistical, to provide robust evidence of compliance or noncompliance.

Baseline Monitoring

Where an area of wetland is considered to be highly variable or the functioning of the wetland type is poorly understood, baseline monitoring may be required prior to carrying out an activity to assess the functioning or status of the wetland. Without a baseline dataset, data outside the agreed thresholds may be wrongly attributed to an impact caused by the activity when it may be due to natural, site-specific variability. Where the interaction between impact and an environmental parameter is well understood, baseline monitoring may not be required, and an agreed impact threshold may instead be included in a license.

Baseline monitoring informs two processes:

(a) It confirms the hydro-ecological functionality of the wetland at risk and thus helps with determining the risk of the activity to the wetland (assessment monitoring). This enables the conceptual hydro-ecological model of the site to be refined and the appropriate license conditions and compliance controls to be applied.
(b) It creates a nonimpacted baseline of wetland condition and environmental parameters against which a measured change in relation to the agreed threshold can be judged.

Monitoring During the Regulated Activity

Monitoring during the activity assesses the level of impact or environmental fluctuation in association with the threshold agreed in the license. An example is monitoring of water levels during an abstraction (Environment Agency 2003). Collection of monitoring data during the activity and regular analysis of this data is crucial to pick up an early indication of unacceptable environmental impact (as determined by the agreed threshold). This dynamic assessment may drive changes to working practices or mitigation measures which prevent further damage to a wetland. For example, water quality monitoring for water discharging to or within a wetland may be required as part of pollution prevention and control measures (The UK Government 1999).

Post “Activity” Monitoring

This stage of monitoring may be required for reviewing whether the objectives of wetland protection and license conditions have been met. For some parameters, such as impact on protected species, collection of a long-term dataset may be required to pick up delayed impacts which may result from the activity or to inform post-activity remediation or restoration work.

Deciding Which Parameters to Measure

For wetlands in a UK setting, many of the broad wetland types have been conceptualized (e.g., WETMEC models; Wheeler et al. 2009) and hydro-ecological data collected and synthesized to produce hydro-ecological guidelines for wetland vegetation types (e.g., Wheeler et al. 2004). The monitoring strategy may include point sampling or produce replicated data which take account of the spatial and temporal variability in natural wetlands. The monitoring design should be:

- Site specific: which areas are likely to be at risk?
- Activity specific: what aspects of the wetland environment is the activity likely to impact upon or which parameters are included in the license condition?
- Representative and targeted: The number of monitoring locations and a frequency of the monitoring suitable for the level of risk posed by the activity and the dynamics of the variables monitored. Monitoring can involve significant costs, and the regulator will take a proportionate, risk-based approach in its requirements for environmental information.

Demonstration of Compliance

Compliance controls aim to ensure that activities in the environment do not lead to unacceptable environmental harm. Compliance monitoring aims to verify that the license holder is complying with the conditions in the license (such as a maximum nutrient level in a surface water discharge or the maintenance of a certain wetland quality standard) and that these conditions are providing sufficient environmental protection. Compliance monitoring allows the early detection of problems and the implementation of contingency measures to remediate or prevent further harm. Some authorities use a standardized compliance assessment scheme (CAS; SEPA 2013b) to assess an operator's level of compliance with a license under a transparent and consistent framework. The CAS records any breaches of license conditions through a combination of site audits and inspections, sampling of discharges and emissions, and the reporting and assessment of monitoring data. This allows a full assessment of compliance by the environmental protection authority. Where compliance is below an acceptable level, the authority will focus additional regulatory effort to ensure the activity returns to compliance. Where failures are significant and/or persistent, the authority may use more formal enforcement tools to ensure compliance, and, in addition and if appropriate, criminal proceedings may be initiated.

Regulatory Monitoring and Self-Monitoring

Compliance monitoring can be carried out by both the operator of the activity and the environmental regulating authority. Regulatory agencies support the process of compliance monitoring by providing clear guidance and requirements to the applicant to allow a straightforward assessment of compliance with environmental standards. Where the operator carries out the compliance monitoring, it is a good practice for the regulator to carry out spot checks (using a risk-based approach) to verify that the operator reporting is accurate or to support enforcement action where noncompliance is reported.

Future Challenges

With a growing demand for environmental and ecological services and resources, there is increased pressure on regulatory authorities to maintain their assessment of compliance. New environmental pressures through changes in economic drivers and developments in industry require updates of existing regulatory guidance and the implementation of new compliance controls. New legislation and government policy directions may also introduce new demands on the operator to demonstrate compliance and conform to new environmental regulations. Pressures on wetland habitats are extensively researched, and there are already a number of threshold values which are used for understanding the level of pressure which triggers deterioration or damage in a wetland. However, some knowledge gaps remain, and there is still demand for primary research to inform future monitoring strategies and the implementation of accurate and realistic compliance parameters.

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Gauging Networks for Wetland Monitoring

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Seb Buckton

Contents

Abstract

Participatory monitoring of wetlands can provide a cost-effective alternative to conventional scientific monitoring which suffers from a number of constraints (such as high costs, difficult logistics of implementation, and sometimes scientific emphasis with little relevance to management). There is evidence that participatory monitoring provides a powerful complementary approach that enhances conservation management interventions even where conventional monitoring is already taking place. It provides relevant information for management, promotes participation of local people in management, and can be sustained using local resources. Generic methods that are suitable for participatory monitoring include patrol records, simple transects, species lists, on-

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the-ground photography, and village group discussions. A limitation for the application of these methods is that in some situations, conventional professional monitoring is required under national or international conservation legislation. Examples of participatory monitoring from the UK, USA, Australia, and Madagascar are given.

Keywords
Wetland monitoring · Community participation · Conservation management

Introduction

Although wetland monitoring is an essential tool in assessing the impacts of actions on wetlands and their biodiversity, it can be a time-consuming and expensive activity that consumes resources from conservation management organizations (Danielsen et al. 2005). Exploring alternative ways of undertaking wetland monitoring utilizing networks of people (participatory monitoring) can help reduce these costs. Participatory monitoring is a relatively recent trend. In the past, wetland monitoring was perceived as only the realm of experts. More recently the benefits of involving the wider public in wetland monitoring have been increasingly recognized.

Participatory monitoring can also be a useful tool in engaging people in conservation activities and raise awareness of the benefits wetlands bring to people, including biodiversity, as well as awareness of the threats to wetlands and their biodiversity. There are both positives and negatives to participatory monitoring, and therefore, it is important to gauge the utility of a network of people for undertaking wetland monitoring.

Constraints to Conventional Monitoring

Monitoring is an essential tool in gathering the evidence needed in conservation management (Sutherland et al. 2004). However, conventional natural resource monitoring schemes often face a number of important challenges (Danielsen et al. 2005):

- The running costs for professionally monitored schemes are often high.
- The high costs mean they may not be sustained over time.
- They can be difficult to implement logistically and can be technically and analytically complex.
- Monitoring is often perceived as being irrelevant to the needs of resource managers.
- Professional monitoring can be perceived as paying insufficient attention to stakeholders other than the scientists implementing the monitoring.

Benefits of Participatory Monitoring

Locally based monitoring of natural resources embraces a broad range of approaches, from self-monitoring of harvests by local resource users themselves to censuses by local government staff, or inventories by amateur naturalists. In all of these approaches, the monitoring is carried out at a local scale, although these may be coordinated at a national scale where resources permit. They often involve individuals with no or only limited formal science training, although their knowledge of the local environment may be considerable.

There are many potential benefits of organizing locally based monitoring of natural resources (Danielsen et al. 2005). They:

- Provide relevant information for management actions
- Can be sustained using locally available resources
- Promote participation of local people in management
- Stimulate discussion about natural resource management among stakeholders
- Build the capacity of field government staff and communities in management skills
- Seek to provide people with direction regarding the aims of sustainable resource management
- Reinforce the consolidation of existing livelihoods through strengthening community-based resource management systems

Participatory Monitoring Methods

In a review of participatory monitoring programs in developing countries, Danielsen et al. (2005) described the field methods used. There was variation between schemes, depending on the local circumstances, but five generic methods appeared particularly suitable:

1. Patrol records, where routine patrols involve completing sheets on key resources, habitats, or extent of resource exploitation
2. Simple dedicated transects allowing monitoring of resources and human resource use
3. Species lists – as presence/absence of resources on fixed-time lists
4. Simple on-the-ground fixed point photography
5. Village group discussions, involving government staff and local volunteer members of “community monitoring groups”

Some of these methods, or adaptations of them, are also used by participatory monitoring schemes in other countries. For example, the UK Wetland Bird Survey (see below) uses species lists (where they are combined with counts of key species); the UK Wider Countryside Butterfly survey involves volunteers recording butterflies along 1 km transect lines (Brereton et al. 2011). Other methods are more likely to be carried out by statutory agencies or the organization responsible for managing the site – e.g., records kept by reserve wardens.

Limitations of Participatory Monitoring

In some situations, there is a statutory requirement for monitoring activity to be carried out to a certain prescription which may require trained monitoring staff. This may be required for sites designated under national, regional, or international legislation. In the UK, the statutory nature conservation agencies have a responsibility for the identification and protection of sites intended to conserve important wildlife and earth science features. These may be designated under national legislation (e.g., Sites of Special Scientific Interest, SSSIs), European Directives (Special Areas of Conservation (SACs) and Special Protection Areas (SPAs)), or International Conventions (Ramsar Sites). Monitoring of the features for which the site was designated is a key requirement for statutory conservation agencies.

In the UK, JNCC (2004) sets out a set of monitoring protocols ("Common Standards Monitoring"), designed to act as a simple, quick assessment of the condition of the relevant features. This sets out the methodology that should be followed for monitoring features of protected sites (SAC, SPA, Ramsar, SSSI) and is supported by limited, more detailed monitoring. JNCC (2004) suggests that it is not possible to restrict the attributes being monitored just to those that can be recognized by untrained people. Botanical expertise may be required for many relevant features, as is sufficient knowledge of wetlands, so that structures and functions can be properly interpreted and likely water sources can be identified. These requirements need to be understood when identifying the resources necessary for monitoring and for training staff.

Examples of Participatory Monitoring

There are many examples of participatory monitoring programs at different scales, from around the world. A useful review is in Danielsen et al. (2005), while some examples of national and local schemes follow.

UK

The Wetland Bird Survey (WeBS) is a national wetland monitoring scheme that involves around 3,000 volunteers counting non-breeding waterbirds at hundreds of wetlands sites around the UK. The aim is to provide the principal data for the conservation of wintering waterbird populations and wetland habitats. The data collected are used to assess the size of waterbird populations, determine trends in numbers and distribution, and assess the importance of individual sites for waterbirds, in line with the requirements of international conservation Conventions and Directives. The scheme began in 1947, and volunteer counters participate in synchronized monthly counts at wetlands of all habitat types, mainly during the winter period. WeBS is administered by the British Trust for Ornithology, in partnership with the Royal Society for the Protection of Birds and the Joint Nature Conservation Committee and in association with the Wildfowl and Wetlands Trust (http://www.bto.org/volunteer-surveys/webs).

USA

The US Environmental Protection Agency provides resources to support volunteers who are monitoring the condition of their local streams, lakes, estuaries, and wetlands. There are multiple local participatory monitoring programs in the USA. For example, the Wetland Health Evaluation Program in Minnesota, which engages citizen volunteers to help monitor wetlands in their community. They provide important information to city and county planners, engineers, resource managers, and others. The data are also used by the Minnesota Pollution Control Agency to track wetland health throughout the Twin Cities metropolitan area (http://www.mnwhep.org/).

The Rock River coalition aims to provide opportunities for people of diverse interests to work together to improve the Rock River Basin, Wisconsin. They organize community wetland and community stream monitoring programs (http://www.rockrivercoalition.org/).

Australia

Natural resources management (NRM) is an approach that aims to work together with the community, industry, and three tiers of government to manage the environment in a way that achieves a balance between the collective need for resources and the needs of the environment. In Australia, the approach is enshrined in the Natural Resources Management Act 2004. Natural Resources SA Murray-Darling Basin is an agency that delivers natural resource management programs in the Murray-Darling Basin in South Australia. They provide resources for volunteer monitoring activities and organize community wetland monitoring days, where the public are invited to learn more about the wetlands of the region and to have a go at monitoring activities such as checking surface and ground water quality or recording birds (http://www.naturalresources.sa.gov.au/samurraydarlingbasin/news/130603-get-involved-wetlands-monitoring).

Madagascar

A participatory monitoring program was established in 2001 by the Durrell Wildlife Conservation Trust for the Alaotra wetlands in Madagascar. Government and nongovernment organizations and local community members collect information on key species and useful natural resources. The program has been instrumental in raising awareness of the importance of the wetlands and existing management systems and has catalyzed new systems of management (Andrianandrasana et al. 2005).

Future Challenges

Participatory monitoring can provide a cost-effective alternative to conventional scientific monitoring. There is evidence that it also provides a powerful complementary approach that enhances conservation management interventions even where conventional monitoring is already taking place (Danielsen et al. 2007). Although there will remain instances where a high degree of expertise or training is required to monitor certain features, the benefits of participatory monitoring are likely to make it an increasingly attractive option.

An important challenge, therefore, will be addressing potential difficulties arising from a dependence on community-based participatory monitoring. One of these relates to determining the factors that influence stakeholders in deciding to adopt a monitoring program and how much they may be expected to contribute. Hockley et al. (2005) predicted that the extent to which communities will contribute to a monitoring program will be affected by their dependency upon the resource being monitored, its cultural importance, vulnerability to overexploitation, and amenability to monitoring. Issues relating to factors influencing collective action of this sort (e.g., Poteete and Olstrom 2004) are also important considerations for ensuring sustainability of participatory monitoring activities.

The increasing emphasis on landscape-scale projects where the aims are to promote wider biophysical processes in the landscape, often involving an ecosystem approach, is likely to have less certain outcomes (Stroh and Hughes 2010), and therefore, devising participatory monitoring programs will present additional difficulties.

There is evidence that participatory monitoring not only provides a cost-effective alternative to conventional scientific monitoring but that it provides a powerful complementary approach that greatly enhances the level of conservation management intervention where conventional monitoring is already being implemented (Danielsen et al. 2007).

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Wetland Monitoring: Reporting

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Contents

Abstract

Approaches for reporting on monitoring data are presented, focusing on indicators, types of reports (e.g., snapshot, scorecards), scales of reporting (from single wetland to national and international scales), and communication of results. Indicators are symbolic representations for communicating a property or trend in a system. They are used to assess current status, monitor or predict changes in condition, or detect sources of stress. Wetland indicators include indicators for wetland health, fauna and water quality indicators, and indicators describing ecosystem services and economic value. Condition reports describe the state of selected, usually biophysical indicators, often in qualitative terms ("poor," "average," "good") and at the time at which the data was collected. Temporal data is

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used to report on trends in wetland condition over time. Reporting trends, especially over longer time periods, is critical for separating the impact of human activities from external climatic drivers. The Ramsar Convention requires that ecological character be reported for each listed wetland. While the unit of analysis is often a survey plot within a wetland or a wetland complex, wetland and water managers typically operate at a watershed or ecoregional scale, so reporting of wetland monitoring and assessment is also often at this scale or at the scale of the administrative unit of interest. Reporting needs to communicate methods and indicators for wetland condition and trends effectively to the target specialist or nonspecialist audiences, and the simplicity or complexity of indicators used needs to be appropriate. Reporting needs to address error, uncertainty, and overall confidence in measurements of wetland condition to provide decision makers with a sense of the reliability of the data on which decisions are based.

Keywords
Wetland Management · Time-Series Data · Condition and Trend · Ecosystem Services

Introduction

Wetland managers are typically faced with scarce yet highly contested water resources. They require robust monitoring of wetland condition and trend for prioritizing restoration and management of wetlands, to target new wetland conservation programs and measure performance of existing programs. Accurate and defensible reporting of wetland monitoring efforts are the information from which wetland managers, and others, make decisions about the efficacy of current investment and the priorities for future investments in wetland management, restoration, and rehabilitation. For example, a wetland manager may want to know the success of actions aimed at halting or reversing the decline of wetland health, while a government policymaker may want to know the return on investment of public money into wetland rehabilitation and restoration. In this chapter, we discuss approaches for reporting on monitoring data, focussing on indicators, types of reports (e.g., snapshot, scorecards), scales of reporting (from single wetland to national and international scales), and communication of results.

Indicators

Indicators are symbolic representations for communicating a property or trend in a complex system or entity (Moldan and Dahl 2007). In the case of wetland monitoring, indicators communicate the specific components and processes of a wetland measured to assess current status, monitor or predict changes in condition, or detect sources of stress (Wardrop et al. 2007; Jacobs et al. 2010; Kotze et al. 2012). Indicators for reporting the condition and trend of wetland health typically fall into one of three categories: (i) hydrology; (ii) vegetation, and (iii) soils (Cole and Kentula 2011). Fauna and water quality indicators can also be used to describe biophysical attributes. More recently, indicators describing ecosystem service and economic value aspects of wetlands are used to link wetland health to human well-being (Stratford et al. 2011; Banerjee et al. 2013; Euliss et al. 2013; Ringold et al. 2013). Composite indicators, such as indices of biological and ecological integrity, are powerful tools for monitoring and reporting the response of wetlands to stressors and human impacts, and the efficacy of wetland protection, restoration, and mitigation activities (Veselka and Anderson 2013).

For the most effective monitoring, a consistent set of indicators should be collected over time, and they should capture the spatial heterogeneity of the wetland (Cole and Kentula 2011). Furthermore, monitoring and associated indicators will be more useful if there is comparison back to a reference site or wetland that describes the best and least impacted condition. The reporting of wetland condition and trend in relation to a reference condition provide a benchmark for assessments of change in condition, which can be used to identify the priority of restoration and mitigation activities, perhaps using a triage approach (Brooks et al. 2006).

Types of Reporting

Condition Reports

A number of reporting types are commonly used to summarize and present information about wetland condition and trend indicators collected for a wetland monitoring program. One of the simpler types of reports is the single snapshot condition reporting. Condition reports describe the state of selected, usually biophysical, indicators and typically use qualitative terms such as “poor,” “average,” and “good,” or even simpler ticks and crosses. These reports describe health at the time at which the data was collected and do not describe temporal dynamics. For example, the Sustainable Rivers Audit of the health of freshwater ecosystems in Australia’s Murray-Darling Basin (Davies et al. 2010) measures indicators of condition in five themes (hydrology, fish, macro-invertebrates, vegetation, and physical form), as well as computing an ecological health score, and compares results back to a reference condition. The Sustainable Rivers Audit reports condition using both qualitative and quantitative measures, including confidence intervals of sampled data to report uncertainty. For example, a number of indicators can be measured and scored to form a combined “Fish Index” and qualitative labels are assigned to ranges of the index for more meaningful reporting (Fig. 1).

The Sustainable Rivers Audit aims to link human interventions to ecological condition variables that are sensitive to human impacts. By making the linkages explicit, managers of Murray-Darling Basin wetlands can monitor the improvement, or otherwise, that may arise from changes to human activity. However, temporal data is required to do this most effectively.

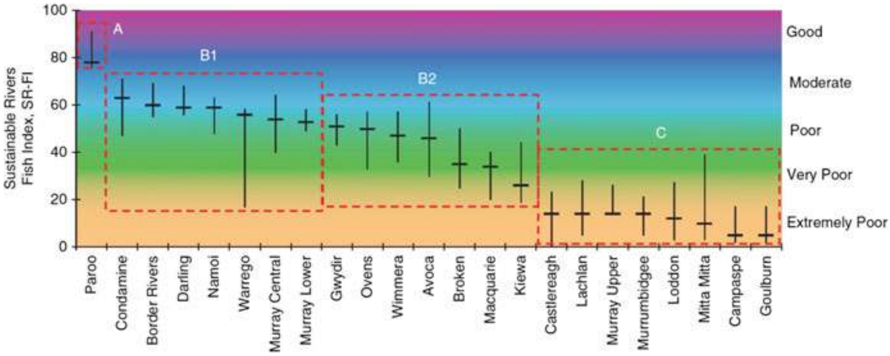

Fig. 1 Condition of major river reaches in the Murray-Darling Basin measured using the Fish Index. Left axis shows quantitative score and right axis shows qualitative value in comparison to a reference condition. Horizontal bars are medians and vertical lines show associated 95% confidence limits (Reproduced from Davies et al. (2010), with permission from CSIRO Publishing)

Trend Reports

Temporal data is used to report on trends in wetland condition over time. Reporting trends, especially over longer time periods, is critical for separating the impact of human activities from external climatic drivers that may impact condition, such as drought and annual wetting and drying cycles. A recent study by Colloff et al. (2015) reviewed published long-term datasets on condition of wetlands and other flow-dependent ecosystems in the Murray-Darling Basin and demonstrated the importance of reporting long-term data for understanding the impacts of multiple anthropogenic and climatic stressors. The US Fish and Wildlife Service has been collecting long-term trend data on the extent of wetlands in the conterminous United States since the mid-1950s (http://www.fws.gov/wetlands/Status-and-Trends/). They report on the changes in spatial extent of wetlands for each 5–10 year time period since data collection began, with the most recent report covering the period 2004–2009 (Dahl 2011). While the US Fish and Wildlife Service do not report on condition, the US EPA National Wetland Condition Assessment aims to collect national 5-yearly time-series data of wetland condition, with the first assessment completed in 2011 (http://water.epa.gov/type/wetlands/assessment/survey/). The UK Countryside Survey, a representative survey of the UK rural environment at 5–10 year intervals since 1978, has collected "pond" wetland extent data since 1984 and since the mid-1990s has collected condition data for some lowland "pond" wetlands. The UK survey collects many biophysical indicators and reports on physical and chemical condition, and biodiversity and biological quality (Williams et al. 2010).

The use of traffic lights (red, amber, green, or variants) or arrows pointing up or down, or some combination of the two, are techniques commonly used to report trend (Fig. 2). The combination of traffic light colours to describe condition and arrows to describe trend is a very powerful communication tool because it can very simply convey a lot of complex information.

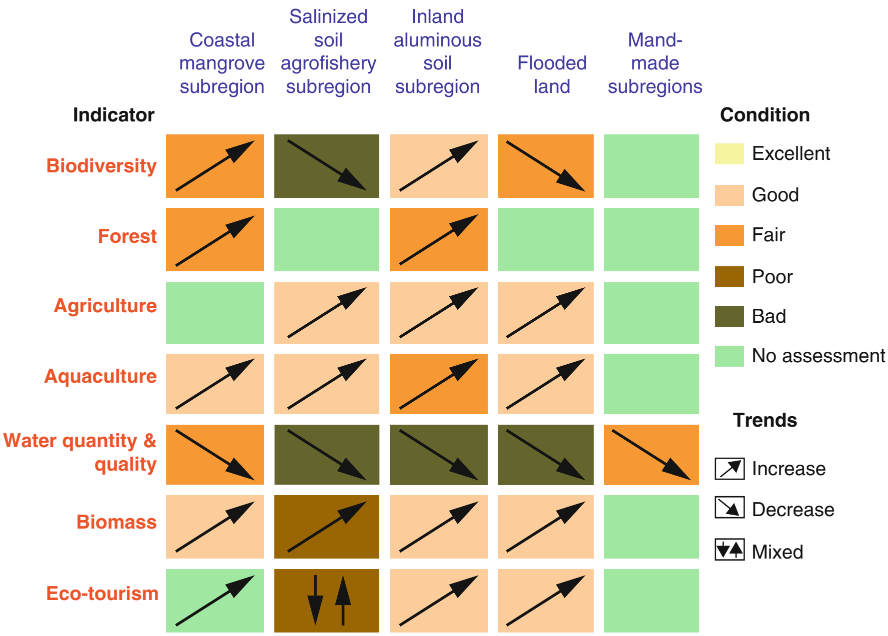

Fig. 2 Status and trends of key ecosystem components of a wetland monitoring program in the Cuu Long River Delta Wetland in a delta region of the Mekong River, Vietnam (Source: Thong et al. (2013); used with permission from Journal of Environmental Science and Management, University of the Philippines Los Baños)

Scales of Reporting

While the unit of analysis is often a survey plot within a wetland or a wetland complex, reporting to support decision making tends to be at the scale of the administrative unit of interest. For example, the US EPA National Wetland Condition Assessment, while using stratified random sampling site scale methods (1179 × 0.1–0.5 ha Assessment Areas), reports its headline components of wetland condition and biological integrity at the very coarse national scale (conterminous 48 States). Wetland and water managers typically operate at a watershed or eco-regional scale, so reporting of wetland monitoring and assessment is also often at this scale. For example, the Sustainable Rivers Audit reports on condition of key attributes for each of the 23 major river reaches in the one million km^2 Murray-Darling Basin (Davies et al. 2010) because decisions about volumes of water extraction for consumptive use or water for environmental purposes are made at the this scale by the Federal Government agency in charge of implementing the water sharing plan.

At the other end of the scale range, the Ramsar Convention requires that ecological character be maintained at each listed wetland (Ramsar Convention 2005). To do this, managers of Ramsar-listed wetlands should measure the components, processes, and benefits and services that characterize the wetland to provide the baseline description of the wetland, and over time identify unacceptable change in ecological character (Davis and Brock 2008). The Montreaux Record of the Ramsar Convention maintains a list of wetlands where changes to ecological character have occurred, are now occurring, or are likely to occur as a result of human impacts. Summary descriptions of ecological character for Ramsar sites are available from the Ramsar website. However, the detailed reporting of changes to ecological character (past, present, or future) is ad hoc and dependent on suitable national reporting mechanisms (Finlayson 2012).

Communication and Target Audiences

Because reporting is by definition a communication exercise, methods and indicators used to monitor and assess wetland condition and trend will be of little use if they fail to communicate effectively to the target audience. The simplicity or complexity of indicators used to report on condition and trend of wetlands should therefore be determined by the target audiences. For example, nonspecialists such as the broader public and media need indicators and reports that are simple and well structured, and may only want to read a popular science style report with a very small number of graphical summaries of key indicators and their trends (e.g., Fig. 2). Others, such as wetland managers and policy specialists, may require shorter reports with more detailed figures and tables summarizing condition and trend, whereas those with deep technical knowledge and interest, such as researchers, may want to access and interrogate original data.

It is very important to report error, uncertainty, and overall confidence in measurements of wetland condition to provide decision makers with a sense of the reliability of data used to support their decisions. For example, many different types of error can exist in a wetland condition study, from sampling errors, to measurement errors, to observer bias, and more (Haukos 2013). Careful study design which includes a robust sampling effort will go a long way toward mitigating error. However, reporting of potential error using statistical tests and confidence bars (e.g., Fig. 1) will mitigate the risk of poor decisions.

Future Challenges

We suggest two major future challenges. Firstly, a challenge is to implement new monitoring and assessment programs of time series data collection as well as maintain existing programs. There is increasing pressure on operating budgets of public agencies charged with undertaking wetland monitoring, assessment, and reporting. Time series data collection is expensive and consequently considerable pressure exists to reduce existing programs and few opportunities exist for new programs to collect long-term wetland condition data. The dearth of detailed data for many Ramsar wetlands of international significance is a good example of how national reporting mechanisms are failing (Finlayson 2012).

Secondly, a challenge is to report on more ecosystem service trends in wetlands. Ecosystem services reporting needs to extend further into the ecosystem services approach. Examples such as the Ramsar Convention's ecological character description and the US EPA National Wetland Condition Assessment include a number of indicators capturing the supporting and regulating ecosystem services (Ramsar Convention Secretariat 2013; Ringold et al. 2013). But there are a number of provisioning and cultural ecosystem services that are rarely measured and reported in wetland monitoring and assessment programs. More effort is required to accurately measure the less easily quantifiable services in the cultural category. The provisioning and cultural ecosystem services are most recognizable and best understood by people. Better reporting of the trend of supply of those services may strengthen the arguments for increased investment in protection and restoration of wetlands.

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Section XVII

Environmental Flows

Michael C. Acreman

Environmental Flows: Overview

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Michael C. Acreman and Angela H. Arthington

Contents

Abstract

The term environmental flows relates to the trade-off between keeping water in a wetland system to meet ecosystem requirements and services to dependent people (such as food, recreation, and cultural identity) versus realizing the direct benefits of removing the water for drinking, growing food, and supporting industry. It describes the quantity, quality, and timing of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems. Environmental flows is a fundamental part of water management.

Keywords

Water management · Water allocation · Environmental water need · Environmental benefits

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Introduction

The term environmental flows relates to the trade-off between keeping water in a wetland system to meet ecosystem requirements and services to dependent people (such as food, recreation, and cultural identity) versus realizing the direct benefits of removing the water for drinking, growing food, and supporting industry (Acreman 1998). It describes the quantity, quality, and timing of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems (Brisbane Declaration http://www.eflownet.org/). The environmental flow concept is a fundamental part of water management.

There are four basic challenges to environmental flows: (1) the political decision of recognizing a need to maintain sufficient water in rivers and wetland systems; (2) defining the flow requirements of the ecosystem; (3) the practical issues and costs of implementation to achieve the environmental flow regime required; and (4) the design and maintenance of robust monitoring systems to assess the ecological and other outcomes of environmental watering in an adaptive management framework.

The majority of environmental flow studies have been on flow (discharge) regimes and on river and floodplain wetlands, so this is the focus of this text. A separate note describes studies of water level requirements of other wetland types.

History of Environmental Flows

Water releases have been made from reservoirs since the 1800s, but originally these "compensation flows" were intended to provide water for downstream riparian users including villages, mills, and navigation, with minimal attention to impacts on river ecosystems. The first environmental flows focused on the concept of a minimum flow for diluting polluted discharges to rivers, based on the notion that as long as the flow is maintained at or above a critical minimum, the river ecosystem will be conserved (Tharme 2003; Acreman et al. 2014a). For example, the UK Water Resources Act 1963 required minimum acceptable flows to maintain natural beauty and fisheries. The US Clean Water Act of 1972 set the objective of restoring and maintaining the chemical, physical, and biological integrity of the nation's waters. Water allocations for ecosystem maintenance have been incorporated into river basin water management plans (e.g., Australia's Murray-Darling Basin Plan; Australian Government 2012), Integrated Water Resources Management (IWRM; Falkenmark 2003), environmental impact assessment (Wathern 1998), and the Ecosystem Approach (Maltby et al. 1999).

Environmental flows are now enshrined in the policies and laws of many countries (le Quesne et al. 2010). South Africa's water law first recognized that water for the maintenance of the environment should be accorded the highest priority (Rowlston and Palmer 2002; King and Pienaar 2011) after basic human needs. The concept of environmental flows is now a key element in many international policies (such as the Convention on Biological Diversity signed by 194 parties and the Ramsar Convention on Wetlands signed by 168 parties) and integrated into the water law in many other countries, e.g., Costa Rica (Jiménez Ramón et al. 2005), Tanzania (Acreman et al. 2006), and Australia (Kildea and Williams 2011) and the European Water Framework Directive (Acreman and Ferguson 2010). Environmental flow management has also become a central part of the policies of major institutions, including the World Bank (Hirji and Davis 2009) and IUCN (Dyson et al. 2003).

Environmental Flows Concept and Methods

There is no simple answer to how much water a wetland ecosystem needs. Environmental flows are set for river reaches, individual wetlands, river basins, or groups of rivers within a biogeographic or political region using a range of frameworks and specific methods. Tharme (2003) identified 207 methods for determining environmental flows and classified them into four main groups – hydrologic, hydraulic rating, habitat modeling, and holistic/ecosystem methods (see Acreman and Dunbar 2004; Arthington 2012, for details). Recently, Acreman et al. (2014b) identified two basic environmental flow approaches based on either (1) constraining alteration from a natural flow baseline to maintain biodiversity and ecological integrity or (2) designing flow regimes to achieve specific ecological and ecosystem service outcomes.

The recent definition of environmental flows described above suggests the need to link river flows explicitly with ecosystem service outcomes and human well-being. This idea underpins the Building Block Methodology initiated during South African-Australian collaborations (Arthington 1998; King and Louw 1998) in which the flow requirement of each biophysical component of the wetland ecosystem (e.g., fish, invertebrates, plants) is analyzed individually and then combined to produce an environmental flow regime. This approach was developed further in Australia (Arthington and Pusey 2003) and the UK (Acreman et al. 2009; Fig. 1) and recommended (UKTAG 2013) for defining environmental flow releases from reservoirs to meet the objectives of the European Water Framework Directive (Acreman and Ferguson 2010). It has been taken up and elaborated by other countries such as Norway (Alfredson et al. 2012) and China (Acreman et al. 2010).

A limitation of the Building Block approach is that it is not easy to identify all the elements of the ecosystem that must be maintained to ensure continued viability (for example, algae and microbes, aquatic/ riparian vegetation, and semiaquatic vertebrates are often neglected), and their flow requirements may not be known.

The natural flow regime paradigm (Poff et al. 1997) has stimulated an alternative perspective on environmental flows. It is based on the argument that organisms evolved and adapted, and communities were assembled and are maintained, by a naturally dynamic flow regime (Lytle and Poff 2004). One or more organisms will have exploited each ecosystem niche generated by the flow regime, such that all elements of the flow hydrograph are important – floods, medium and low flows – including their magnitude, timing, duration, and frequency. Thus any modification of the natural flow regime will alter riverine, riparian, and floodplain communities and processes, and there may be limits to hydrological change beyond which significant

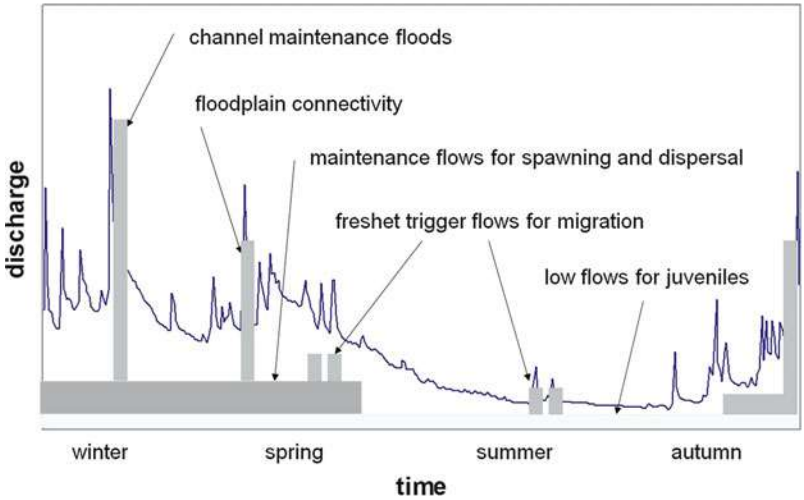

Fig. 1 Basic elements of flow regime required to deliver components of the river ecosystem superimposed on the recorded flow regime (After Acreman et al. 2009)

(or unacceptable) ecological alteration takes place (Richter et al. 1996; Bunn and Arthington 2002; Arthington et al. 2006). Within this paradigm it is not always necessary to link elements of the flow regime explicitly to components of the river ecosystem, as all elements are considered to be important. Olden and Poff (2003) concluded that 66 hydrological metrics were required to describe adequately most of the major flow-regime components that influence the structure and processes of river ecosystems. However, only a small number of variables is required to characterize a flow regime statistically, as many of the scores of available hydrologic variables show high redundancy (Olden and Poff 2003). Furthermore, some argue that not all indices are "ecologically relevant," i.e., are known to have, or can be hypothesized to have, some demonstrated or measurable ecological influence (Arthington et al. 2006).

A central question for environmental flow science is – how much change in each hydrological variable (and in the composite flow regime) is acceptable? Richter et al. (2011) suggested a precautionary standard of 10% alteration in any flow variable from natural flows to afford a high level of ecological protection, in the absence of detailed site-specific data, though it was recommended that specific threshold values should be defined for the wetland under study. Acreman et al. (2008) used this approach to define maximum abstractions from UK rivers ranging from 7.5 to 35% of the natural flow depending on river type and flow rate. These limits were incorporated into the Environment Agency's abstraction licensing program (Environment Agency 2001) for England and Wales.

Indicators of hydrological alteration (e.g., IHA, Richter et al. 1996) have been used widely to assess the risk of river ecosystem change under different scenarios. A risk map was produced for all major European rivers to assess the implications of projected future flow regimes in 2050 resulting from climate change and water use

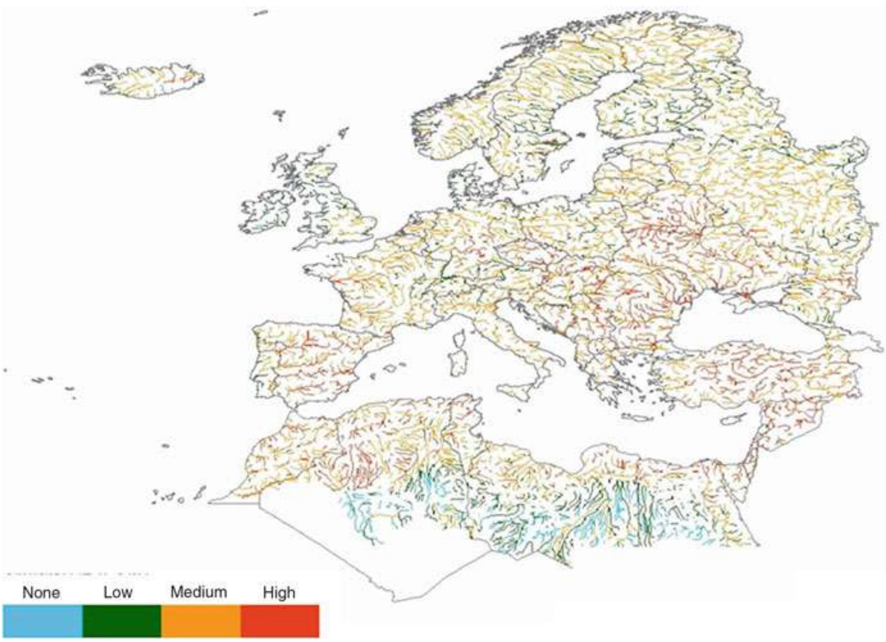

Fig. 2 Risk of river ecosystem change in European rivers by 2050 (after Laizé et al. 2014)

(Laizé et al. 2014; Fig. 2). Similar analyses have been undertaken for other regions such as the Mekong basin (Thompson et al. 2014). Some ecologists question the use of a fixed natural flow regime as the appropriate baseline in regions where climates and flow regimes are predicted to change (Acreman et al. 2014b). In addition, some rivers have been managed in particular ways for thousands of years. In such cases, the baseline for further study and manipulation of the flow regime may be recent recorded flows prior to a new hydrological impact (Booker et al. 2004a).

The two concepts of the natural flow paradigm and explicit flow-ecosystem relationships were brought together in a generic framework entitled the Ecological Limits of Hydrologic Alteration (ELOHA) designed to synthesize hydrological and ecological databases from many rivers within a user-defined region to develop scientifically defensible and empirically testable relationships between flow alteration and ecological responses (Poff et al. 2010). ELOHA is based on concepts presented in Arthington et al. (2006) and uses response curves that relate flow alterations to fish, invertebrate, or plant communities in rivers of distinctive hydrological character (defined by classification). Because external forces on ecosystems tend to occur in synchrony rather than as individual pressures (Ormerod et al. 2010), and may be additive, synergistic, or antagonistic (Acreman et al. 2014a), response curves may vary between river classes and between natural and regulated rivers within classes. Dunbar et al. (2010) analyzed macroinvertebrate data for numerous rivers across England and found a strong interaction between flow and channel morphology in determining ecological response. Figure 3 shows that the flow-

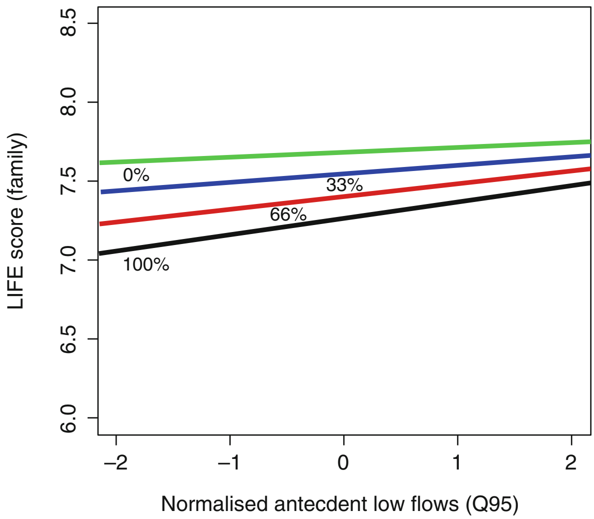

Fig. 3 Variations in relationships between flow (normalized low flow Q95) and river ecosystem score (LIFE of macroinvertebrates) with channel alteration (given as % modification from natural)

ecosystem relationship for natural channels (0% modification) has a shallow slope showing that the river ecosystem is robust, or insensitive, to flow change, whereas the relationship for modified channels has a steeper slope demonstrating that the river ecosystem is more sensitive to flow change in highly altered channels (<100% modification). This is because natural rivers tend to have diverse morphology, so that whatever the flow, there will be a diversity of physical habitats (deep, shallow, fast, slow). In contrast, modified river channels, such as those straightened and deepened for flood defense, tend to have homogenous, often trapezoidal, morphology in which good habitat only occurs at certain flows. Even multiple pressure research has focused primarily on external drivers, whereas internal ecosystem processes, such as biotic interaction and trophic relationships that govern flows of energy and carbon, may also control ecosystem type and condition. Alterations to single external pressures, such as flow regime, may interact in complex ways with these internal ecological processes (Shenton et al. 2010) and require further investigation. There is a need for biologists and ecologists to work more closely with hydroecologists to address the challenges of combining flow effects with internal ecosystem dynamics.

As the name suggests, at a broad level environmental flows are focused on discharge (m^3s^{-1}) as the volume of flow along a river over time regulates, inter alia, the downstream provision of nutrients and energy that drive ecosystem productivity. However, it is essential to understand other roles of river flows, such as the provision of physical habitats, including features of importance to biota, such as wetted area, water depth, velocity, heterogeneity, and connectivity of habitat patches (Bunn and Arthington 2002). Ecohydraulics has developed as a subtheme of environmental flows to address the interactions of flow, channel morphology, and substrate characteristics (James and King 2010; Maddock et al. 2013). Recent innovations include the development of hydrological indicators of hydraulic conditions that drive flow-biota relationships (Turner and Stewardson 2014).

The Physical Habitat Simulation (PHABSIM) computer model developed by the US Fish and Wild Life Service (Waters 1976) produces a relationship between river flow (discharge) and physical habitat available for target species, such as trout and salmon (Bovee 1982). The use of this approach is a legal requirement for setting environmental flows in rivers in some US states. PHABSIM has been used in the UK to assess the habitat implications of channel modification in restoration schemes, such as the River Wey (Acreman and Elliott 1996) and effects of channelization in urban areas of the River Tame (Booker et al. 2003). Derivatives of PHABSIM have been widely used in other countries (Parasiewicz and Dunbar 2001) including France, Norway, New Zealand, Australia, and Germany. Habitat modeling approaches have diversified, for example, to consider flow management for whole catchments, such as the River Itchen (Booker et al. 2004a), rather than river reaches for which PHABSIM was originally devised.

Habitat suitability is typically measured empirically by recording the depth and velocity of the river at points where fish are usually located and could be impacted by reduced flows. Bioenergetic modeling has demonstrated causal mechanisms for hydraulic habitat selection, for example, feeding salmonid fish preferentially select areas of river where high velocity provides a net energy gain when drift feeding, but move to lower velocity areas when resting (Booker et al. 2004b). A limitation of PHABSIM is that it requires collection of hydraulic data (river depth and water velocity) at not less than three flows (high, medium, and low) at a set of cross-sections (normally ten or more) along the impacted river reach (Waddle 2012). However, methods of rapid habitat assessment have been produced based on measurements of river width and depth that could be collected during one short site visit (Booker and Acreman 2007; Lamouroux and Jowett 2005).

Implementation of Environmental Flows in Management Regimes

Despite the large number of published flow-ecology studies, models, training courses, and applications to countless locations, there remains a global lack of implementation of environmental flows (le Quesne et al. 2010). Historically, much water has been allocated to direct human use, and many abstractors have rights to water that would need to be given up to put sufficient water back into rivers and wetlands. In the UK, compensation flows from reservoirs were set by individual Acts of Parliament, requiring a major political process to make alterations. Relinquishing water from irrigation or industry to achieve environmental flows may have major economic implications. In Tanzania, water from the Kihansi River is diverted to generate hydropower that provides one third of the country's electricity. Rediverting water back to the river, to protect the rare and endemic Kihansi spray toad, would mean reduced industrial expansion and lower economic growth in the country (Hirji and Davis 2009). Furthermore the beneficiaries of environmental flows may be different than the beneficiaries of other water uses, creating difficult political choices. For example, on the Senegal River, beneficiaries of the hydropower generation at Manantali Dam are urban dwellers in major West Africa cities (Dakar, Bamako, Nouakchott), whereas beneficiaries of the environmental flow releases are farmers and herders on the Senegal floodplain (Acreman 2003).

There also may be practical issues to successful implementation. Ensuing environmental flows defined as deviations from the natural flow regime require knowledge of what the natural flow would be. This necessitates real-time recording or computer modeling of flow at a reference point upstream of the abstraction point or reservoir that provides the natural flow information. Creating environmental high flows (such as to inundate a floodplain) is often more problematic than low flows, since reservoir releases, for example, need to coincide with high flows entering the river from tributaries to achieve target high flows downstream (at the floodplain). Much greater quantities of water would be needed if high flows are generated purely by reservoir releases (i.e., when tributary inflows are low).

When there is insufficient water to restore overbank floods, an emerging alternative is to use floodplain infrastructure – such as levees, weirs, regulators, and pumps – to control water levels within floodplain wetlands (Pittock et al. 2013). Dehydrated floodplain wetlands along the River Murray in Australia are being watered by pumping river water into flood runners and by spray and drip irrigation systems (MDBA 2011). While environmental infrastructure and engineered watering systems are proving beneficial for some local ecological communities (e.g., riparian vegetation, frogs, waterbirds), the natural biogeochemical exchanges and movements of native fish to and from floodplains can be vastly constrained by altered hydrological connectivity (Bond et al. 2014). Altered hydraulic and water regimes in engineered wetlands may favor spawning and recruitment of exotic fish species, such as common carp (Bice and Zampatti 2011).

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Environmental Flow Requirements Setting: Desktop Methods

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Denis A. Hughes

Contents

Abstract

Desktop approaches to setting environmental flow requirements (EFRs) have the potential to be useful in situations where resources (time, expertise, and information) are limited, but they will almost always provide estimates with lower confidence, or higher uncertainty, than more comprehensive assessments. There is a continuum between desktop approaches, rapid methods based on limited data collection, and comprehensive assessments involving field collection (and interpretation) of hydraulic, geomorphological, and biotic data. Under most situations, the resources required increase, and the confidence decreases, along the continuum. However, the decrease in confidence could be lower in poorly understood and complex systems because even comprehensive studies will be subject to resource constraints. Similarly, the confidence of desktop or rapid assessments could be increased given the availability of appropriate regional information and given that such information can be used within the less resource-intensive methods.

Keywords

Desktop methods · Environmental flows · Rivers

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Introduction

Desktop approaches to setting environmental flow requirements (EFRs) have the potential to be useful in situations where resources (time, expertise, and information) are limited, but they will almost always provide estimates with lower confidence, or higher uncertainty, than more comprehensive assessments. There is a continuum between desktop approaches, rapid methods based on limited data collection, and comprehensive assessments involving field collection (and interpretation) of hydraulic, geomorphological, and biotic data (Fig. 1; see also Acreman 2005). Under most situations, the resources required increase, and the confidence decreases, along the continuum. However, Fig. 1 also suggests that the decrease in confidence (and therefore the cost/benefit ratio) could be lower in poorly understood and complex systems because even comprehensive studies will be subject to resource constraints. Similarly, the confidence of desktop or rapid assessments could be increased given the availability of appropriate regional information (Arthington et al. 2006; Poff et al. 2010), given that such information can be used within the less resource-intensive methods.

Worldwide, the pace of implementation of EFRs has been slow (Acreman 2005; Richter et al. 2012), and therefore, the validity of any method of determination (in terms of achieving the desired ecological objective) remains largely untested. This problem is exacerbated by the costs of long-term monitoring to ensure

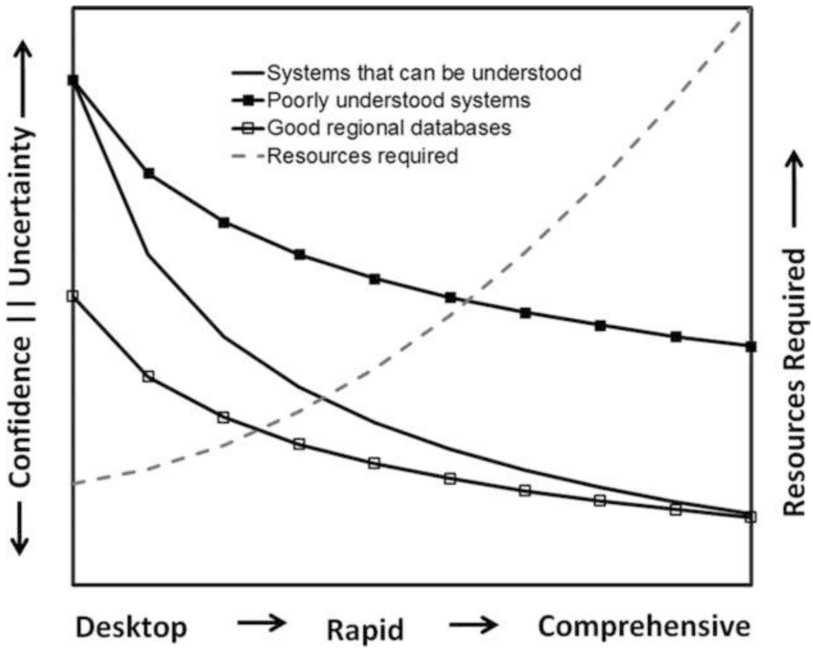

Fig. 1 Stylized relationships between methods of EFR determination, the resources required to complete them, and the level of uncertainty or confidence in the results

compliance both in terms of the EFRs themselves and the desired ecological objectives. Given that comprehensive determinations are time-consuming, expensive, and often relatively uncertain (Beven and Alcock 2011), an alternative approach is to use cheaper and more rapid desktop approaches to set and implement initial EFRs which can then be refined based on monitoring programs and an adaptive management approach. This could not only benefit individual site-specific determinations, but could also lead to refinement of the structure of desktop models and the manner in which they are applied (using default or regional parameter values) in different regions.

Desktop Approaches

Available desktop approaches have been based on purely hydrological data (Tennant 1976; Smakhtin et al. 2004, 2006), hydrological data with some implicit accounting for the relationships between flow and ecological response (Hughes and Hannart 2003; Poff et al. 2010; Richter et al. 2012), while others include the links between hydrology, hydraulics, and biotic response more explicitly (Maddock 1999; Hughes and Louw 2010; Hughes et al. 2014). Including more components into a desktop approach increases the input information requirements and therefore the potential for uncertainty if such information is lacking. However, one of the advantages of more components is that a desktop method can be better aligned, in terms of structure, with the type of approach that is used in more comprehensive assessments (Hughes et al. 2014). It is therefore possible that the same model could be used across the continuum of methods (Fig. 1), with more site-specific and reliable information being added in the move from desktop to comprehensive. This approach also allows for a feedback process when the model is applied in detailed or comprehensive assessments in that any model shortcomings can be identified and a database of regional model parameters can be developed for use in purely desktop applications.

All of the recently proposed desktop models support the natural flow paradigm (Poff et al. 1997) that proposes that increasing alterations of flow regimes away from the natural regime are likely to be associated with increasing risk to environmental sustainability and therefore decreasing levels of protection. Richter et al. (2012) refer to sustainability boundaries, which refer to bands of permitted or acceptable change from natural conditions, and represent a useful concept for integrating environmental protection and the socio-economic demands for water use. In some cases, deviations from the natural regime are measured by a multitude of hydrological factors (Richter et al. 1997), while others have focused on changes in the magnitude-frequency relationships of the flow regime using flow duration curves (Hughes and Hannart 2003; Smakhtin et al. 2004, 2006; Hughes et al. 2014). Another approach based on deviations of the frequency of available hydraulic habitat from natural conditions through links between hydrology and hydraulics presents the results as deviations from natural flow duration curves (Hughes 2014). Desktop approaches to setting environmental flows are meant to be pragmatic solutions to a water management problem, and therefore, the outputs have to be in a format that can be used by water managers for decision-making in both planning and operation (Acreman 2005; Hughes 2006; Hughes and Mallory 2008). Differences in the way in which water resources are managed in different parts of the world will therefore be reflected in different approaches to rapid desktop assessments, and there will never be a "best" international standard.

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Environmental Flows: Habitat Modeling

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Ian Maddock

Contents

Abstract

Aquatic habitat modelling is commonly used in riverine wetlands for environmen.tal flow assessments as a means of defining the empirical relationship between environmental variables, and usable habitat for selected target species, life stages or aquatic communities. Aquatic habitat modelling is used to determine environmental flow strategies by estimating the effects of historic, current or future flow scenarios on habitat availability. The origins of aquatic habitat modelling for determining environmental flows can be traced to the development of the Instream Flow Incremental Methodology (IFIM) and the set of computer programs required to implement an IFIM study, *i.e.* the Physical Habitat Simulation (PHABSIM) system. More recently, other models have used alternative approaches, such as CASiMiR (Computer Aided Simulation Model for Instream Flow and Riparia) that utilises a mix of expert opinion and fuzzy logic based rules to describe the habitat use of target species or the application of multidimensional, i.e., two-dimensional and to a lesser extent three-dimensional hydraulic-habitat models. These enable an enhanced representation of the hydraulic environment and allow the calculation and modelling of turbulent flow properties. However, a

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knowledge of how biota respond to and are influenced by these properties remains an ongoing challenge to incorporate them into environmental flow setting.

Keywords
IFIM · PHABSIM · CASiMiR · Habitat suitability curves · Hydraulic habitat models · Fuzzy logic

Introduction

Aquatic habitat modelling is commonly used in riverine wetlands for environmental flow assessments as a means of defining the empirical relationship between environmental variables, usually discharge, and usable habitat for selected target species, life stages, or aquatic communities (Fig. 1). Environmental variables can be simple physical properties of the channel, such as the wetted perimeter (Gippel and Stewardson 1998) as used by hydraulic rating methods, or can refer to the hydraulics of the river reach in question (e.g., water depths and velocities) and how these vary with discharge. Aquatic habitat modelling is used to determine environmental flow strategies by estimating the effects of historic, current, or future flow scenarios on habitat availability, for example caused by reservoir releases (Maddock et al. 2001), surface or groundwater abstraction (Olsen et al. 2009), or climate change (Ormerod and Durance 2012). The most common target species used in habitat modelling studies have been salmonid fish species because of their economic importance and sensitivity to flow (Dunbar et al. 2012), although cyprinid fish species (Costa et al. 2012), benthic macroinvertebrates (Maynard and Lane 2012), aquatic macrophytes (Janauer et al. 2013), and occasionally terrestrial species such as birds that utilize or are associated with wetland environments have also been used (Rodriguez and Howe 2013). In static or slow flowing wetlands, such as floodplain wetlands, fens, bogs, and salt marshes associated with estuarine and tidal wetlands, water depth rather than velocity, which is controlled by topography and water level fluctuations, becomes the primary hydraulic variable determining habitat availability. Far fewer studies have attempted to determine the environmental flow requirements of this type of wetland when compared to riverine wetlands, but as environmental managers strive to take an holistic catchment management approach, more habitat modelling studies of coastal and floodplain wetlands are being executed (Rodriguez and Howe 2013; Beesley et al. 2014).

Origins of Habitat Modelling and PHABSIM

The origins of aquatic habitat modelling for determining environmental flows can be traced to the mid-1970s with the development by the US Fish and Wildlife Service of the Instream Flow Incremental Methodology (IFIM) and the set of computer programs required to implement an IFIM study, i.e., the Physical Habitat Simulation (PHABSIM) system (Waters 1976; Bovee 1982). The models require field data

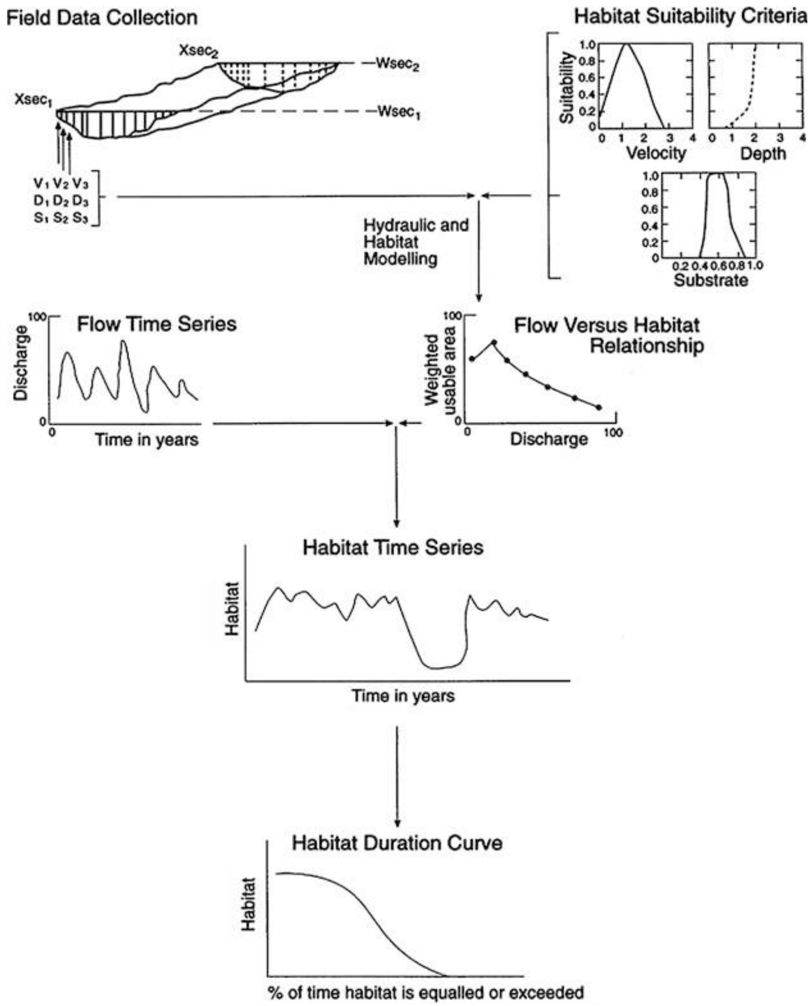

Fig. 1 The basis of PHABSIM (After Nestler et al. 1989) showing the integration of hydraulic measurements and habitat suitability criteria to define the flow versus habitat relationship. This can be combined with a flow time series to produce a habitat time series and habitat duration curve

collection of water depths, velocities, substrate sizes, cross-sectional morphology, and water surface profiles collected at representative or critical cross-sections that sample the range of morphological units present in the reach (such as pools riffles, runs and glides). One-dimensional hydraulic models are used initially to simulate the hydraulics at unmeasured flows (Fig. 1). Knowledge of the univariate relationship between these environmental variables and their associated habitat quality, preference or suitability for the target species and life stages are described with Habitat Suitability Curves (HSCs) (Fig. 2). HSCs can be created by the use of professional judgment or detailed field observations at the site of interest, or a neighboring

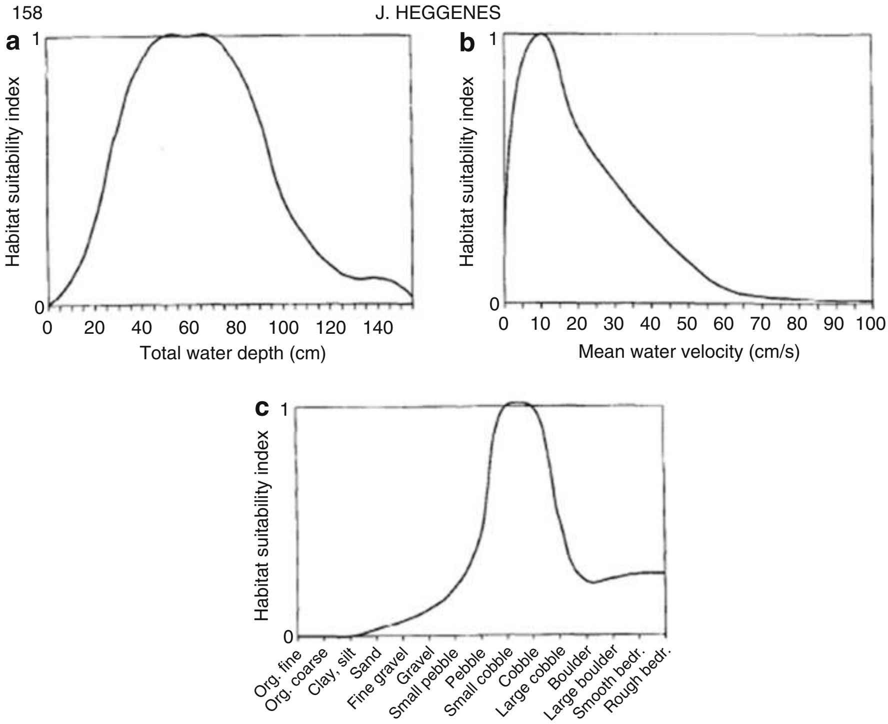

Fig. 2 Example of a set of habitat suitability curves for habitat use by brown trout (*Salmo trutta*) in small streams in summer (Heggenes 1996)

unimpacted site. The habitat modelling stage combines the outputs of the hydraulic models with the HSCs to estimate usable habitat (weighted useable area (WUA)) at a range of simulated flows. Other similar approaches have been developed, such as RHABSIM (Riverine Habitat Simulation) developed in the USA (Thomas R. Payne and Associates 2013) and RHYABSIM (River Hydraulics and Habitat Simulation) created in New Zealand (Jowett Consulting 2011).

PHABSIM has had the most widespread use of all aquatic habitat models worldwide since its introduction. It has also received criticism due to the oversimplification of the flow-habitat relationship, the lack of correlation between WUA and biomass in some studies, the use of HSCs and their treatment of habitat variables as univariate entities, and issues of the transferability of HSCs between studies.

Other Modelling Approaches

Some of the criticisms over the use of HSCs have been addressed by models such as CASiMiR (Computer-Aided Simulation Model for Instream Flow and Riparia) developed at the University of Stuttgart in Germany (Noack et al. 2013) that utilize a mix of expert opinion and fuzzy logic-based rules to describe the habitat use of target species. Fuzzy logic is applied in situations where only imprecise information is available (Mouton et al. 2013). Therefore it is well suited to the creation of HSCs. Fuzzy analysis commonly relates ranges of habitat values to linguistic rules (e.g., a given range of depths, velocities, or substrate sizes are “good”, “medium”, or “poor”) rather than numerical definitions such as those used by traditional suitability indices that vary from 0–1. By using fuzzy habitat models, approaches such as CASiMiR can retain the uncertainty that is inherent in the development and application of HSCs.

More recent developments have seen the application of multidimensional, i.e., two-dimensional (e.g., River 2D) and to a lesser extent three-dimensional (e.g., Delft 3D) hydraulic-habitat models (Dunbar et al. 2012). These enable an enhanced representation of the hydraulic environment compared to more traditional one-dimensional models, made possible through the development of field instrumentation to collect high frequency 2D and 3D hydraulic data (Wilkes et al. 2013), enhanced data storage capabilities to deal with the large volumes of data these instruments produce, and the processing power of PCs that can model with these large and complex data sets. These data also enable the calculation and modelling of turbulent flow properties although a knowledge of how biota respond to and are influenced by these properties and hence how they determine habitat quality and quantity remains an ongoing challenge to incorporate them into environmental flow setting.

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248 Environmental Flows: Building Block Methodology

Jacqueline M. King

Contents

Abstract

The Building Block Methodology was one of the earliest holistic methods globally for assessing environmental flows and remains one of the few that has a detailed users' manual. It was developed in real water-development projects in South Africa, starting in the late 1980s when the government and water scientists became concerned at the condition of the nation's rivers. It is a prescriptive approach that describes the flows required to maintain a chosen ecological condition in the river. It does not provide detailed predictions of ecological change or the social implications of river change and was not designed to address the requirement for larger (>1:2 yr) floods although this is now sometimes included.

Keywords

Building Block Methodology · Environmental Flows · Prescriptive Approach · Ecological condition Category

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Introduction

The Building Block Methodology was one of the earliest holistic methods globally for assessing environmental flows and remains one of the few that has a detailed users' manual (King and Pienaar 2011). It was developed in real water-development projects in South Africa, starting in the late 1980s when the government and water scientists became concerned at the condition of the nation's rivers (Ferrar 1989; King and Tharme 1994). Accepting that decision-makers and developers often cannot wait for long-term research results on water-resource projects, the scientists stepped forward with their best available information and understanding to begin "speaking on behalf of the river" (Tharme and King 1998).

By 1990, it was already understood that the greater the divergence from a natural flow regime, the more a river ecosystem will change, and thus that the concept of a minimum flow for ecosystem maintenance was flawed. Instead, an approach was needed which recognized that rivers can be held at various flow-related levels of ecosystem condition, and that the level could differ from river to river across the landscape. At the time, the government had no structure for identifying what the future condition should be for any one river and so the scientists were asked to recommend a future condition and define the flows that would help attain and maintain this (King and Brown 2010).

Building Block Methodology: The Concept

They did this by developing a method called the Building Block Methodology (BBM) (King and Louw 1998; King et al. 2000). They first assessed the land use, water availability, demographics, conservation areas, and present and future needs for water of a basin, and then recommended a future condition for the river. Using data, global literature, expert opinion, and local knowledge, they then compiled a flow regime consisting of different "blocks" of flow – low flows, small and medium floods – that they thought would support the recommended condition (Fig. 1). The higher (>2 year) floods were not addressed because at that time it was assumed that the dams would not greatly alter these. The low flows defined the fundamental future nature of the river, reflecting its location in a summer or winter rainfall area and its degree of perenniality of flow. The higher flows addressed specific ecosystem dependencies, such as migration and spawning flows for fish. The results were provided as recommended monthly volumes of low flows, and the duration, timing, and magnitude of floods, for both maintenance and drought years.

Some basic concepts captured in the early development of the BBM reflect the emerging thinking on holistic flow assessment methods.

- It recognized the need for an holistic approach that could address the condition of the whole river ecosystem, including its riparian zones, floodplains, and estuary.
- It recognized the need for a multidisciplinary approach, with the core disciplines involved being hydrology, hydraulics, sedimentology, fluvial geomorphology,

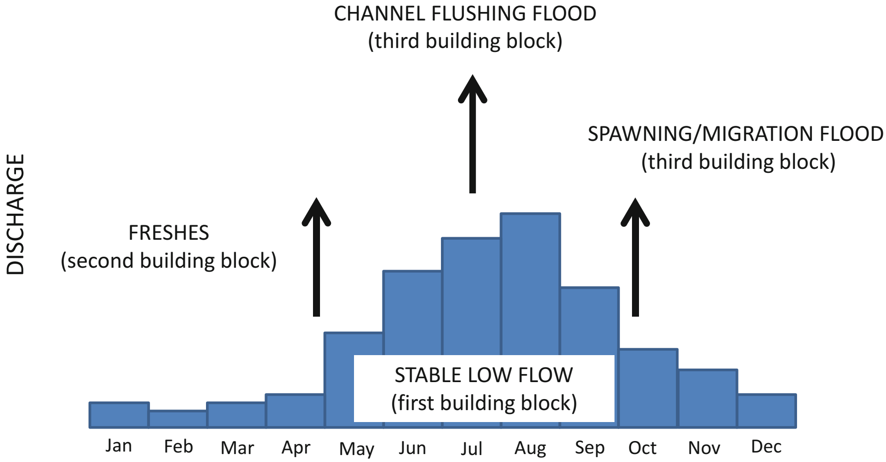

Fig. 1 The building blocks of an environmental flow requirement as described by the Building Block Methodology (Figure reconstructed from King and Tharme (1994); used with permission from Water Research Commission, South Africa)

water chemistry, ichthyology, riparian and aquatic botany, zoology (invertebrates), and socioeconomics. Other biophysical and socioeconomic disciplines were included as required.

- It recognized that the whole flow regime needed to be managed, not just the low flows, and divided the regime into rudimentary ecologically relevant flow classes.

Implications for Water Legislation in South Africa

Although modest compared with more recent holistic approaches, the application of, and results from, the BBM persuaded the legal team writing South Africa's new, postapartheid water law that water for ecosystem maintenance could be quantified and thus enforced. In what was seen as a major recognition of the importance of healthy ecosystems for people, a reserve of water for ecosystem maintenance was written into South Africa's 1998 National Water Act (NWA) as one of only two water rights – the other being for basic human needs. For this and other reasons, the NWA was hailed as one of the most advanced in the world, and in recognition the South African Minister of Water Affairs received the Stockholm Water Prize in 2000 (www.siwi.org).

Future Challenges

Reflecting its origins, the BBM is a prescriptive approach – it focuses on a specific objective and describes the single flow regime to achieve this. This has emerged as a flaw in many cases for two main reasons. First, stakeholders questioned the right of scientists to define a future condition for a river, seeing this as a subject for negotiation in which they should be included. Second, it is poorly suited to negotiation because most effort is directed toward justifying the single recommended flow regime; queries on the consequences of modifying the BBM's recommended flows could not easily be answered. Although the prescriptive approach does have many useful applications, it became clear that in basin planning where there is a wish to explore options and facilitate discussion and negotiation a more suitable approach is through interactive, scenario-based methods. The BBM has thus evolved into the interactive method DRIFT, described elsewhere in this document.

Although dated, the BBM still maintains a presence in flow assessments and has been or is being widely applied internationally, from Canada to China and the Congo to Iran. Its simple approach and user manual offer a useful introduction to the concepts and methods of environmental flow assessments and appear to be popular with many developing countries newly working in this field.

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Environmental Flows: Downstream Response to Imposed Flow Transformations (DRIFT)

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Jacqueline M. King

Contents

Abstract

DRIFT (an acronym for Downstream Response to Imposed Flow Transformations) is an interactive, holistic scenario-based environmental flow assessment method. It consists of procedures and software developed in South Africa to predict the expected ecological and socioeconomic implications of proposed water-resource management activities. Its outputs complement the engineering and economic information usually available on such activities and so DRIFT helps provide a more balanced picture of the full suite of costs and benefits linked to water resource proposals. Using scenario analysis, DRIFT can address the different options for location, design, and operation of a specific dam; rehabilitation of a river; or any other management activities that could affect the flow or inundation patterns of inland waters. DRIFT is increasingly applied in basin-wide strategic planning exercises. Training courses are available.

Keywords

DRIFT · Environmental Flows · Holistic Basin-wide strategic planning · Project Planning

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Introduction

DRIFT (an acronym for Downstream Response to Imposed Flow Transformations) is an interactive, scenario-based environmental flow assessment method. It consists of procedures and software developed in South Africa to predict the expected ecological and socioeconomic implications of proposed water-resource management activities (Brown and Joubert 2003; King et al. 2003; Brown et al. 2013). Its outputs complement the engineering and economic information usually available on such activities and so DRIFT helps provide a more balanced picture of the full suite of costs and benefits linked to water-resource proposals. Using scenario analysis, DRIFT can address, for instance, the different options for location, design, and operation of a specific dam; rehabilitation of a river; or any other management activities that could affect the flow or inundation patterns of inland waters. DRIFT is also increasingly applied in basin-wide strategic planning exercises.

Employing a multidisciplinary team, DRIFT can be used to predict for any considered scenario the potential changes in, for instance, the flow regime; channel configuration; hydraulic habitat; bank erosion; water chemistry; riverine and floodplain vegetation; river gardens; river, estuarine, and near-coastal marine fisheries; rare species; pest species; human and livestock river-related health; availability of baptism areas; household incomes; GDP; job creation; hydropower production; and more, as well as offering possible options for biodiversity offsets and other mitigation (King and Brown 2010; King et al. 2014). Detailed information such as this on the potential costs of water-resource developments has only become available to decision-makers with the advent of holistic assessment methods in the last two decades.

Structure of the DRIFT System

The DRIFT Decision Support System (DSS) has three main modules: Set Up, Knowledge Capture, and Analysis (Fig. 1). In the Set Up module, the specialist team is established, and a basin-delineation exercise is completed. This defines the boundaries of the work and summarizes the nature of the basin, its river system, and its people, leading to the selection of modelling/study sites, scenarios, and indicators (attributes of the river and its social structures that could change with flow change).

The Knowledge Capture module houses the hydrological data, and the relationships between flow, ecosystem, and social indicators. The hydrological simulations, which address the baseline (usually present day) situation and all chosen scenarios, consist of simulated daily (or sub-daily for peaking hydropower dams) flow sequences for 30 or more years for each study site. The flow sequences are summarized in the DSS in terms of ecologically relevant flow indicators, such as Onset of the Dry Season, or Wet Season Duration. External models, if available, also supply inputs on sediments and water quality. Relationships between the flow, ecological,

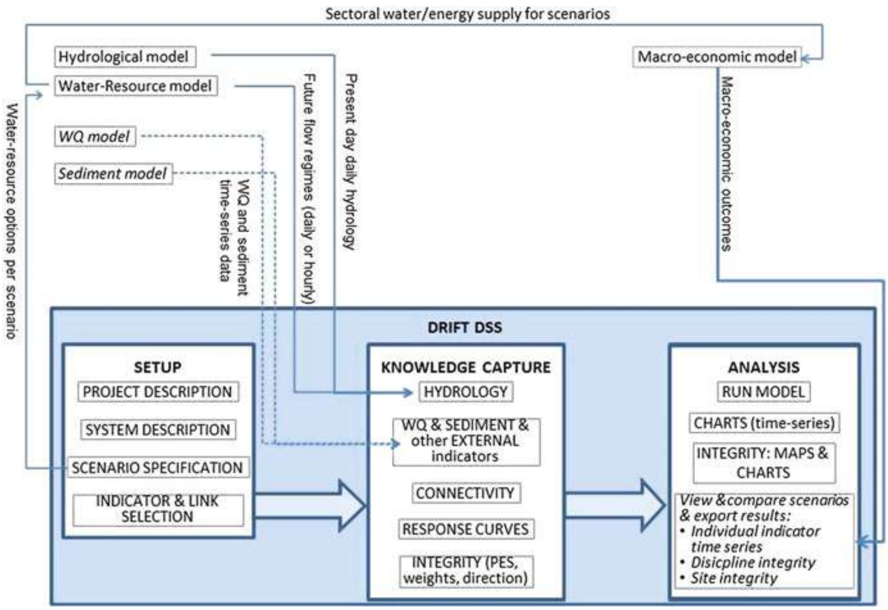

Fig. 1 Arrangement of modules in the DRIFT DSS and inputs required from external models (Figure from Brown et al. (2013); used with permission from Water Research Commission, South Africa)

and social indicators, derived from the literature, river-specific data, expert opinion, and local knowledge are described by the team and entered in the DSS as response curves. Each response curve describes the relationship between a driving and a responding indicator, for instance between the abundance of fish guild A and household incomes. Up to 70 ecological indicators (e.g., sand bars, upper wetbank trees, middle floodplain herbivores) and 10 social indicators (e.g., household income from reeds, baptism sites, potable water) have been used in river basin assessments.

The Analysis module uses the DSS's knowledge base (the hydrological summaries and the response curves) to provide predictions of change in the indicators. These are shown as time series for each scenario/site, over the same period as the hydrological simulation. Other outputs include a summary of all the individual ecological indicators' responses as an overall integrity rating of change in ecosystem condition per scenario/site, a similar summary of the social predictions, and a range of other report-friendly options.

Innovative features of the DRIFT approach are: (1) the automated flow analyses, which summarize vast amounts of hydrological data in a form that ecologists and socioeconomists can understand and use; (2) its use of time-series for all hydrological, ecological, and social analyses, which allow long-term trends to be depicted; and (3) the quantitative or semiquantitative predictions of change that provide the specifics for monitoring programmes (see also http://www.Southernwaters.co.za).

Applications of DRIFT

The process of compiling the response curves requires specialists from many disciplines to work together and may push them to the limits of their understanding of the ecological and social systems involved. This triggers new focused research, teaching, and postgraduate studies, which is beneficial for a country in terms of skills development, and for water managers in that they have contemporary research outputs for use in their projects. For instance, between 1990 and 2010, South Africa produced more than 500 scientific articles or book chapters and 54 postgraduate theses on the subject of environmental flows or linked topics, made 350 presentations at national and international conferences, and ran 22 national training courses.

DRIFT has been applied widely inside and outside of its country of origin, including in many transboundary investigations in Africa, South America, and Asia, and has been recognized by various international institutions and organizations, including The Permanent Court of Arbitration in The Hague, IUCN, the Asian Development Bank, WWF, and the World Bank.

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Environmental Flows: Ecological Limits of Hydrologic Alteration (ELOHA)

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Angela H. Arthington

Contents

Abstract

The ELOHA framework addresses the demand for transferable hydroecological relationships and environmental flow guidelines for many rivers across large spatial scales. The framework is sufficiently flexible to inform decisions at the planning stage of new developments by predicting likely ecological responses to proposed changes in flow regime, or can be used to guide flow restoration and conservation programmes at regional (multiple river) scales in various governance contexts.

Keywords

Environmental flows · Hydrological classification · Hydroecological relationships · Effects of flow regime change

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Introduction

The ELOHA framework aims to set limits on levels of ecological change in rivers as a consequence of hydrologic alteration. ELOHA differs from most environmental flow frameworks by addressing the demand for transferable hydroecological relationships and environmental flow guidelines rather than managing for the "uniqueness" of each river's flow regime (Arthington et al. 2006). The framework involves hydrologic modelling of entire river networks, classification of river segments into ecologically relevant types based on flow metrics, assessment of flow regime alterations, and testing ecological responses to flow alterations for each river type (Fig. 1). The idea is that robust flow alteration-ecological response relationships developed for each hydrologic class of river can be transferred to less well-studied rivers of the same class and guide environmental flow decisions for each distinctive river type (Poff et al. 2010). The framework is sufficiently flexible to inform decisions at the planning stage of new developments by predicting likely ecological responses to proposed change in flow regime, or be used to guide flow restoration and conservation programmes at regional (multiple river) scale in various governance contexts.

The ELOHA Procedure

A guiding principle of ELOHA is that ecological responses to particular features of the altered flow regime can be interpreted most robustly, and usefully, when there is some mechanistic or process-based relationship between the ecological response and the particular flow regime component (Poff et al. 2010). Flow alteration-ecological response relationships can be compiled from existing data, or new data collected along a flow regulation gradient, and tested statistically to determine the form (e.g. threshold, linear) and degree of ecological change (positive or negative) associated with a particular type of flow regime alteration (Arthington et al. 2006).

In the final stages of an ELOHA application, scientists, stakeholders, and managers give consideration to a suite of flow alteration–ecological response relationships to specific types of flow alteration. Where there are clear threshold responses (e.g., overbank flows needed to support riparian vegetation or provide fish access to backwater and floodplain habitat), a "low risk" environmental flow would be one that does not cross the threshold of hydrological alteration for overbank flows. For a linear response where there is no clear threshold for demarcating low from high risk, a consensus stakeholder process may be needed to determine acceptable risk to a valued ecological asset, such as an estuarine fishery dependent upon freshwater inflows (e.g. Loneragan and Bunn 1999).

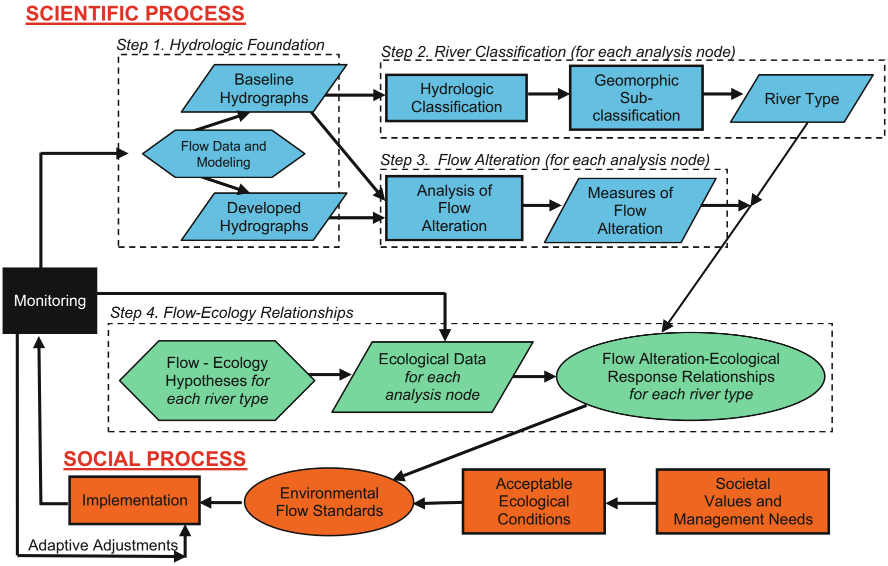

Fig. 1 The ELOHA (Ecological Limits of Hydrologic Alternation) framework for ecological change in rivers as a consequence of hydrologic alteration (Source: Poff et al. 2010)

Applications

Projects applying the ELOHA framework are diversifying, especially in the USA (Kendy et al. 2012), Spain (Belmar et al. 2011), China (Zhang et al. 2012), and Australia (Arthington et al. 2012). Most effort to date has involved development of hydrological classifications and methods to compare baseline and altered flow regimes (e.g. Kennard et al. 2010; Olden et al. 2012; Mackay et al. 2014). A test of the ELOHA framework in Australia identified a suite of hydrologic variables (frequency of low flows and floods, duration of zero and low flow spells, low flow variability, and the predictability of seasonal flow patterns) that together govern the structure (species richness, composition) of riparian vegetation, aquatic plant and fish assemblages, and measures of species abundance (Arthington et al. 2012). The largest impacts to fish populations and community diversity occurred where streams that naturally stop flowing have received managed water releases from dams and now rarely cease to flow (Rolls and Arthington 2014).

Applications of ELOHA in flow regime restoration have demonstrated that increases in flow magnitude reduced riparian encroachment in the Cheoah River (Upper Tennessee River Basin) 4 years after flow restoration, as predicted from relationships between flow alteration and riparian responses (McManamay et al. 2013). Fish richness did not increase as predicted, probably because of interactions among flow, temperature, substrate characteristics, barriers, and the effects of a peak flood event. Multivariate models were useful in unravelling these confounding effects on Cheoah River fish communities, as observed elsewhere (*cf* Arthington et al. 2012, 2014).

Future Development of ELOHA

The ELOHA framework is envisaged as proceeding in an adaptive management context involving monitoring and fine-tuning the ecological response relationships as new knowledge of responses becomes available (Fig. 1). Implementation of ELOHA and other environmental flow restoration frameworks as ecosystem-scale experiments (Poff et al. 2003; King et al. 2010) will undoubtedly increase understanding of multiple pathways, ecological feedback loops, and their management implications in regulated and restored rivers.

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Environmental Flows: The Savannah Process

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Andrew T. Warner

Contents

Abstract

Because The Nature Conservancy's approach to site-specific environmental flow assessment and implementation was first described for the Savannah River in Georgia, USA, it has acquired the nickname "the Savannah Process." Like the DRIFT method and the Building Block Methodology (BBM), the Savannah Process addresses the linkages between diverse flow characteristics and ecosystem components. This holistic method relies on facilitated expert consensus to prescribe environmental flows. The process consists of five steps. Step 1 is a one-day orientation meeting to inform and engage interested scientists, water managers, government agencies, and other stakeholders and provide a forum to express their values and concerns for the river. Step 2 is the preparation of a literature review and summary report describing existing data and knowledge of the river-floodplain-estuary system, species, and their flow dependencies to describe the annual and inter-annual flow or inundation patterns needed to support ecosystem health. Step 3 is a facilitated expert workshop, typically about two days, with participants representing expertise in all riverine ecosystem components. During this step, scientists are tasked with developing a set of environmental flow components (EFCs), which can be discussed by workshop

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participants in breakout groups. The whole group then reconvenes for a final review and agreement upon a unified environmental flow prescription. Step 4 is the initial implementation of the flow prescription. Following the flow workshop, scientists continue a dialogue with water managers to identify opportunities for implementing portions of the recommendations. Step 5 consists of additional data collection and research as needed to refine the environmental flow prescription. To date, the Savannah Process has been applied in a range of contexts around the world, mostly to guide changes in existing reservoir operations.

Keywords
Environmental flows · Savannah Process · Water management · Water allocation · Environmental water need · The Nature Conservancy · Environmental flow components

Introduction

The Nature Conservancy's adaptive, interdisciplinary, and science-based approach to site-specific environmental flow assessment, implementation, and adaptive management has been applied in a range of contexts around the world. Because it was first demonstrated and described for the Savannah River in Georgia, USA (Richter et al. 2006), it has acquired the nickname "the Savannah Process." Similar to the Downstream Response to Imposed Flow Transformation (DRIFT) method (King et al. 2003) and the Building Block Methodology (BBM) (King and Louw 1998), the Savannah Process is considered holistic because it addresses the linkages between a full range of flow characteristics and diverse ecosystem components. All three of these holistic methods rely heavily on facilitated expert consensus to prescribe environmental flows. To date, the Savannah Process has been most applied to helping guide changes in existing reservoir operations, with each of these case studies including consideration of floodplain and/or coastal (estuarine) wetland systems. Figure 1 illustrates the Savannah Process.

The Savannah Process

Step 1 is a one-day orientation meeting. The purpose of the orientation meeting is to inform and engage interested parties – including scientists, water managers, government agencies, and other stakeholders – in the process of prescribing environmental flows and provide a forum for them to express their values and concerns for the river. The meeting begins with an overview of the proposed process. During breakout sessions, participants discuss the details of the process, identify additional scientists who should be involved, and identify sources of information that can inform the process.

Step 2 is the preparation of a literature review and summary report describing existing data and knowledge of the river-floodplain-estuary system, native species, and their flow dependencies. The primary purpose is to quantitatively

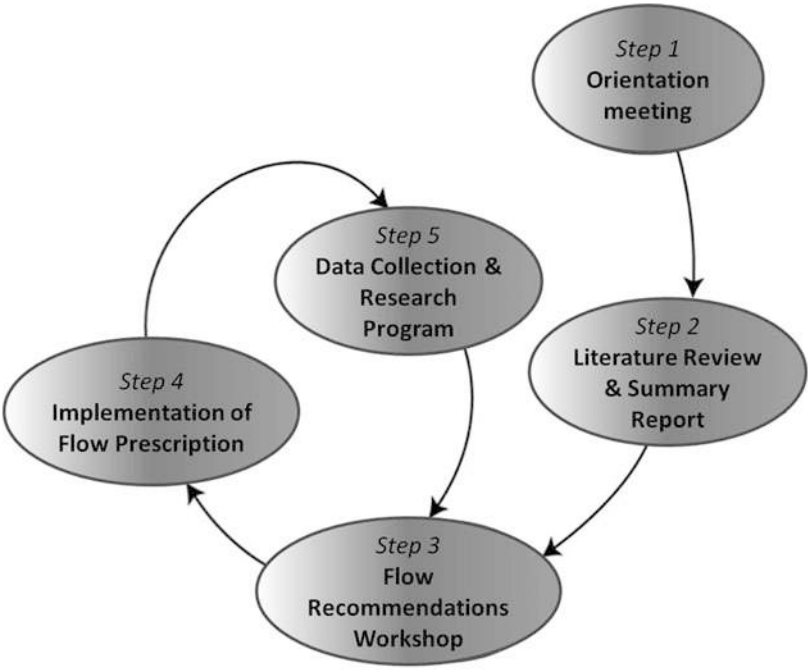

Fig. 1 Adaptive Management of Environmental Flow Restoration in the Savannah River, Georgia, USA, and subsequently applied to most of the rivers involved in the Sustainable Rivers Project (Based on Richter et al. 2006; used with permission from John Wiley and Sons)

describe the annual and interannual flow and inundation patterns needed to restore or sustain ecosystem health, as well as to capture additional qualitative flow-ecology relationships. During this step, specific habitat requirements for a diversity of species life stages are articulated, along with their links to specific conditions of flow or inundation. The Indicators of Hydrologic Alteration (IHA) software may be used to analyze unaltered versus altered river hydrology or wetland inundation; for example, the current hydrology of the river compared to the predevelopment hydrology. Typically, this report is contracted to an interdisciplinary team, with members representing a diversity of technical disciplines. Richter et al. (2006) describe the basic structure of the report in detail, noting that it is helpful to organize information about life stages and ecological functions - such as specific timing and frequency - according to specific environmental flow components. Box 1 lists a number of example outcomes of Step 2.

Box 1. Example outcomes of Step 2 of the Savannah Process

- *Supporting the Development of Flow Recommendations for the Stretch of Big Cypress Creek below Lake O' the Pines Dam* is a literature review and summary report prepared by a team from Texas A&M University in support of an environmental flows workshop (Step 3) for Big Cypress Bayou and

(continued)

Caddo Lake, one of the Sustainable Rivers Project (SRP) demonstration sites and a RAMSAR site.

- Summary Report Supporting the Development of Ecosystem Flow Recommendations for the Savannah River below Thurmond Dam is a literature review and summary report prepared by a team from the University of Georgia to inform an environmental flows workshop (Step 3) for the Savannah River, one of the SRP sites.
- Preliminary IHA Analysis for the Middle Fork Willamette River at Jasper OR describes an analysis of hydrologic alteration using the Indicators of Hydrologic Alteration (IHA) software (link to IHA software) to inform an environmental flow workshop (Step 3) for the Willamette River in Oregon, USA, one of the SRP sites.
- Summary Report to Assist Development of Ecosystem Flow Recommendations for the Coast Fork and Middle Fork of the Willamette River, Oregon, is a literature review and summary report prepared by a team from Oregon State to inform an environmental flows workshop (Step 3) for the Coast and Middle Forks of the Willamette River, Oregon, USA, one of the SRP sites.
- Indicators of Hydrologic Alteration Analysis for the Patuca River describes an analysis of hydrologic alteration using the Indicators of Hydrologic Alteration (IHA) software to inform an environmental flows workshop (Step 3) for the Patuca River in Honduras.
- Ecological and Social Impressions of the Middle Patuca River and Potential Consequences of the Patuca 3 Hydropower Project is a literature review and summary report for the Patuca River in Honduras, for which very few data – but considerable local knowledge – were available.

Source: compiled from information available at https://www.conservationgateway.org

Step 3 is a facilitated expert workshop, which typically runs about two days. The workshop participants should be highly interdisciplinary, representing expertise in all riverine ecosystem components. The literature review and summary report is provided to all participants, typically 3–4 weeks prior to the workshop to allow time for review. During this workshop, scientists are tasked with developing a set of flow recommendations, also known as an environmental flow prescription. Scientists are encouraged to articulate these recommendations quantitatively, describing recommended ranges of flows throughout the year in terms of magnitude, duration, frequency, timing, and rate of change. The flow recommendations can be provided in the form of Environmental Flow Components (EFCs), such as low flows, high-flow pulses and floods, and recommendations can vary between dry, average, and wet years. Initial recommendations usually are developed within breakout groups, each focusing on a major portion of the river (e.g., confined river vs. floodplain river) or major groups of organisms (e.g., fish, riparian vegetation, estuarine wetlands). Each breakout group prepares flow recommendations with ecological justification for its area of emphasis. Next, workshop participants are reorganized into new breakout groups focused on different EFCs (e.g., low flows, high-flow pulses, and floods), during which differences between the previous breakout group recommendations are resolved. The whole group then reconvenes for a final review and agreement upon a unified environmental flow prescription. Significant knowledge gaps are captured throughout the breakout group discussions and are used in a session toward the end of the workshop to discuss and prioritize future research needs. Box 2 presents some example outcomes of Step 3.

Box 2. Flow workshop reports illustrating example outcomes of Step 3, of the Savannah Process

- Defining Ecosystem Flow Requirements for the Bill Williams River, Arizona, summarizes a flow workshop and results for the Bill Williams River, a highly regulated river in an arid climate, as synthesized into the preworkshop literature review and summary report (Step 2 product).
- *Environmental Flows Workshop for the Middle Fork and Coast Fork of the Willamette River, Oregon* summarizes a flow workshop which developed environmental flow prescriptions for a regulated river to inform dam reoperation. This is synthesized into the preworkshop literature review and summary report (Step 2 product) and includes figures generated by Regime Prescription Tool (HEC-RPT) modeling software developed by the US Army Corps of Engineers Hydraulic Engineering Center (HEC; http://www.hec.usace.army.mil/).
- Environmental Flow Assessment for the Patuca River, Honduras: Maintaining ecological health below the proposed Patuca III Hydroelectric Project summarizes recommendations developed during flow workshops for the Patuca River, a pristine river in an extremely data-poor context prior to dam construction.

Source: compiled from information available at https://www.conservationgateway.org

Step 4 is the initial implementation of the flow prescription. Following the flow workshop, scientists continue a dialogue with water managers to identify opportunities for implementing portions of the recommendations. Often, sufficient management flexibility exists to begin implementing some of the recommendations immediately, which can be framed as flow experiments. These provide a valuable opportunity to test the flow-ecology relationships articulated in the environmental flow prescription and to improve scientific understanding of the flow conditions necessary to effect desired ecological changes or processes. Therefore, monitoring of both flow changes and ecological response are critical at this stage (e.g., Higgins et al. 2011). By carefully tracking the response of an ecosystem to flow management, the flow prescriptions can be further refined, thus helping to ensure that river management accomplishes its objectives. While some recommendations can be implemented or tested relatively quickly, other recommendations may require further modeling or research to reduce physical, economic, or political constraints or uncertainties (Warner et al. 2014). Box 3 presents an example of Step 4 outcomes.

Box 3. Report document with example outcomes of Step 4 in the Savannah Process

- Environmental Assessment and Findings of No Significant Impact: Modification of Regulation and Operation of Green River Lake, Kentucky, is the Environmental Assessment and Finding of No Significant Impact for the re-regulation of Green River Reservoir, Kentucky, USA. The Green River was the first Sustainable Rivers Project site. On the Green River, the Army Corps of Engineers began to reoperate the dam to provide environmental flows. This reoperation was eventually formalized in a revision of the dam's Water Control Plan. This document is part of the environmental review required to make such a revision.

Source: based on information available at https://www.conservationgateway.org

Step 5 is additional data collection and research as needed to refine the environmental flow prescription. Konrad (2010) examined monitoring data collected for this purpose at five different sites where the Savannah process is being applied.

Additional information and resources are available at: https://www.conservationgateway.org/ConservationPractices/Freshwater/EnvironmentalFlows/Pages/environmental-flows.aspx

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Environmental Flows: Three-Level Approach for Developing and Implementing Environmental Flow Recommendations

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Jeffrey J. Opperman

Contents

Abstract

The three-level approach for developing environmental flow methods spans relatively simple desktop estimates of flow needs to a highly sophisticated program of research and modeling to refine environmental flow targets. Each level builds information, capacity, and support for subsequent levels of sophistication as deemed necessary. In this way, proactive, practical implementation can begin immediately upon completion of the level of assessment most suitable to the resources available to a particular water body, jurisdiction, and associated circumstances.

Keywords

Environmental flows · Flow regime · Rivers · Aquatic ecology · Floodplain · River management · Hydrology

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Introduction

The three-level approachfor developing environmental flow methods spans relatively simple desktop estimates of flow needs to a highly sophisticated program of research and modeling to refine environmental flow targets. Each level builds information, capacity, and support for subsequent levels of sophistication as deemed necessary. In this way, proactive, practical implementation can begin immediately upon completion of the level of assessment most suitable to the resources available to a particular water body, jurisdiction, and associated circumstances.

The key characteristics that define this framework include:

- Funds for research and modeling to support flow assessment and implementation are invested strategically to address the most important issues and reduce the most vexing uncertainties; methods are matched to the level of certainty required and the level of funding available.
- The framework is iterative such that higher levels are deployed to the extent they are necessary and information generated at one level can provide the foundation for, and identify the need for, higher levels.
- Processes for flow assessment and flow implementation are intertwined; many of the key characteristics of the assessment process are designed to lay the foundation for flow implementation.

Three Levels for Developing Environmental Flows

The three levels for developing environmental flows are described below.

Level 1

A Level 1 approach is based on hydrological "desktop" methods, which generally rely upon hydrological data (e.g., daily flow rates). This approach is appropriate for regional planning and preliminary standard setting or for organizing and preanalyzing information for a Level 2 approach. Hydrologic desktop methods are commonly used due to their primary advantage of being relatively quick and inexpensive. However, their primary drawback is that they generally provide simplistic flow levels that do not fully encompass current understanding of river functions and processes. We suggest that hydrologic desktop methods are most appropriate when combined with a review of available information for a given river system *and augmented by basic understanding of river functions*. Under these circumstances, they can produce credible and comprehensive initial flow recommendations.

A comprehensive hydrologic desktop approach synthesizes two primary sources of information: (1) a hydrological analysis tool that is capable of assessing a range of flow levels; and (2) a literature review of the linkages between the flow regime and key riverine resources. This review should incorporate all the available relevant information for the specific river system and be augmented by broader literature on riverine processes. In the absence of specific information on the focal river, practitioners can draw on broader literature with an emphasis on information relevant to similar river types (e.g., in terms of geomorphology, drainage area, valley characteristics) and ecosystems. This integration of a literature review to the hydrological analysis is what advances the comprehensive hydrologic desktop approach beyond simple "rules of thumb."

The Nature Conservancy developed the Indicators of Hydrologic Alteration (IHA) software to support a comprehensive desktop approach (see Box 1). For regional-scale assessments, a team of leading river scientists representing ten international scientific organizations developed the Ecological Limits of Hydrologic Alteration (ELOHA), a flexible, scientific framework for assessing and managing environmental flows across large regions, when limited time and resources preclude evaluating individual rivers (Poff et al. 2010). ELOHA can act as a Level 1 framework that combines desktop hydrologic analysis of river and stream systems across a region with a review of existing ecological databases and literature.

Box 1 Indicators of Hydrologic Alteration (IHA)

Indicators of Hydrologic Alteration (IHA) is free statistical software that provides useful information for those trying to understand natural and/or altered hydrologic systems, such as river flows or lake or wetland water levels (Richter et al. 1996, 1998). For altered or potentially altered systems, the software supports efforts to develop environmental flow recommendations. The software assesses 67 ecologically relevant statistics derived from daily hydrologic data (e.g., flow or water levels). For instance, IHA can calculate the timing and maximum flow of each year's largest flood or lowest flows, then calculates the mean and variance of these values over some period of time. Comparative analysis can then help statistically describe how these patterns have changed for a particular river or lake, due to abrupt impacts such as dam construction or more gradual trends associated with land- and water-use changes. In addition to statistics on annual and monthly high, low, and average flows, IHA derives statistics for five major components of flow that are ecologically important: extreme low flows, low flows, high flow pulses, small floods, and large floods. These five components of flow regimes are known as "environmental flow components."

IHA supports the understanding of natural hydrologic patterns and/or how those patterns have been altered but it does not generate the actual recommendations for appropriate flows for a given system (Mathews and Richter 2007). Determining environmental flow needs is inherently an interdisciplinary

(continued)

process, and ecologists, water managers, and other stakeholders can use the information generated by IHA to decide on an acceptable amount of flow alteration for different times of the year and for different years. For more information and a free IHA download can be found at: http://www.conservationgateway.org/ConservationPractices/Freshwater/EnvironmentalFlows/MethodsandTools/IndicatorsofHydrologicAlteration/Pages/indicators-hydrologic-alt.aspx

Level 2

A Level 2 approach involves engaging an expert panel and building upon the basic approach from Level 1: initial flow recommendations are developed through an integration of hydrologic analysis and information on the linkages between flow regime and river processes. The initial flow recommendations do not require new data collection or modeling but, instead, rely primarily on expert judgment supported by an assessment of hydrology and a thorough literature review. The primary distinction between this level and Level 1 is that this synthesis of information and articulation of expert judgment into flow recommendations occurs within the framework of a flow workshop with diverse – and typically a larger number of – participants.

The Nature Conservancy developed an example of a Level 2 approach, referred to as the "Savannah Process" because it was first implemented on the Savannah River, USA (Richter et al. 2006). This process develops scientifically credible environmental flow prescriptions in a relatively short time through facilitated expert workshops. The Savannah Process has been applied in many rivers in the United States, through the Sustainable Rivers Project (e.g., Savannah River [Georgia/South Carolina], Big Cypress-Caddo Lake [Texas/Louisiana], Bill Williams River (Arizona), Willamette River [Oregon]), and has also been applied to the Yangtze River in China and the Patuca River in Honduras supported by traditional ecological knowledge (Esselman and Opperman 2010).

At the site scale, a Level 2 approach produces a set of flow recommendations that can either lead to experimental flow releases or support management decisions (e.g., permitting or changes in dam operations). Monitoring of such flows can provide a learning opportunity to improve understanding of river processes and support adaptive management.

At the regional scale, scientists and water managers in the Connecticut (USA), Susquehanna (USA), Upper Ohio (USA), Magdalena (Colombia), and Potomac (USA) river basins have elevated ELOHA to Level 2 by using expert panels to assess flow needs for river types, rather than for individual rivers, throughout their entire river basins.

Level 3

A Level 3 process is appropriate for situations that require a high degree of certainty to justify management decisions. Such situations may include those where water is over-allocated and heavily contested, endangered species are present, defined policies dictate processes, or binding long-term decisions are being made (e.g., the latter two are characteristic of the 30–50 year licensing of hydropower dams by the US Federal Energy Regulatory Commission). In these situations, decision makers will require a higher threshold of rigorous analysis before initiating an environmental flow program. Thus, a Level 3 process is characterized by greater upfront investment in more sophisticated methods for examining tradeoffs and predicting results of management decisions.

However, we recommend that a Level 3 process retain many of the features of a Level 2 process that are intended to develop collaborative relationships — facilitating subsequent implementation — and those features that strategically target research funds at the most important issues and most problematic uncertainties. For example, the suite of technical studies, models, and analyses within a Level 3 approach can be selected through interdisciplinary workshops. Informed by expert input in the workshops, the research and modeling program of a Level 3 approach can incorporate a broad range of specific methods, including both typical environmental flow assessment techniques as well as diverse approaches for studying riverine processes (e.g., water quality or ecological models, fish passage studies).

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Environmental Flows: Wetland Water Levels

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Michael C. Acreman and J. Owen Mountford

Contents

Abstract

The term environmental flows emphasises the importance of water for the natural environment. For wetland ecosystems, a critical factor is the depth of water on the surface or the level of the water table in the ground. In the UK, scientists have produced quantitative guidelines on the water level requirements of wetland plant communities (according to the UK National Vegetation Classification). For each class, information is provided on a) water supply mechanisms; b) water-regime; c) nutrient regime; d) management regime; e) vulnerability; f) the restorability of the type; and g) gaps in the knowledge. In terms of water-regime parameters, the guidelines indicate the water-depths (maximum, minimum, duration of exposure *etc*) that are favourable, the depth-ranges that the class can tolerate for short periods, and those that are damaging.

Keywords

Wetlands · Environmental flows · Water table level · Tolerance

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Introduction

The definition of the term environmental flows clearly emphasises the importance of water moving through the natural environment. The focus of environmental flow studies has thus been on rivers, where flow (discharge m^3s^{-1}) is a dominant process moving sediments and nutrients and providing habitat for aquatic organisms. Yet even in rivers it is recognised that flow itself is often not the ultimate determinant of a river's ecological condition. Moreover, hydraulic variables, such as river water depth that result from the interaction of flow with channel geometry and plant biomass, control the physical habitat available for animal and plant communities. This concept is particularly true of many wetland types, including bogs and mires. Whilst water does flow through them, such as over the surface of a floodplain, it is primarily the resulting depth of water on the surface or the level of the water table in the ground that influences the wetland ecosystem. Thus water elevation (relative to the surface) is ecologically the most important aspect of the water regime for wetlands.

Guidelines for Wetland Water Levels

Building upon a considerable body of theory and practice in wetland eco-hydrology, there have been attempts by a consortium of UK academic organisations (Open University, University of Sheffield and the Centre for Ecology & Hydrology) to provide objective and quantitative guidelines on water level requirements of wetland plant communities (Mountford et al. 2004; Wheeler et al. 2004). Development of the guidance has been stimulated by agri-environment schemes and nature protection designations aimed at maintaining and restoring wetlands, notably in floodplain grasslands (Gowing et al. 2002) and grazing marsh (Mountford et al. 1997). These studies began with investigations of altered nutrient regimes and then proceeded to experimentally manipulated water-levels, integrating the hydrological and nutrient requirements to produce the guidelines. Wetlands were defined according to the UK National Vegetation Classification (NVC; Rodwell 1991–2000) and for each class eco-hydrological guidelines provide information on (a) water supply mechanisms; (b) water-regime; (c) nutrient regime; (d) management regime; (e) vulnerability; (f) the restorability of the type; and (g) gaps in the knowledge. In terms of water-regime parameters, the guidelines indicate the water-depths (maximum, minimum, duration of exposure *etc*) that are favourable for the habitat, the depth-ranges that the habitat can tolerate for short periods, and those that are damaging for the habitat.

Figure 1 shows a "traffic light-based" water level regime zones diagram that depicts the mean water table requirements for each month of the year for a grassland habitat (Midelney soil, coded MG13). The green area shows preferred conditions. The amber area denotes conditions that are not ideal, but which the plants can withstand for short periods. The red area marks conditions which the vegetation cannot tolerate. The guidelines have been used for many additional studies, such as part of a method for assessing the suitability of floodplains for species-rich meadow restoration (Duranel et al. 2007). The method was tested on the floodplain of the

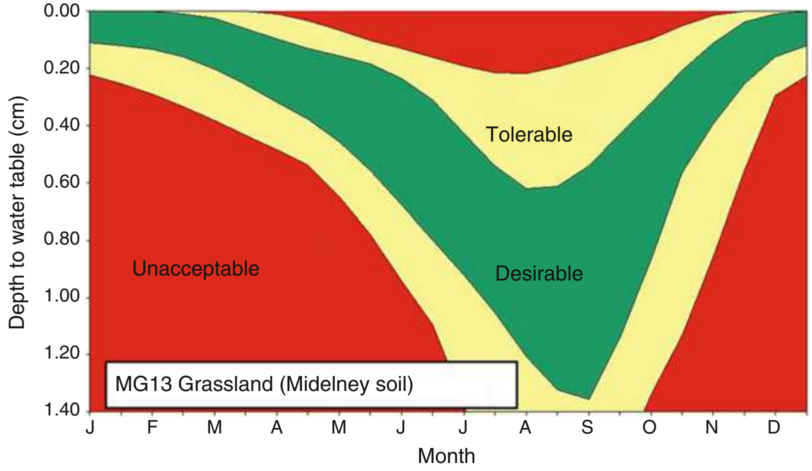

Fig. 1 Water regime requirements of MG13 wetland plant community (Wheeler et al. 2004; used with permission under Open Government License v3.0, UK)

River Thames in central southern England, where it was found that both the maximum duration of flood events in autumn and winter and the depth of the groundwater table during the summer exceeded the requirements of the target species. The likely impacts of climate change on UK wetlands were assessed by using the guidelines to determine whether future water level regimes (projected by hydrological models) would be desirable, tolerable or unacceptable for their plant communities (Acreman et al. 2009). The conceptual framework of these water level regime diagrams has been adopted to define water requirements of other features of wetlands, such as wintering and breeding birds and archaeological remains (Acreman et al. 2013).

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254 Environmental Flows: Environmental Watering

Nick Bond

Contents

Abstract

Environmental watering is a term that has gained traction, primarily in Australia, to describe the delivery of environmental flows to rivers and wetlands. The term was defined in Australia's federal Water Act (2007) as "the delivery or use of environmental water to achieve environmental outcomes." Environmental water in this context refers to water available under an access or delivery right that may be released from storage as well as the protection of flows in unregulated systems by implementing rules that limit the degree of water extraction to preserve particular parts of the hydrograph. Environmental outcomes include the conservation and restoration of biodiversity and ecosystem functions, including the protection of water quality. More recently the term "environmental watering" has also come to be associated with the use of floodplain infrastructure, such as levees, weirs, regulators, and pumps, put in place to deliver water to floodplain wetlands in cases where there is otherwise insufficient water to restore run-of-river flood events via reservoir releases. This form of engineered environmental watering may be critical during droughts, but does not emulate many of the processes associated with natural flood events.

Keywords

Environmental flows · Environmental water · Environmental watering · Engineered flooding · Floodplain wetlands

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Introduction

Environmental watering is a term that has gained traction, primarily in Australia, to describe the delivery of environmental flows to rivers and wetlands. The term was defined in Australia's federal Water Act (2007) as "the delivery or use of environmental water to achieve environmental outcomes" (Cth 2007). Environmental water in this context refers to water available under an access or delivery right that may be released from storage as well as the protection of flows in unregulated systems by implementing rules that limit the degree of water extraction to preserve particular parts of the hydrograph. *Environmental outcomes* include the conservation and restoration of biodiversity and ecosystem functions, including the protection of water quality. As pointed out by Hamstead (2007), the term "environmental water" has previously been associated with several different meanings leading to some confusion in the literature, and the term seems to make most sense within an entitlements framework in which the environment is seen as an entitlement holder similar to other water users.

Floodplain Infrastructure to Deliver Water

More recently the term has also been used more specifically to refer to the use of floodplain infrastructure, such as levees, weirs, regulators, and pumps, put in place to deliver water to floodplain wetlands in cases where there is otherwise insufficient water to restore run-of-river flood events via reservoir releases (Bond et al. 2014). The term "watering" is particularly apt in this context as the approach was first used to prevent the loss of floodplain wetlands and riparian forests during drought (Murray-Darling Basin Authority 2011). This form of environmental watering, also referred to as engineered flooding, has stimulated some concerns over its limited ability to mimic the natural biogeochemical exchanges and movements of biota to and from floodplains (Bond et al. 2014), and because of operational challenges and potential long-term opportunity costs and ineffectiveness under a future climate (Pittock et al. 2012).

While it is clear that environmental watering of floodplains using engineering approaches can be effective in maintaining floodplain vegetation during drought (Murray-Darling Basin Authority 2011), there is also some data highlighting the risks of the approach when trying to manage other biota. In particular the response by floodplain fish communities to artificial watering is much reduced relative to the benefits provided by overbank floods (Vilizzi et al. 2012; Stoffels et al. 2014). These differences likely arise because of the reduced degree of lateral connectivity from environmental achieved by means of floodplain infrastructure (both in terms of the duration and nature of lateral connections), but may also reflect the loss of important physical and chemical cues such as changes in temperature and water-chemistry associated with natural floods (Stoffels et al. 2014). One potential benefit of reduced lateral connections is the ability to maintain floodplain wetlands whilst excluding invasive species such as carp

C. carpio, although periodic overbank floods will likely mean that selective exclusion is only temporary.

As well as reduced benefits, artificial watering can also generate risks to aquatic fauna if not monitored and conducted at the right time of year. For example, "blackwater" events can occur as a result of carbon being leached from organic matter on inundated floodplains. Although such events occur naturally, they are more likely if water is artificially ponded on the floodplain for long periods (Howitt et al. 2007; Baldwin et al. 2011).

Long-Term Outcomes

The long-term outcomes from artificial environmental watering remain to be seen, but in some water stressed landscapes it is clear that the approach can achieve strong local positive environmental outcomes. Howevere, while the approach is efficient in

Table 1 Examples of environmental watering schemes from different parts of the world that incorporate some form of engineering infrastructure to drive floodplain inundation. See Bond et al. (2014) for additional examples

Location	Engineering approach	Ecological target	Proposed or constructed	Reference
Arkansas "Greentree reservoirs", USA	Levees, locks, weirs for impounding water around riparian hardwood forests	Increase habitat and foraging for waterbirds		King et al. (1998)
Azraq wetlands, Jordan	Pumping of water into wetlands	Increase wetland area for migratory birds	Pumping of water into wetlands since 1994	http://www.rscn.org.jo/RSCN/HelpingNature/ProtectedAreas/AzraqWetlandReserve/tabid/98/Default.aspx
Missouri River, USA	Managed flooding, levees, combined active and passive management	Improved bird and fish breeding	Re-operation of existing structures	Galat et al. (1998)
Murray River, Hattah Lakes, Australia	Pumping of water into wetlands and use of regulators to increase water residence time	Improve health of riparian and floodplain forests, increase aquatic habitat area	Partially completed	Murray-Darling Basin Authority (2011)
Yellow River Delta, China	Channels and sluice gates	Increased seasonal wetland flooding	Constructed	Cui et al. (2009)

terms of water volumes, the reduced degree of lateral connectivity and extent of lateral inundation reduces the flux of materials and biota that are known to drive the productivity of river-floodplain ecosystems (Galat et al. 1998; Jardine et al. 2012), which may limit larger-scale ecosystem benefits. For this reason, Bond et al. (2014) highlight the need to more explicitly understand the links between ecosystem processes and the hydrodynamics of flooding (such as residency time, water velocities, flow paths, etc.) where infrastructure is used to overcome the natural hydrologic drivers of inundation.

While there are a number of environmental watering schemes in place in various parts of the world (Table 1), successful implementation of engineered environmental watering over the long-term remains a relatively untested operational challenge.

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Environmental Flows and Integrated Water Resources Management

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Ian Overton

Contents

Abstract

Water resources in many of the world's river basins are highly modified and over-allocated. The resulting decline in ecosystem services impacts on human well-being. Since the 1990s, the legitimacy of environmental requirements alongside economic and social needs for water has been a part of Integrated Water Resources Management (IWRM). The environmental flows concept supports determining the required flow regimes but technical complexity, lack of ecohydrological data, institutional challenges and poor understanding of ecosystem services have restricted the implementation of environmental flows as a core component of IWRM. Integrating environmental flows into IWRM requires a participatory process with major stakeholders developing a shared vision on the interdependencies of water, food, and energy, resulting in flows for multiple benefits. Research should focus on simplified recommendations for integrated basin flow assessment, on operational rules that optimise multiple outcomes, and on uncertainties such as climate change and economic growth.

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Environmental flows · Integrated water resources management · Stakeholder participation · Ecosystem services · Integrated river basin management · ecohydrology

Introduction

In many of the world's river basins the water resources are over-allocated and/or highly modified, access to good quality water is limited or competitive, and aquatic ecosystems are degraded (MEA 2005). The decline in aquatic ecosystems can impact on human well-being by reducing the ecosystem services provided by healthy rivers, wetlands and floodplains. The over-allocation of water resources reduces the security of supply to all users, especially in river basins with highly variable flows. Sound river basin management includes multiple objectives, such as ecosystem integrity, social well-being and economic security and is fundamental to helping reduce poverty by providing income, clean water, sanitation, and sustaining the environment that provides many direct and indirect benefits (MEA 2005; Dudgeon 2010). However, water demands are often in conflict and consequently river basins and the livelihoods and the biodiversity they support are increasingly under pressure. The pressure is amplified by population growth increasing demands for energy, food and habitat, and climate change increasing the variability and, in many cases, reducing rainfall.

Integrated Water Resources Management and Environmental Flows

In most countries water planning and management traditionally considered water flowing to the sea as 'wasted' or 'surplus' flows. Rivers were harnessed through dams and diversions to the extent that was technically feasible. As degradation of the environment became evident, concerns grew that food and energy production cannot be sustained without key ecosystem processes. This gave rise to the Integrated Water Resources Management (IWRM) approach commonly used in many countries and widely promoted since the 1990s, which established the environment as a legitimate water user. IWRM requires the determination of water allocation among competing stakeholders including the environment, social needs and economic development and has been promoted as the key approach to managing water resources (Overton et al. 2014). IWRM should balance water use (of surface water and groundwater) for multiple water consumers.

Most countries seek to increase gross domestic product, but this ignores environmental impacts (Costanza et al. 2014). The impacts of water abstraction and manipulation on flow volumes, timing, temperature and water quality are felt most when the economic return from water use decreases due to declining ecosystem condition. This can occur when water quality declines, which increases infrastructure maintenance and possibly requires alternative and more expensive sources, such as desalinisation or piping from other basins. Moore and Forslund (2008) describe an IWRM goal as a societal choice point in water extraction that optimises cumulative benefits as a function of economic benefit and environmental condition. Nature is increasingly viewed as the main priority for water allocation to ensure environmental sustainability, as well as livelihood and economic development (Iyer 2005). Despite environmental needs being identified, it is in the interpretation and implementation of these principles where securing water for the environment can fail. Meeting environmental concerns may impose constraints on other water uses, so a new framework is required (Iyer 2005).

Along with determining environmental water requirements, managing environmental flows includes policy making for water allocation and operation, implementation and monitoring (Le Quesne et al. 2010). Manipulation of flows to achieve desired outcomes for public supply, food production, energy, mining and manufacturing have been implemented for many years, and more recently for ecological outcomes (Hirji and Davis 2009). The first step has usually been the determination of ‘minimum flow’ requirements to be released from dams, or limitations on water extractions, for downstream environmental water needs. However flow regimes are the drivers of biological diversity and ecosystem health. The complexity of determining the required flow regimes and the interdependencies between stakeholder outcomes has restricted the implementation of environmental flows as a core component of Integrated Water Resources Management. Policy, physical constraints and a lack of evidence in predicting environmental outcomes, can limit the implementation of desired environmental flows. The traditional approach of the trade-off between conservation and development is a conflict of values and interests. Ecosystem services are often unaccounted and improving them is generally not funded (Sarewitz 2004).

Limitations to Environmental Flow Regime Implementation

Several institutional failings inhibit flow reallocation, including lack of reallocation mechanisms, unclear property rights, the high political and economic costs of reallocation, the long-term nature of water investments, localism of water, weak institutional capacity, poor understanding of ecosystem services, over-lapping and/or gaps in institutional mandates and conflicting interpretations. All these challenge the legitimacy of applying environmental flows. In addition, traditional approaches assume stationarity, in contrast to approaches of ‘adaptive institutions’ (Sarewitz 2004). Further impediments to implementation of environmental flows include a lack of data on hydrology, ecology and ecosystem response to flow change, and thus on the flows required to deliver ecosystem services and desired levels of biodiversity. The barriers to implementation can be critical in highly contested and transboundary rivers. Complexity, variation, and uncertainty are inherent properties of social and natural systems and water resources management must strive to reflect this to ensure sustainability (Galaz 2007; Richter 2009). There is an overall lack of monitoring following environmental flows that could be used to adaptively manage riverine systems. Monitoring is essential to ensure environmental flow compliance and effectiveness (Arthington 2012).

Institutional structure and policy increasingly prioritise food and energy security over environmental health, especially in developing economies, increasing the difficulty of including environmental goals in river management. There is therefore a need to incorporate human well-being into the assessment of environmental flows (Meijer 2007). Full recognition of environmental benefits requires that all stakeholder values and benefits are taken into consideration which can be achieved using an ecosystem services framework. This includes both specific objectives set by society and the under-pinning ecological processes and functions that are required to meet these objectives, but may not be explicitly identified by those defining the objectives. Such a framework will protect the fundamental ecological processes that support water, food and energy security, and underpin a sustainable future (Overton et al. 2014). The ecosystem approach (EA) (Maltby et al. 1999) defined by the Convention of Biological Diversity provides a number of principles for managing ecosystems within resource development planning.

Integrating Environmental Flows and IWRM

Along with flows to improve ecosystem condition there are also flows for social values. These include cultural flows for maintaining traditional use and spiritual connection, such as with indigenous communities in Australia (Weir 2010), and water for religious events, such as in India (Jain and Kumar 2014). Other social values include recreation such as tourism and boating, maintaining places of aesthetic beauty and scientific interest. Some flows are required to sustain environments that are not natural, such as cold water releases in the Delaware River to support the trout fishing industry. The environmental flows needed to realise improved triple-bottom-line outcomes may not need to replicate the quantity or regime of natural flows. In most cases it will make sense to base environmental flows on the natural regime, but in cases where required ecosystem services were not provided under natural conditions, for example introduced fish, the regime may be based on required flows. This approach differs fundamentally from most environmental flow methods which use the natural flow paradigm as a starting point (Richter et al. 1996). Environmental flows within IWRM need to consider that novel ecosystems may be a more desirable outcome (Acreman et al. 2014).

An integrated environmental flows approach, as part of an Ecosystem Approach, identifies flow regimes using a participatory process, with representatives of major stakeholders developing a shared vision of basin goals (Dyson et al. 2003). Environmental flows provide a way to identify the interdependencies of the water, food, and energy nexus, and there are opportunities to use flows for multiple benefits. Environmental flows help to preserve the resilience of aquatic ecosystems, and thereby offer the promise of improved sustainability and well-being for people as part of broader ecosystems. Research and applications of improved methodology for the management of river basins should focus on implementation strategies for environmental flows that a) simplify recommendations in one integrated basin flow assessment; b) allow for operational rules that optimise triple bottom line outcomes; and c) take uncertainties such as climate change and economic growth projections into account (Overton et al. 2014). Water governance reforms to reallocate water to environmental flows will often depend on identifying and harnessing larger social, economic, and environmental drivers. The inclusion of environmental flows in IWRM will result in increased effectiveness of environmental outcomes along with many benefits to social well-being and economic return (Hirji and Davis 2009).

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Section XVIII

Wetland Management Planning

Mike Alexander

Wetland Management Planning: Overview

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Contents

Abstract

Management planning for wetlands is a dynamic, iterative, and adaptive process to identify the important features of a wetland, to decide what objectives people want to achieve, and then what needs to be done and which resources are needed to achieve these objectives. The planning process also involves communication with all stakeholders and is a joint learning process. The process starts with a preparation phase for consultation with all people and groups likely to affect or be

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affected by management decisions. Management plans need to address six main questions, leading to the following plan structure: (1) Policy and Legislation (Why are we here?), which addresses the policy and legislation applying to the wetland; (2) Description (What have we got?), with general information and maps, and information about the environment, the people involved, and the wetland itself; (3) Evaluation (What are the important features?), in which the features of the wetland which will become the focus of the management process are identified; (4) Factors (What are the important influences?), in which the internal and external influences on the wetland, both natural and anthropogenic, are identified; (5) Objectives (What do we want?) describing the desired outcomes of the management in a specific, measurable, achievable, relevant, and time-specific format with a clear vision and indicators; (6) Action Plan (What must we do?), in which the management actions for achieving the objectives are defined with a clear rationale and work program. The chapter provides recommendations for each of these sections.

Keywords
Management-planning · Planning · Conservation · Adaptive-planning · Adaptive resources · Objectives · Features · Communication · Protected areas · Evaluation factors · Performance indicators · Ecosystem services · Biodiversity · Favourable conservation status · Attributes · Work programme

Introduction

Planning is an essential component of almost all areas of significant human endeavor. Management planning for wetlands is not substantially different to any management planning process concerned with the safeguard of wildlife or natural features. This is irrespective of size or complexity, the conservation status, or whether a place or ecosystem is managed primarily, or only in part, for natural features. Planning is the intellectual component of management. It is always concerned with change, and so it must be a dynamic, iterative, and *adaptive* process. It is about identifying the things that are important (the features) and making decisions about what people want to achieve and what needs to be done. It is also about identifying the resources needed to achieve the objectives. Planning is about sharing this process with others so that agreement can be reached; it is about communication; it is about learning. It is perhaps the most important of all management activities, and the emphasis should be on thinking and decision-making, without devoting too much effort to the production of overelaborate, wordy documents.

Planning should always precede management: management is about taking control in order to obtain and maintain desirable conditions. The management of places, habitats, and species is nearly always about controlling influences (factors) or about taking remedial action following the impact of a factor. Control means the removal, maintenance, adjustment, or application of factors, either directly or indirectly. "Control" does not necessarily mean doing something: it could mean choosing to do nothing.

Preparation

Small is Beautiful

This is so true of planning. The resources made available for planning should always be proportionate to the complexity of a system, its features, and the total resource available for management.

Who Should Prepare a Plan?

Planning must be an inclusive process, which takes account of the interests of local communities and all other stakeholders. “A stakeholder is any individual, group or community living within the influence of a site or likely to be affected by a management decision or action, and any individual, group or community likely to influence the management of a site” (Alexander 2012). Everyone who is involved in the management of an area, or will be affected by management decisions, should at least be consulted and, whenever possible and appropriate, included in the decision-making process. The planning process should, as far as possible, encourage stakeholder involvement in all appropriate aspects of management planning and management (Thomas and Middleton 2003; Borrini-Feyerabed et al. 2004). Some of the most important people are those responsible for implementing the plan. There are many examples of plans that are produced by external consultants but never implemented. This can happen when managers are not involved and subsequently reject plans prepared by others with no experience of managing that particular area.

Protected areas are never isolated from their surroundings; it is usually only possible to safeguard them with the co-operation of others. Conservation managers have come to realize that they must involve and consider the interests of, stakeholders (particularly local communities) in the management of protected areas. The capacity to appreciate and enjoy wild places and wildlife, and not simply to regard them as an essential resource, is often restricted to individuals who do not have to depend on these areas for their livelihood. Protected area managers should recognize that local stakeholders can make a very significant contribution to managing a site. Once stakeholders gain a sense of ownership, there are many different ways in which they can help: for example, local knowledge and traditional skills are often essential, especially when these complement good science.

Plan Contents and Structure

All management plans contain six key sections, and these are best dealt with as a sequential series of questions (Alexander 2005).

- Why are we here?
- What have we got?

- What are the important features?
- What are the important influences?
- What do we want?
- What must we do?

Policy and Legislation: Why Are We Here?

Everything that is done on a site, in a protected area, or for a particular species should be guided by policies. These are high-level statements setting out the purposes of an organization. Policies are an expression of the organization's intentions, or a course of actions that it has adopted or proposed. The policies may have been adopted voluntarily, imposed by legislation, or they may be a combination of both.

Organizations need general policy statements that cover their entire operation; their responsibility for sites may be just one of a very much wider range of activities. It is rarely necessary for a plan to include details of all the policy statements of an organization, but a reference to their existence may be appropriate. The policies and, in some cases, the remit of an organization will determine how it manages its sites and for what purpose. Ideally, organizations should prepare policies that are specific to site management: these should be unequivocal and concise. Broad-based, general policies will be open to interpretation and are often difficult to apply.

On statutory conservation areas, management may be governed almost entirely by legislation. All places will be influenced, to some extent, by legislation. This can be an extremely important and positive factor, but legislation can, occasionally, severely restrict the ability to make decisions or carry out essential management. The important point is that planning must reflect, and be guided by, policies and legislation. In this way, management is always relevant to the requirements of the organizations responsible for protecting these places and fully compliant with the law of the land.

Description: What Have We Got?

The description of a site or protected area is a collation exercise. All *relevant* data are located and arranged under systematic headings. The order in which the headings are organized is of no particular significance, and initially each entry should be given equal value. A description is concerned with recording facts and not about making judgments. One exception is that any important shortfall in the information should be recorded, along with an assessment of its potential to impede the planning process. Information is collected and recorded: later in the plan these data will provide the basis for evaluation and decision-making. All data entries should be as succinct as possible.

Completing a description calls on a wide range of expertise, and it is unlikely that any individual would be able to complete all the sections without assistance. The planner adopts an editorial role, coordinating the input from others. Whenever existing material is reasonably relevant and concise, there is little point in rewriting

Typical contents of the description section in a management plan

(**Note:** these are the broad categories, in mostcircumstances there is a need for tiers of sub-divisions.)

1. **General Information**
   Location & site boundaries
   Tenure
   Management / organisational infrastructure
   Site infrastructure
   Map coverage
   Photographic coverage
   Zones or compartments
2. **Environmental information**
   Physical
   Biological
   Cultural
3. **People**
   Public interest / local communities.
   Access and tourism
   Current interpretation provisions
   Current educational use
   Current research use and facilities
4. **Landscape**
5. **Bibliography**

Fig. 1 Typical contents of the description section in a management plan

it for the plan. However, all reproduced descriptions should be attributed to the original author. All sections in the description will require review and update as additional information becomes available.

Figure 1 shows an example of the contents of a description for a typical plan. There is rarely any need to use all the subheadings for small, uncomplicated sites. The headings are best regarded as prompts to guide the structure of a description. It is not uncommon for plans to contain disproportionately large descriptions. Often, they can be larger than the remainder of the plan. Placing too much emphasis on the description must be a concern whenever resources are scarce. Plans do not need exhaustive or definitive descriptions, and it is important to differentiate between the things that we need to know and the things that we would like to know.

Evaluation: What Are the Important Features?

Planning always requires a focus. The key components of every management plan are the objectives, which define the purpose of management. This is quite different to the management processes. Purpose should always be focused on the most important components: the features. Nature conservation features can be habitats, communities, or populations. Other features of interest can include geological, archaeological, historical, and cultural features. Ecosystem services can also be treated as features.

For protected wetlands, the presence of biodiversity or natural features will have been the basis of selection or designation. Evaluation is simply the means of identifying these features and confirming which will become the focus for the remainder of the planning process. It is about asking a question of each potential feature: is this feature, in its own right or in association with other features, sufficiently important to be regarded as one of the prime reasons for maintaining the protected area?

The scale at which features are defined is always an important consideration. Whenever possible, the focus should be at system or catchment level. When this is not possible, the habitat level can be appropriate. Once planning is reduced to safeguarding features at a community level, particularly when the communities are dynamic (e.g., within a salt-marsh), objectives can become inappropriately prescriptive. When species are recognized as independent features, it is essential that the habitat which supports them is also treated as a feature.

Unfortunately, in reality, there are rarely sufficient resources even to manage the most important features on a site. Consequently, the planner may have to be selective and, in extreme cases, restrict management to features of national and international status. There will always be a need to draw a line somewhere. Sometimes, there are conflicts between features. These can often be resolved by understanding the relationship between the different features.

The characteristics, qualities, or properties of a feature are called its attributes. Attributes are used to quantify the condition of a feature so that it can be monitored (see Performance Indicators below).

Ecosystem Services as Features

Ecosystem services are defined as the benefits provided by the natural environment to humans (MEA 2005). The Millennium Ecosystem Assessment framework is a widely accepted method that has categorized ecosystem services into four broad categories:

- Supporting Services, such as soil formation, nutrient cycling, water cycling, and primary production. These underpin the provision of the other "service" categories
- Provisioning Services, such as food, fiber, fuel, bio-materials, and water
- Regulating Services, such as climate regulation, flood protection, pollination, and air/soil/water quality
- Cultural Services, such as education, cultural heritage/sense of place, health, recreation, tourism, and aesthetic value

The Economics of Ecosystems and Biodiversity (TEEB) is a global initiative focused on drawing attention to the economic benefits of biodiversity. Its objective is to highlight the growing cost of biodiversity loss and ecosystem degradation (Russi et al. 2013). The TEEB categorization of ecosystem services is similar to that of the MEA, but the definition of supporting services is extended to become "Habitat and supporting services." *"The importance of ecosystem services, a* *concept that focuses on the benefits of nature to people, society and the economy (i.e. an anthropocentric view of the importance of biodiversity), needs to be seen together with the intrinsic value of biodiversity – the value of biodiversity for its own sake .... Biodiversity is the quantity and variability among living organisms within species (genetic diversity), between species and between ecosystems. Biodiversity is not itself an ecosystem service but underpins the supply of services"* (TEEB 2008). This is further defined by TEEB:

> *Habitats for species*: *Habitats provide everything that an individual plant or animal needs to survive: food; water; and shelter. Each ecosystem provides different habitats that can be essential for a species' lifecycle. Migratory species including birds, fish, mammals and insects all depend upon different ecosystems during their movements.*
>
> *Maintenance of genetic diversity*: *Genetic diversity is the variety of genes between and within species populations.* (TEEB 2008).

It is important that ecosystems are valued for what they provide, for example, a carbon store, agriculture, forestry, fisheries, and leisure. In that sense, the recognition of "ecosystem services" as a means of demonstrating our dependence on the natural world is welcome. However, if the value of these services becomes the only way in which ecosystems are valued or measured, then all good intentions towards maintaining biodiversity could fail. Unfortunately, there is a common assertion that if ecosystems are delivering the desired goods and services then biodiversity will be conserved (Norton 1991). A logical, but dangerous, consequence is the replacement of policies and objectives concerned with conserving biodiversity with those concerned with maintaining ecosystem services. It is wrong to believe that if an ecosystem is delivering the desired services, then biodiversity will be conserved. In reality, a seriously depleted ecosystem that fails to meet its biodiversity potential, and with many rare and endangered species absent, can provide valuable, but not necessarily sustainable, ecosystem services. Many ecosystem services are provided by individual species or groups of species, and not by intact, functional ecosystems. Some of these species may be robust and resilient to adverse anthropogenic factors. Some species can be replaced by others which provide the same or similar services. An example is woodland, which can provide many services including carbon sequestration, the supply of water, and erosion control. The presence of specific native tree species, along with their associated and often dependent flora and fauna, is far less important in this context than the presence of any tree species, including alien species. A commercial coniferous woodland plantation, because it can provide timber, acts as a carbon store, prevents soil loss, contributes to flood management, and provides leisure opportunities, could easily be considered as being more valuable than, for example, a native broad-leaved woodland.

Biodiversity can be a by-product of management aimed at delivering ecosystem services, but if the quantity or quality of these services is the only measure of achievement applied to ecosystems, there is no guarantee that by default biodiversity will be conserved. The value of ecosystem services should never become a surrogate measure of biodiversity. The only measure of biodiversity is life itself: the variety of species, communities, and habitats.

There is also an issue of the "burden of proof." If biodiversity is only valued in terms of the ecosystem services that it delivers to people, the burden of proof in defending biodiversity, habitats, or species is placed on the conservationists: they will have to prove that the habitats or species have value to people. If however the intrinsic value of biodiversity is recognized, then the burden of proof shifts to society at large. Intended actions would need to be justified whenever they place biodiversity at risk (Meff 1997). Perhaps these divisions should be blurred. Individuals or organizations might be motivated by intrinsic values of habitats and species, but that is not necessarily in contradiction to promoting ecosystem services.

The most important point is that ecosystem services are regarded and treated as features in the adaptive management planning process. Whenever conserving biodiversity is a concern, objectives for the wildlife features must be prepared, and, in addition, ecosystem services, at least the more tangible and quantifiable services, can be treated in exactly the same way. SMART objectives should be prepared for the desirable ecosystem services.

Factors: What Are the Important Influences?

A factor is anything that has the potential to influence or change a feature or to affect the way in which a feature is managed. These influences may exist, or have existed, at any time in the past, present, or future. Factors can be natural or anthropogenic in origin, and they can be internal or external.

Unfortunately, the use of language in conservation is not standardized. Different organizations and individual authors use different words to say, more or less, the same thing. Factors have been given a variety of different labels. The Millennium Ecosystem Assessment (MEA 2005) and The Economics of Ecosystems and Biodiversity (TEEB 2008) talk of "drivers of change." Their definition: "A *driver is any natural or human-induced factor that directly or indirectly causes a change in an ecosystem.*" Thus, a driver is simply another name for a factor when applied to an ecosystem. The word is unimportant, but the application of factors in planning is essential.

The management of habitats and species is nearly always about controlling factors or taking remedial action following the impact of a factor. Control means the removal, maintenance, adjustment, or application of factors, either directly or indirectly. For example, grazing can be a very important factor when managing some wetland systems. Grazing can be removed, reduced, maintained, increased, or introduced. The ability to achieve conservation objectives will always be constrained by the ability to control factors. Factors are considered at several key stages in the planning process:

- Monitoring key factors will provide essential performance indicators.
- If current and future factors can be identified, it will in some cases be possible to predict the direction of change and to identify the attributes of a feature that are most likely to demonstrate the change.

Management is concerned with controlling factors, and control often requires a management input. Knowledge of the more significant factors can be used to identify the management requirements. There is a complication: although individual factors may have a limited impact on a feature, in combination they can become a serious issue. This means that factors should be considered both individually and collectively.

Certainty that all factors have been identified never exists, nor should it be assumed that the implications of each factor are fully understood. However, management planning is a process that can only react to what is known and understood at any given time.

There are many good reasons for providing a standard range of headings and subheadings that can be used as an aide-memoire to help identify the wide range of factors that have a potential to influence the management of features. These will vary from place to place, but there are four main divisions that are constant:

- Internal anthropogenic factors
- Internal natural factors
- External anthropogenic factors
- External natural factors

There is always a need to distinguish between internal and external factors. This is mainly because internal factors are usually controlled by direct on-site intervention, while external factors are rarely controlled through direct action by the site managers. The indirect control of external factors is usually through influencing others, informally or formally, for example, by providing evidence when developments are planned.

Even when external factors cannot be controlled, they cannot be ignored either. Where there is evidence to demonstrate that external factors are damaging a feature, and particularly when this happens on statutory sites, the evidence may be used to help justify political or legislative changes. External global factors, for example, climate change, are extremely difficult to deal with. These all-pervading influences will probably have a greater effect on the ability to conserve wildlife than a combination of all the other factors. In circumstances where change is taking place and defensive measures are possible, taking action may be justified if the impact of global change can be delayed. There is, of course, a counter argument: if these changes are taking place, why not accept the inevitable? But by keeping options open for as long as possible, some choices for the future may be provided.

The division between natural and anthropogenic factors (human influences or the consequence of human activities) is also important because it will help to differentiate between factors which are regarded as having a positive influence and those which are considered negative. This can be an extremely difficult division as it is often impossible to differentiate between changes to a feature which are the consequence of natural processes and those which are a consequence of anthropogenic processes or a combination of the two.

Objectives: What Do We Want?

Objectives should define the outcomes of management: the things that we want to achieve must be quantified so that they can be monitored (Thomas and Middleton 2003; Ramsar 2002). Objectives lie at the very heart of a management plan and are the single most important and essential component. Regrettably, the development of objectives is often the weakest section in a plan. Many plans contain excellent site descriptions and comprehensive lists and descriptions of the management actions, but critical information about what they want to achieve is missing. Where an indication of outcomes is given, it is often vague or ill-defined and not recognizable or measurable. Planners will talk, for example, of "maintaining and where possible enhancing the ecological value of all natural and semi-natural habitats." This is an approach that fails to inform the reader what is meant by "maintain" or "enhance." What state should be required of a feature to determine that it is worthy of maintenance, and when is enhancement complete?

There is a very convincing argument for applying the SMART definition of an objective to management planning (Clarke and Mount 1998; Eurosite 1999; Alexander 2012). Generally, the acronym SMART stands for:

- Specific

Objectives for features must specifically address the feature. Specific also implies that objectives are clearly defined and not open to different interpretations. This is particularly important when preparing objectives for statutory sites.

- Measurable

Objectives must be quantified and measurable. If they are not, we will never know that they are being achieved.

- Achievable or Aspirational

The business use of "SMART" is concerned with identifying achievable objectives. Many commercial definitions also suggest that sufficient resources must be available for an objective to be considered achievable. However, this is not appropriate and should not apply to objectives for nature conservation. Provided that an outcome could be achieved if resources were available, then the objective should be considered achievable. In the world of nature conservation, there is an inevitable time lag between the application of management and changes to the status of a feature. It can take decades, even centuries, to obtain the management objectives. It is also important to recognize a long-term obligation towards maintaining the conditions defined by the objectives. In other words, objectives should not be time limited. It is probably wiser to regard conservation objectives as aspirational.

- Relevant

Objectives must be relevant and must comply with the strategies, policies, and legal obligations that govern the organization responsible for managing the site or feature.

- Time-based

  This is not an easy concept when applied to nature conservation. Start times are relevant and this would usually be the time when an objective is adopted for a feature and when management commences. Management plans are occasionally written for a specified period, often 5 years; they are then rewritten or revised. Such plans sometimes contain short-term objectives describing a condition that should be achieved within the planning period (NCC 1983). Occasionally, and particularly when dealing with restoration sites, there are good reasons for considering short-term or intermediate objectives. These can be extremely useful when there is a need, for financial or political reasons, to demonstrate achievement at specified stages. However, short-term objectives are only valid if they are entirely consistent with obtaining a long-term objective.

  In most instances, the concept of a time-based objective takes on a different meaning. The objective for a feature is a description of the condition that is to be achieved, and thereafter maintained, in the long term. The problem is that there is no widely accepted definition of long term. Perhaps it is simply something that exists in the mind of the beholder; it is as far ahead as anyone can envisage. All natural features will change, and the degree and direction of change is not always predictable. An objective is a reflection of the values, knowledge, and aspirations at the time that it is written.

  The SMART definition, with some modification, is particularly relevant to objectives that define outcomes for habitats and species.

  Adaptive management processes are only possible if both management and objectives are reviewed, and if necessary modified, at specified intervals. The length of the interval will be determined by the confidence in the objective, and this will be influenced by a range of different issues. The most important are:

- The knowledge and understanding of a feature; features, species, and habitats often need to be managed when there is very little available information.
- The natural dynamics of a feature. Some habitat features, for example, sand dunes, can be very dynamic, even ephemeral; other features can be very stable.
- The quality of the scientific evidence that is available.
- The direct experience of, and competence in, managing the feature.
- Changing environmental factors, for example, global climate change.
- Changing human values and perceptions.

There is often no need to modify an objective. The review may confirm that an objective is appropriate and should remain unchanged (Alexander 2012).

Preparing Measurable Objectives

It is so much easier to recognize the need for measurable objectives than it is to achieve that aim in practice. The main issue is that it is not possible to quantify or measure everything that is required of an ecosystem, a habitat, or a population.

Even when it is possible to quantify some of the attributes, the resources required to monitor them may not be available. Care should be taken that an objective is not reduced to a list of the things that organizations can afford to measure. In other words, "if we can't count it, it doesn't count."

This issue can be resolved by accepting that although it is not possible to measure everything, the aspirations or vision for a feature can be described and, most importantly, shared with others. A range of performance indicators can be identified and monitored which will provide some of the evidence needed to assess the feature. So, an objective can become a composite statement that contains a vision and performance indicators. It is very important that these two elements are always linked and recognized as a single entity. Monitoring, surveillance, and surveys are undertaken on sites for a range of different purposes. These activities must not displace, or become confused with, the need to specifically monitor management objectives.

Vision

The main purpose of the vision is communication. The fundamental need to consult and engage with stakeholders is no longer questioned. Planners must recognize that the audience for their plan, particularly for the vision, should include anyone with a legitimate interest, and most of these people will not be scientists. Visions are best written in plain language, but this must not mean that the statement is diminished in any way.

The obligation of managers towards habitats and populations could simply be to obtain the optimal condition or status for these features. "Optimal" has been defined as Favorable Conservation Status (FCS) which, as a concept, is recognized throughout Europe as a consequence of its adoption by the European Union and globally as a result of it use by the Ramsar Convention (see definition of Favorable Conservation Status of wetlands). The practical application of FCS can provide a useful and appropriate basis for defining the desired status of habitats and species at any geographical scale, from the entire geographical range to a defined area within a site. A generic vision for all wildlife features could be "to maintain the feature at FCS." Any generic statement is useful in that it can provide broad guidance, but is useless in any specific application. FCS is a good starting point and provides a framework around which a vision may be constructed. An example of a vision, along with its relationship to the definition of FCS, is given in definitions of favorable Conservation Status and Performance Indicators).

Performance Indicators

Performance indicators can be used to provide the evidence that a feature is at FCS or otherwise. This evidence will rarely be sufficient to allow a conclusion to be proven beyond any reasonable doubt, but we are dealing with wildlife and not criminal law.

If performance indicators are used, a feature should only be considered to be favorable when the values of all the performance indicators fall within their specified range. All performance indicators must be monitored, that is, their entire purpose. There are two main types of indicators that may be used:

Quantified attributes with limits which, when monitored, provide evidence about the condition of a feature. Attributes are the characteristics, qualities, or properties of a feature which are inherent to, and inseparable from, the feature:

- Attributes should be indicators of the general condition of a feature.
- Attributes must be measurable.
- Attributes should, whenever possible, be indicators of what may happen in the future rather than of something that happened in the past.

Factors with limits which, when monitored, provide the evidence that the factors are under control or otherwise. This is entirely consistent with the reliance on FCS for the vision. The definition of FCS for habitats requires that the factors are under control, and for species the definition implies that "the population must be sustainable in the long term," that is, the factors are under control.

Action Plan: What Must We Do?

Rationale

The rationale is the process of identifying, in outline, the most appropriate management for the various features (Alexander 1996). There are two distinct stages:

- Status of the Feature

The first stage in the rationale is concerned with the difference between the current status of a feature and the required status for a feature: the required status is the objective. It is the difference between "what we have" and "what we want." This assessment is only strictly possible when objectives have been completed and the performance indicators for the feature have been monitored. For example, if a feature is found to be at Favorable Conservation Status and there is evidence that it has been so for some time, then it can be concluded that any management currently in place is appropriate and effective. Conversely, if a feature in unfavorable, it can be concluded that management is inappropriate or that it has not been in place for long enough.

- Factors

The second stage in the rationale considers the relationship between factors and the condition of the feature, and the implications of the factors for management. Factors, or at least the key factors, are agents of change, and conservation management is nearly always indirect, concerned with keeping factors under control. Once the key factors are known, it is possible to identify those which require some form of management. For example, water level is a common factor for many wetland sites and is a particularly important issue when wetlands have been drained in the past. Alien invasive species are another obvious factor, often requiring management intervention. Management can occasionally be even further removed from the feature. For example, illegal hunting or fishing in protected areas can be a serious factor, but it is the underlying or secondary factors, hunger, and poverty, which should be the target for management.

Work Programs and Projects

This is a rather "mechanical" section in a plan. It is entirely concerned with preparing work programs and schedules, and it requires a methodical and structured approach. An action plan begins with the preparation of descriptions for every management, recording and administration project, previously identified in outline when the performance indicators were selected, or later in the rationale. Together, these projects describe all the work that needs to be carried out in order to meet the objectives.

Each project is described in sufficient detail to enable the work to be completed and should contain a range of basic information (i.e., when and where the work should be completed, who should do the work, the priority, what it will cost, etc.). This detailed information can then be aggregated and used to produce a wide range of work programs, for example, annual programs, programs for a specified period, programs for an individual, financial programs, long-term programs.

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Management Planning for Nature Conservation: Core Principles

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Tom Hellawel

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Abstract

This chapter summarizes the core principles for preparing a management plan for nature conservation, particularly for European Natura 2000 and Ramsar sites, as prepared and endorsed by participants in a workshop held in 2008 in North Wales, UK, representing 11 European countries. General principles included, among others, that in addition to the management of nature conservation features, plans should also address stakeholder interests, cultural aspects, visitor management/tourism, education and interpretation, and social and economic aspects; that the precautionary principle is important in the context of conservation management and planning; and that planners should integrate site planning with wider sectorial and land use plans. More specific recommendations focused on the importance of stakeholder involvement and an inclusive approach which takes account of the interests of all stakeholders and encourages their involvement in all appropriate aspects of planning and site management. The planning process should be adaptive, with an iterative and developmental process of which monitoring is an essential component. Management plans need to include a description of the

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features of the wetland, and management objectives should be described in terms of the desired state of the key features of the site. All management plans should contain an action plan which identifies the total resource requirement and costs all the activities required to obtain and maintain features at favorable conservation status. The action plan should identify priorities for all management activities and identify who will be responsible for implementing the activities.

Keywords
Wetland management planning · Conservation · Adaptive management · Favorable conservation status · Precautionary principle

Introduction

The following statement outlines the core principles that should be applied when preparing a management plan for nature conservation, and which apply particularly for European Natura 2000 and Ramsar sites. These principles were identified during a workshop *Establishing and Confirming Management Planning Principles on Natura 2000 and Other Conservation Sites,* held in 2008 at Plas Tan Y Bwlch, the Snowdonia National Park Study Centre in North Wales, UK. The workshop was organized by the Conservation Management System Consortium (CMSC; an international partnership of conservation organizations from the UK and the Netherlands; http://www.cmsconsortium.org/) to review the wide variety of current management planning protocols and identify core management planning principles that should be applied to any conservation management plan. The statement was prepared and endorsed by 21 participants, representing 11 European countries.

General Principles

- Ideally, there should be one comprehensive plan for multidesignation sites.
- In addition to planning the management of nature conservation features, plans should also consider stakeholder interests, cultural aspects (including historical, archaeological, religious and spiritual interests), visitor management/tourism, education and interpretation, and social and economic aspects.
- The precautionary principle is important in the context of conservation management and planning. It should guide the planning process and influence the way in which sites, habitats, and species are managed.
- Planners should recognize the need to integrate conservation site planning with wider sectorial and land use plans.
- The planning approach should be as uncomplicated as possible (the simpler the better). A management plan should be as large as the site requires and no larger.

- Organizational support for the planning process is essential, and this should include a formal approval process.
- Management plans should be easily understood by everyone who has an interest in the site. This will include people who do not have a scientific or technical background. The language used in the plan should, whenever possible, be plain and accessible to all.
- Plans, and in particular plans for large, complex sites, should include a summary. These can be presented as text, but the addition of annotated maps and illustrations will help to explain issues.
- Individuals involved in managing a site should, whenever possible, have an involvement in the planning process and, in all cases, ownership of the plan.
- A record should be kept of all the individuals engaged, at any level, in preparing the plan.
- All consultees and advisors, individual and corporate, should be acknowledged in the plan.
- The plan should contain a glossary of terms.
- Plans must be implemented.

Stakeholder Involvement

Conservation managers must recognize the need to adopt an inclusive approach which takes account of the interests of all stakeholders and, as far as possible, encourages their involvement in all appropriate aspects of management planning and site management. One of the key issues when building and maintaining successful relationships is to have a shared appreciation of what can and cannot be negotiated.

The Planning Process

- Management planning should be a continuous cyclical, iterative, and developmental process (adaptive planning).
- Monitoring must be recognized as an integral and essential component of any planning process.
- It is good practice to record all actions undertaken in accordance with a plan.
- Factors must be identified and integrated in the planning process.
- All planning and management actions should incorporate current best practice and be open to new and innovative ideas.
- Management should be reviewed continually within a timescale that is appropriate to the features. Fragile and vulnerable habitats or populations will require more frequent attention than robust and secure features.
- Routine internal management reviews should be supplemented with formal reviews at predetermined agreed dates. It may, in some cases, be appropriate to hold external reviews.

Information

Plans require a descriptive section which contains, or provides reference to, the information that will be *needed* to help decide what is important and to undertake the planning process. This is a collation exercise and is generally not dependent on the generation of new information. Further information requirements should be identified during the planning process.

Features as a Focus

- Features should provide the focus for management planning. Management by defining conservation outcomes for features is a reflection of the legal requirement to protect specified features on statutory, and other, sites. This is of particular relevance to Ramsar sites. The desired status for each feature is defined, and these are the *management objectives.*

Objectives

- Objectives should lie at the very heart of a management plan. They are the outcomes of management and the single most important component of any plan. An objective is the description of something that we want to achieve. Wildlife outcomes are habitats, communities, or populations (features) at Favourable Conservation Status.
- SMART objectives, as generally applied to business, can, with modifications, be applied to wildlife objectives.
- Objectives for conservation features must not be diminished to accommodate a shortfall of resources. An objective should be an expression of the legal and moral obligations towards features on sites.
- Objectives should, when appropriate, take account of natural and other processes.

The Action Plan

- All management plans should contain an action plan which identifies the total resource requirement.
- The action plan should identify and cost all the activities required to obtain and maintain features at Favourable Conservation Status.
- The action plan should identify priorities for all management activities.
- The action plan should identify all individuals or organizations that will be responsible for implementing the activities.
- The action plan should identify realistic, achievable, and effective management actions.

Adaptive Management Planning

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Mike Alexander

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Abstract

Planning should be a continuous, iterative and developmental process. Adaptive management can be applied to any site, regardless of size. It is, in its more complex form, an approach to experimental management that enables changes to be linked to cause and to management operations. This section briefly considers the main versions of adaptive management and introduces a minimal version: a basic approach, but with some significant differences. It is not experimentation but a simpler system based on monitoring and then, if necessary, modifying management. The cyclical, adaptable management process allows site management to: respond to natural dynamic processes; accommodate the legitimate interests of others; adapt to the ever-changing political and socio-economic climate; and, in the long term, succeed, despite uncertain and variable resources. There is a continuum, from trial and error to full scale active adaptive management, and somewhere within this range there lies a version of adaptive which is appropriate for any given place and time. Management reviews are an integral and essential component of the adaptive management process.

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Keywords
Adaptable management · Adaptive management · Audit, Monitor · Objective · Rationale · Review

Introduction

There are very few certainties in life, but we can be sure that our natural environment and the wildlife that it supports will change. It has always changed. The values that we apply to our environment and its components will change, as will the condition of habitats and populations. The management actions that we take will also change. As a generality, we must learn to accept, and even welcome, change.

Adaptive management as a concept can be traced back to Frederick Taylor in the early 1900s. Taylor was a mechanical engineer who tried to improve industrial efficiency and laid the foundations for a scientific approach to management. However, it is Holling, an ecological theorist, and his colleagues who developed adaptive management in the late 1960s (Holling 1978; Walters 1986).

Versions of Adaptive Management

There are three divisions of adaptive management (Allan and Stankey 2009; Holling 1978; Lee 1999; Stankey et al. 2005; Walters and Holling 1990). However, when considering these it is important to recognize that there is a continuum from trial and error to full-scale active adaptive management, and somewhere within this range there lies a version of adaptive which is appropriate for any given place and time.

1. Evolutionary adaptive management
   This is a very simplified approach but much better than doing nothing. In essence, it is about learning from experience, sometimes described as "observational monitoring." It is, or should be, a formalized version of trial and error.
2. Passive adaptive management
   This approach focuses on the implementation or application of management techniques or policies which have a successful track record or are believed to represent "best practice." Implementation is followed by monitoring, review, or evaluation of the results, and, if necessary, the technique is modified and the cycle repeated. This approach will identify effective management techniques. Adaptive management treats human interventions (management) as experimental probes (Lee 1993). Consequently, the need for replication and the use of controls is also implied. Without controls it will not be possible to differentiate between changes that would have happened had the management not been applied or changes that are a

result of other unidentified factors. This process clearly provides a structured approach to learning.

3. Active adaptive management
   This shares the basic concept of passive adaptive management but is specifically designed to test various hypotheses (Lee 1993). It involves the application of a range of management techniques and should eventually identify "best practice."

The characteristics common to most definitions of adaptive management are:

- There should be clear and specific objectives.
- Monitoring with a feedback link to management or policies is essential.
- Learning is an explicit objective.
- There is a social or stakeholder dimension.
- Management is not delayed by uncertainty.
- Adaptation is essential. This means changing management interventions or assumptions in response to new information gained through focused monitoring.

Some definitions also suggest that adaptive management should:

- Retain a focus on statistical power and controls.
- Use computer or conceptual models.

Adaptive Management: A Minimal Approach

This is the process advocated by the Ramsar Convention on Wetlands (Ramsar Convention 2002). It is a modified version of "trial and error," which is a useful and entirely relevant management approach. It is useful because it is a process where management is implemented in spite of uncertainty. In reality, it is never certain that management is entirely appropriate. However, provided that management is directly linked to a clear, quantified, and measurable objective for a feature, the condition of the feature can be monitored before and after the implementation of management. This version of adaptive is not experimentation. It is not dependent on replicating management actions or establishing control plots.

This minimal adaptive approach can be used to:

- Demonstrate that management is appropriate and effective and that resources are well spent.
- Ensure that we learn from experience.
- Ensure continuity of effective management.
- Encourage and enable communication between managers and stakeholders, both within and between sites and organizations.

The adaptive management cycle consists of the following steps (Fig. 1):

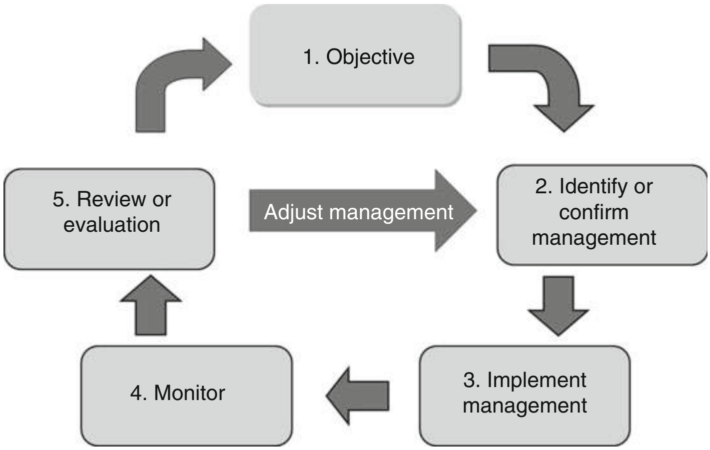

Fig. 1 The adaptive management cycle

1. Prepare an objective for each feature.
   An objective is prepared for each feature, and performance indicators (which will be monitored) are identified for each objective.
2. Identify or confirm management – the rationale.
   Decisions are based on the best available information, evidence, expertise, and experience. Management is implemented for a predetermined period of time.
3. Implement management.
   Management which is believed to be the most appropriate is applied. All management activities must be carefully recorded.
4. Monitor the condition of the feature.
5. Review or evaluation.

The results of monitoring, along with reports of management activities and any other relevant observations (including external information), are considered. Three questions should be answered:

- Is there any reason to change the objective?
- What is the condition of the feature?
- Are the factors under control?

If the feature is in a favorable condition and the factors are under control, then management is assumed to be appropriate and effective. If the feature is unfavorable and/or one or more of the factors are not under control, management should be reconsidered and, if necessary, changed. In some circumstances, it is concluded that a particular management regime has not been in place for long enough for the required changes to have taken place. In that case, providing there are no signs of deterioration, the existing management is continued for a longer period.

The adaptive management process is both cyclical and repetitive. Adaptive management recognizes that wildlife managers may be unsure of their objectives and management requirements. However, each time a management cycle is completed, the management activities are tested, new knowledge is obtained, and skills are improved.

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Performance Indicators and Monitoring 259

Mike Alexander

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Abstract

Wetland managers need a warning when things are going wrong and confirmation when things are satisfactory. The objectives of wetland management and conservation are defined as the required conditions of the features of the wetland (the formulated standard) that are achieved through adaptive management. Monitoring consists of repeated measurements to ensure that these objectives are being met. Objectives are measured by linking them to performance indicators. Indicators can be quantified attributes, such as of the area occupied by the habitat or by one or more constituent communities; or the presence, abundance, relative proportions, or distribution of individual species. Indicators can also be specific limits for factors that influence key features, within which they are considered acceptable are defined. Performance indicators should provide managers with the confidence that they can detect changes in a feature. Managers must be able to differentiate between undesirable and desirable change, and it is usually the direction of change that will influence management decisions. The chapter provides examples of performance indicators of features and factors for a raised bog ecosystem.

Keywords

Wetland management planning · Wetland conservation · Adaptive management · Objectives · Features · Performance indicators

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Introduction: What is Monitoring?

Monitoring, according to common use, can mean almost any kind of measurement, including survey, census, and even research. In fact, it has such a broad range of meanings that without a clear definition, it is almost useless for planning purposes. Managers need a warning when things are going wrong and confirmation when things are satisfactory. This means that they have to decide what conditions they require for a feature: this is the objective or formulated standard. They must then make repeated measurements to ensure that their objective is being met. For nature conservation purposes, monitoring is defined as surveillance undertaken to ensure that formulated standards are being maintained (JNCC 1998).

Performance Indicators

The integration of monitoring in the adaptive planning process occurs when the objectives for the features are formulated. An objective must be measurable, and this is achieved by including *performance indicators* that are directly linked to, and part of, the objective. Two different kinds of performance indicators are used to monitor an objective for a habitat or population. These are:

1. Quantified attributes, which are the characteristics of a feature that, when monitored, provide evidence about the condition of a feature. Attributes are selected because they provide the evidence that is needed to assess the condition of the feature. Examples of attributes for a habitat are (Alexander 2013):
   – The size of the area occupied by the habitat or by one or more constituent communities
   – The distribution of the habitat or of one or more constituent communities
   – Physical structure (a wide range of attributes is possible here, and they are very feature specific)
   – Presence, abundance, relative proportions, distribution of individual species, or groups of species, indicative of condition
   – Presence, abundance, relative proportions, distribution of individual species, or groups of species, indicative of change
2. Specific limits for factors that influence key features. A factor is anything that has the potential to influence or change a feature. The key factors are identified, and the limits within which they are considered acceptable are defined. The factors can then be monitored to provide the evidence that they fall within the specified limits, in other words to demonstrate that they are under control or otherwise (Alexander 2013).

Performance indicators were developed in recognition of the fact that the entirety of any biological feature cannot be measured. It is important that sufficient performance indicators are identified and monitored to provide managers with the

Table 1 Examples of performance indicators for a raised bog (the management objective is given in the definition of favorable conservation status) (NRW 2013)

Performance indicators for features' condition: raised bog		
Attribute	Attribute rationale and other comments	Specified limits
A1. Extent of active raised bog	The lower limit is the extent mapped in 2003.	*Upper limit*: 100% of primary bog surface *Lower limit*: 186 ha
A2. Condition of active raised bog	A low cover of *Sphagnum* is indicative of degradation including loss of peat forming capacity. Characteristic raised bog Sphagna indicating rainfall as the prime source of waterlogging are *Sphagnum cuspidatum*, *S. pulchrum*, *S. tenellum*, *S. capillifolium*, *S. papillosum*, *S. magellanicum*, *S. subnitens*, *S. austinii*, and *S. fuscum.* Hummock sphagna: *S. capillifolium, S. papillosum, S. magellanicum, S. austinii*, and *S. fuscum* are important peat builders and good indicators of high-quality raised bog. Encroachment of *Molinia* indicates loss of quality (often due to aerial pollution, drainage, fire, or a combination of these). The monitoring plots established in 2003 are considered to give a fair reflection of the overall habitat condition.	*Upper limit*: not required *Lower limit*: >50% of sample points in "active raised bog" monitoring plots 3, 4, and 7 (established 2003) have (i) 25% or more ground cover of characteristic raised bog *Sphagnum* species (ii) Presence of one or more hummock-forming *Sphagnum* species per sample plot (iii) *Molinia caerulea* is absent *and* (iv) The *Rhynchosporion* depressions on peat substrates of the bog (see A4.) are above the lower limit for extent/ condition
A3. Typical microform hummock/hollow patterning	The surface of the mire expanse shows the typical microform hollow/hummock patterning, with vegetation dominated by species of bog moss in lawns and hummocks. The ratio of wet *Sphagnum* hollows to hummock and ridge microforms is extremely important.	*Lower limit:* microform hollow/hummock patterning covers at least 80% of the primary active raised bog surface *Upper limit:* not required *Lower limit:* 20% wet hollow vegetation within a 10 m radius of any point on the higher bog surface *Upper limit:* 30% wet hollow vegetation within a 10 m radius of any point on the higher bog surface

(*continued*)

Table 1 (continued)

Performance indicators for features' condition: raised bog		
Attribute	Attribute rationale and other comments	Specified limits
A4. Extent and condition of depressions on peat substrates of the *Rhynchosporion*	"*Rhynchosporion"* vegetation is defined as: Permanently wet topographic hollows/depressions in which (a) *Sphagnum cuspidatum* and *S. pulchrum* form >20% cover (b) *Rhynchospora alba* is abundant, or *Drosera anglica* is present (c) *Molinia caerulea* and *Trichophorum cespitosum* are absent.	*Upper limit*: 70% *Lower limit*: 15% sample points in plots 1, 2, 5, and 6 are referable to "*Rhynchosporion* pool" vegetation

confidence that they can detect both undesirable and desirable changes in a feature. It is also essential that managers can differentiate between undesirable and desirable change. It is usually the direction of change that will influence management decisions. In order to achieve this, limits (upper and/or lower) are applied to the performance indicators, and when the limits are exceeded, the feature is considered unfavorable. (These must not be confused with the US Limits of Acceptable Change. The LAC system, one of the most widely used approaches in the USA, is a management process based on the recognition that all recreational use of wilderness causes some impact. LACs define the carrying capacity of an area by placing limits on the amount of change that can be tolerated.)

Finally, it should be noted that the performance indicators are *part of* the conservation objective, not a substitute for it. Tables 1 and 2 give an example of

Table 2 Examples of factor limits for a raised bog (the objective is given in the definition of favorable conservation status) (NRW 2013)

Performance indicators for factors affecting the feature		
Factor	Factor rationale and other comments	Operational limits
F1. Hydrology – water table	Hydrology is probably the single most important condition influencing peatland ecology, development, functions, and processes. The water table within active raised bogs normally lies within the range 0–15 cm below mean surface level and falls to 30 cm or more are rare. Drainage around and internal to the bog is a key determinant of water table residence time within the optimal range	*Upper limit*: mean bog surface level (at established central dome sample site)
		Lower limit: 90% residence within 30 cm of mean surface (at established central dome sample site) and 15 cm for at least 6 months
F2. Water chemistry	Raised bogs depend solely on atmospheric deposition to maintain waterlogging, and flooding with seawater (or groundwater) has the potential to destroy the site conservation features. Although gradations from raised bog to saline marsh are a natural feature on the margins of the bog, past management has exposed a greater part of the site to potential seawater flooding. A regional flood/drainage management policy which redresses the balance in favor of raised bog development is vital	*Upper limit*: surface water chemistry to be free of saline influence across the entire primary bog dome
	Water chemistry is critical in relation to the key species, e.g., Sphagna associated with raised mires. External agricultural operations, e.g., liming combined with drift can affect water quality on the mires	*Upper limit*: pH 4.5
		Lower limit: pH 2.7

(*continued*)

Table 2 (continued)

Performance indicators for factors affecting the feature		
Factor	Factor rationale and other comments	Operational limits
F3. Atmospheric deposition of nitrogen (N)	Raised bogs are highly sensitive to pollution from chemical nutrients, notably nitrogen (N) and phosphorus (P), which may be deposited as dust or as solutes in rainfall. Inorganic N derived from agricultural activities and fossil fuel combustion is implicated in the degradation of raised mires in Wales, causing increases in purple moor grass (*Molinia caerulea)* and birch *Betula* and a decline in *Sphagnum* cover. The bog has inorganic N loadings below the estimated upper critical threshold of 10 kg/ha/year (estimated at 8.7 kg in 2006), and this must continue for vegetation condition to be favorable	*Upper limit*: 9 kg/ha/year inorganic nitrogen
F4. Fire	Drainage of raised bogs or their margins makes them highly vulnerable to fire, particularly during drought conditions. Fire is potentially devastating for raised bogs. It destroys the vegetation cover and the microtopography, adversely affecting the carbon balance and the peat archive as well as the flora and fauna. Recovery can take several decades and sensitive species may be eliminated	Fire will be prevented as far as possible
F5. Scrub	Scrub or tree species such as birch *Betula* and willow *Salix* are not natural components of the active raised bog vegetation at this site (except in the marginal "lagg" zone), and their presence is indicative of drainage modification. Scrub/trees affect the nutrient balance, hydrology, and ecology of the bog, and without control, their impact will intensify and spread to the detriment of the natural raised bog community	*Upper limit*: extent of scrub/woodland mapped in 2003
F6. Livestock grazing	Active raised mire vegetation is a natural climax community. Grazing is unnecessary for its maintenance and would be damaging	*Upper limit*: no livestock grazing on active raised bog

performance indicators in terms of features (Table 1) and factors (Table 2) for a raised bog ecosystem.

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Abstract

Favorable conservation status (FCS) was defined for habitats and species by the EU Habitats Directive. For habitats, it includes requirements for the stability of the area of the habitat in the long term, for its quality, for typical species of the habitat to be at FCS, and for the control of factors that affect the habitat. For species, FCS requires that the size of the population must be maintained or increasing, the population must be sustainable in the long term, the range of the population must not be contracting, sufficient habitat must exist to support the population in the long term, and the factors that affect the species, or its habitat, must be under control. FCS can be used to support formulation of objectives for wetland site management but should not be used as an objective in its original form. It is important that objectives are site specific, and a commitment to maintaining biodiversity must include an obligation to ensure that local distinctiveness is maintained. An example is given of how FCS can be used to formulate site-specific management objectives.

Keywords

Wetland management planning · Wetland conservation · Adaptive management · Objectives · Features · Performance indicators · Favorable conservation status

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Definition of FCS

The definition of FCS for habitats and species used in this text is based on, and consistent with, the statutory definition of FCS for habitats and species given in Article 1 of the Habitats Directive (European Union 1992). The main aim of the Habitats Directive is to promote the maintenance of biodiversity by requiring member states to take measures to maintain or restore natural habitats and wild species listed on the Annexes to the Directive at a favorable conservation status. In applying these measures, member states are required to take account of economic, social, and cultural requirements as well as regional and local characteristics.

For a habitat feature to be considered to be at FCS, all of the following must be true:

- The area of the habitat must be stable in the long term or increasing.
- Its quality (including ecological structure and function) must be maintained.
- Any typical species must also be at FCS, as defined below.
- The factors that affect the habitat, including its typical species, must be under control.

For a species feature to be considered to be at FCS, all of the following must be true:

- The size of the population must be maintained or increasing.
- The population must be sustainable in the long term.
- The range of the population must not be contracting.
- Sufficient habitat must exist to support the population in the long term.
- The factors that affect the species, or its habitat, must be under control.

The Ramsar Convention on Wetlands describes the concept of FCS as: "A useful approach for habitats and species, which can be applied anywhere . . . These generic definitions of favourable conservation status for habitats and species are simply an expression of what would be wished of any habitat or species that requires management and could be applied to any feature on any site. Clearly, the generic statement must be developed into one with rather more meaning for particular features of the site, but this in an excellent starting point" (Ramsar Convention 2010).

A decision to use FCS at site level is made entirely for practical purposes. There is no legal requirement to do so. The practical application of the concept can provide an extremely useful, and entirely appropriate, basis for defining the desired status of habitats and species at any geographical scale, from the entire geographical range to a defined area within a site.

Relationship Between FCS and Management Objectives

Writing an objective for a feature will always be challenging, but it is much easier when it is based on the definition of favorable conservation status (Alexander 2013). FCS is an uncomplicated and common sense expression of what should be attempted to achieve for all important features. It is a generic statement that could be applied anywhere but should not, in its original raw form, be used as an objective. It is very important that objectives are site specific. A commitment to maintaining biodiversity

Table 1 An example of an objective based on favorable conservation status (FCS) for an active raised bog (NRW 2013)

Generic definition of FCS	Site-specific version of FCS
The area of the habitat must be stable in the long term or increasing	The total extent of the raised bog is at least 186 ha across the three domes (please refer to site map)
Its quality (including ecological structure and function) must be maintained	The active raised bog shows the typical features of a fully functional raised bog including central microform patterning, steep peripheral rand, and marginal lagg fen. The peat domes are waterlogged with the water table at the surface or within a few centimeters of the surface for most of the year (the water table is within 15 cm of the surface for at least 6 months of the year and within 30 cm at all times)
	The surface of the mire expanse shows the typical microform hollow/hummock patterning, with vegetation dominated by species of bog moss *Sphagnum* in lawns and hummocks. This covers at least 80% of the primary active raised bog surface
	The ratio of wet *Sphagnum* hollows to hummock and ridge microforms is extremely important. There is a mosaic of not less than 20% and no more than 30% wet hollow vegetation within a 10 m radius of almost all points on the higher bog surface
Any typical species must also be at FCS, i.e., the population must be stable or increasing	Vascular plants such as cross-leaved heath *Erica tetralix*, heather *Calluna vulgaris*, and hare's tail cotton grass *Eriophorum vaginatum* form a low-growing patchy canopy. Other species such as bog rosemary *Andromeda polifolia*, deergrass *Scirpus cespitosus*, and round-leaved sundew *Drosera rotundifolia* are less frequent but still fairly abundant
	Purple moor grass is scarce; *Cladonia* lichens and hypnaceous mosses are locally frequent on naturally drier mature hummocks
	There are wet hollows on the mire surface which contain bog mosses such as *Sphagnum pulchrum*, *S. cuspidatum*, *S. auriculatum*, and bog asphodel *Narthecium ossifragum*
	Plant communities dominated by bog mosses extend down the sloping sides of the raised mire where there are a series of transitions to other plant communities. Typically, this is into a wet

(*continued*)

Table 1 (continued)

Generic definition of FCS	Site-specific version of FCS
	heath with purple moor grass *Molinia caerulea*, cross-leaved heath *Erica tetralix*, tormentil *Potentilla erecta*, and deergrass *Scirpus cespitosus*. This in turn grades into purple moor grass *Molinia caerulea* "grassland" but still with abundant mire species
	At the bottom of the lagg fen, a poor fen and wet woodland communities are present. The poor fen is dominated by sedges such as star sedge *Carex echinata*, purple moor grass *Molinia caerulea*, and rush Juncaceae species. The ground layer has abundant bryophytes
	The *Rhynchosporion* pool vegetation forms an intimate mix with the plant communities of the active raised mire. The hollows are wet all year round except during very dry periods. The vegetation is dominated by bog mosses that favor these very wet conditions. Plants such as many-headed cotton grass, bog asphodel *Narthecium ossifragum*, and white beak-sedge *Rhynchospora alba* should form a scattered canopy over the lawns of bog moss. These hollows are frequently encountered on the tops of the raised mires. *Rhynchosporion* pool vegetation covers at least 15% and no more than 70% of the primary raised bog surface
The factors that affect the habitat, including its typical species, must be under control	Raised bogs depend solely on atmospheric deposition to maintain waterlogging, and flooding with seawater (or groundwater) has the potential to destroy the site conservation features. Although gradations from raised bog to saline marsh are a natural feature on the margins of the bog, past management has exposed a greater part of the site to potential seawater flooding. Surface water chemistry must be free of saline influence across the entire primary bog dome
	Water chemistry is critical in relation to the key species, e.g., Sphagna, associated with raised mires. External agricultural operations, e.g., liming combined with drift, can affect water quality on the mires. Natural systems with favorable status are known to have oligotrophic conditions with pH levels within the range 2.7–4.5, and the pH on this site is within these parameters
	Raised bogs are highly sensitive to pollution from chemical nutrients, notably nitrogen (N) and phosphorus (P), which may be deposited as dust or as solutes in rainfall. Inorganic nitrogen derived from agricultural activities and fossil fuel combustion is implicated in the

(*continued*)

Table 1 (continued)

Generic definition of FCS	Site-specific version of FCS
	degradation of raised mire mires in Wales, causing increases in purple moor grass (*Molinia)* and birch and a decline in *Sphagnum* cover. This site (unlike many other UK raised bogs) has inorganic N loadings below the estimated upper critical threshold of 10 kg/ha/year (estimated at 8.7 kg in 2006), and this must continue for vegetation condition to be favorable
	Fire is potentially devastating for raised bogs. It destroys the vegetation cover and the microtopography, adversely affecting the carbon balance and the peat archive as well as the flora and fauna. Fire will be prevented as far as possible
	There is no peat cutting or moss collecting
	On the mire expanse, the following species are scarce or absent: thistles *Cirsium* spp., tufted hair grass *Deschampsia cespitosa*, great hairy willow herb *Epilobium hirsutum*, reed grass *Glyceria maxima*, soft rush *Juncus effusus*, reed canary grass *Phalaris arundinacea*, reed *Phragmites australis*, bracken *Pteridium aquilinum*, and nettle *Urtica dioica. Molinia caerulea* is at most scattered at low cover
	On the mire expanse, scrub and trees (including invasive species such as *Rhododendron ponticum*) over 20 cm high are absent or very scarce
	Active raised mire vegetation is a natural climax community. Grazing is unnecessary for its maintenance and would be damaging. There should be no livestock grazing on the active raised bog
	All the other factors which are affecting, or may affect, the feature are also under control

must include an obligation to ensure that local distinctiveness is maintained. Generic objectives that can be applied everywhere have very limited value anywhere. This does not mean that a fresh start is needed each time an objective is formulated. Examples of objectives prepared earlier or elsewhere can be used to help formulate objectives for new management plans, but this must be done intelligently. An objective must be tailored to meet the particular conservation values of a feature in any specific location.

An objective can be built around the FCS definition by dealing with each section of the definition in turn. Before beginning to create a structured objective, it helps to jot down, in any order, the qualities or attributes of the feature that are clearly desirable. The current condition of the feature on the site must be considered. If any part or parts of the feature appear to be in the required condition, this provides an excellent starting point for deciding what favorable might mean. In situations where features are not in a favorable condition, the question should be: why is the feature unfavorable, and what is the difference between what is observed presently and what is desired in the future? Experience from other similar places where the feature is considered to be favorable may help, but the importance of local distinctiveness must not be forgotten. Table 1 provides an example of a management objective for a raised bog ecosystem which is based on FCS.

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Stakeholder Participation in Management Planning

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Paul Goriup

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Abstract

Stakeholder involvement is fundamental to preparing and implementing wetland management plans. Stakeholders are persons, groups, or organizations with an interest in a particular area, through direct involvement or because they have influence on management outcomes. Formal and informal communication and dialogue tools are used to identify, approach, and engage stakeholders. The first step, stakeholder analysis, is to find out who they are and what roles they play. Key or core stakeholders have a direct decision-making role such as government bodies, land owners, tenants, and community leaders. Influential stakeholders are or may be affected by management of the site and can exert pressure on the preparation and implementation of the plan (e.g., local businesses, fishermen, hunters, farmers, environmental NGOs, and research institutes). Interested stakeholders have some involvement with the site, usually for educational or recreational purposes. Stakeholder analysis also means providing stakeholders with information about the management process and their participation. Engagement

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of stakeholders usually involves an iterative process between those preparing the plan and the stakeholders. The process usually consists of formal workshops which address all aspects of the management process, or more informal meetings and other forms of communication. Addressing conflicts and conflict resolution are important parts of the stakeholder participation process. Permanent participation of stakeholders in management can take the form of a formalized consultative representative body or multistakeholder management bodies where key stakeholders take direct responsibility for site management.

Keywords
Adaptive management planning · Conflict resolution · Multi-stakeholder management · Natura 2000 network · Stakeholder analysis · Stakeholder categories · Stakeholder engagement · Stakeholder participation

Introduction

Stakeholders are those persons, groups, or organizations with an interest in a particular area, whether they are directly involved with it or because they may have some influence on management outcomes. The area concerned may have been designated and delimited under a legal act and subject to a defined form of management usually for the protection of certain features (e.g., water supply, flood control, biodiversity) or a place where activities are carried on according to traditional practices (e.g., the *hema* system in Arabia). However, the concept of stakeholders as a category impinging on site management is relatively recent: in the EU, for example, neither the Birds Directive of 1979 (European Union 2010) nor the Habitats Directive of 1992 (European Union 1992) which underpin the Natura 2000 network include any specific provisions for wider stakeholder involvement in site designation or management.

It is now generally recognized that stakeholders are fundamental to preparing and implementing any type of formal management plan for an area. Even in the simplest case, the author(s) of a management plan are themselves, by definition, stakeholders. More often, there is a wide range of stakeholders to be taken into account. Accordingly, in recent decades some basic tools have been developed to identify, approach, and engage stakeholders effectively. These include formal and informal communication and dialogue, taking an adaptive approach to management planning, and having a sound ecological basis for decision-making. The successful deployment of these tools implies a multidisciplinary approach involving social scientists, economists, lawyers, and public relations/marketing specialists as well as ecologists.

Stakeholder Analysis

The first step in working with stakeholders is to find out who they are and what role(s) they play. Depending on the level of site designation, they can include local occupants, regional authorities, national government ministries, and, for transboundary or globally important sites, international bodies (European Commission, Ramsar Convention, World Heritage Convention, UNESCO, UNEP) on the one hand, and sectoral representational organizations on the other (environmental NGOs, churches, schools, fishing and farming cooperatives, hunting associations, trade unions).

This stage frequently involves first contact so it is a very delicate operation that requires careful planning and sensitive handling in order to build trust. Some (often many) local stakeholders may not know that the site has been designated and their relationship with it will change. Others will not want to be identified for commercial reasons or because they carry out illegal activities. Yet others simply refuse to cooperate because they believe they have overriding *de jure* or *de facto* rights and powers over the site and will try to exert influence through their networks outside of the consultation process. Moreover, in many places the communities are relatively closed and stakeholders will have family ties, educational affiliations, friendships, business ties, and other shared interests. Information is rapidly passed around the community, and it is easy to start negative rumors. A badly organized stakeholder contact operation can very quickly lead to unpleasant consequences, including organized protests and adverse national media coverage.

Collecting information about stakeholders also means providing them with information in return: why they are being contacted, how the new management regime will affect them (both in terms of constraints and potential benefits such as new employment opportunities, investment, incentives, and increased property values), and how they can get involved in the management planning process. As far as possible, they should be met in person (more than once if necessary) and structured information collected using a questionnaire. It is important to bear in mind that a person may play more than one role as a stakeholder: as a government official, as a church-goer, as a member of an NGO, as a user of the site for recreation, and so on.

A summary of the stakeholder information obtained is normally stored in a table (Table 1).

Table 1 Stakeholder information

Stakeholder contact details	Interests and how affected by site management	Capacity and importance	Possible actions to address interests
Central government and dependent bodies			
Local authorities and dependent bodies			
Research institutes			
Non-government civic organizations			
Private sector			

The stakeholders are typically divided into three broad categories:

1. Key or core: those who have a direct decision-making role such as government bodies, land owners, tenants, and community leaders.
2. Influential: those who are or may be affected by management of the site and can exert (positive or negative) pressure on the preparation and implementation of the plan such as local businesses, fishermen, hunters, farmers needing irrigation water, environmental NGOs, and research institutes.
3. Interested: other stakeholders who do or wish to have some involvement with the site, usually for educational or recreational purposes.

Stakeholder Engagement

Once the main stakeholders have been identified and their roles and interests established, the next stage is to engage them in the management planning process. This usually involves an iterative process between those preparing the plan (the site management authority or consultants) and the stakeholders, especially those in the first two categories above. The process is often conducted as a series of workshops, preferably under the direction of an independent facilitator, which considers the management aspects more or less in the following sequence (the number of topics covered at each workshop will vary site by site):

- Confirmation of stakeholder interests
- Scope and general objectives of management plan explained
- Site visit, boundary confirmation, main points of interest
- Compilation of available site information, including assessment of threats
- Evaluation of current conservation status of site features
- Site vision and objectives established
- Conservation measures needed for site features
- Site monitoring plan development
- Draft final management plan presented and discussed
- Collection of comments for the final plan text

The degree to which stakeholders actually engage in the preparation of the management plan depends on a variety of factors. Those with the most direct interests, or having some statutory role, tend to contribute throughout. However, attendance at workshops can fall off quite quickly after the first one if stakeholders feel their interests are not affected, or are not being adequately addressed, or the workshop materials are not well prepared. The cost of attending workshops should also be considered when setting locations and time spans.

Other ways to engage stakeholders apart from the rather formal setting of workshops include organizing events, holding individual or sectoral meetings, setting up a website with a feedback facility, distributing printed materials, articles in local newspapers, and giving radio and TV interviews. It is very important to respond promptly to comments and to acknowledge stakeholder contributions.

Addressing Conflicts

It is to be expected that management planning for designated sites, particularly new ones, involves addressing conflicts between different stakeholders. The archetypal situation is management authority versus the local population living in and around the site, with each regarding the other as an enemy. The management authority may accuse local people of poaching wildlife, overgrazing, polluting the waters, and other illegal activities. Local people for their part claim the management authority is run by corrupt politicians, in the pocket of big business and other vested interests (especially where fishing is concerned), and has no interest in their livelihoods or social welfare which have been undermined by the site designation. Both sides may exaggerate the issues and adopt entrenched attitudes.

For this reason, many countries legislate that management plans should be prepared by independent consultants and insist on transparency and adequate stakeholder involvement in the entire process. In any case, the conflicts have to be elucidated and documented by the team preparing the management plan. Dialogue is needed to reduce tension, build a degree of trust, and convert confrontation to collaboration.

One useful technique for this is known as artisanal GIS. The management planners present large paper maps of the site to stakeholders, cover them with acetate sheets, and ask the group to draw polygons of interest to them. As the layers build up, it is possible to pinpoint the areas of overlap where conflicts are greatest and where the attention of the group can be focused. It often turns out that conflicts can be resolved by spatial or temporal separation of activities and incorporated in the site zonation system. For example, at a wetland site in Malaysia an argument over setting up a core zone that was opposed by the indigenous population was resolved when an equivalent area was established that corresponded with a region regarded as sacred for ancestors and where hunting was taboo.

More complex issues dealing with access to and use of resources can be worked through with the concerned stakeholders using interactive decision support systems, with decision trees projected on to a screen and the effects of different policies explored. Unfortunately, the best policies can turn out to be counter-intuitive, so participants have to be well prepared before undertaking the exercise.

Stakeholder Participation

The ultimate aim of identifying site stakeholders and engaging them in management plan preparation is to gain their permanent constructive participation in implementation of the plan. This usually takes the form of a more or less formalized consultative body that is representative of the main stakeholders which can communicate with the management authority according to agreed rules of procedure. Other models include multistakeholder management bodies constituted as foundations or trusts where key stakeholders take direct responsibility for site management, having regard for the objectives laid down in the management plan.

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262 Wetland Management Planning and Inclusivity: Making the Case for an Inclusive Approach to Planning

Mike Howe

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Abstract

Many important conservation sites are managed without management plans. Despite this, management planning is essential. It is the “thinking” part of the management process, and management without proper planning can result in wasted resources and wasted opportunities to be successful. Planning is also important for continuity, ensuring that agreed ideas about management are recorded for future managers to understand and to take forward. This paper advocates an inclusive participatory approach that uses the available knowledge among managers and stakeholders to the full and gives them plan ownership. The first step in the process is a joint visit to the site with as many stakeholder representatives as possible to bring together the collective knowledge and experience of the site. Further written information, including maps, surveys, and any other literature that describes the site and main features of interest, is then sent to the management team for consultation. These first two steps are crucial to achieve collective recognition and familiarity with the structure and contents of the plan and to avoid confusion about the key objectives. Effective communication from

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the beginning will also create a sense of ownership and confidence among future site managers and other stakeholders, and reduce the likelihood of conflicts. Following this inclusive approach gives a better than even chance of full implementation of the management plan.

Keywords
Adaptive management planning · Conflict resolution · Stakeholder engagement · Stakeholder participation

Introduction: The Importance of Management Planning

It is easy to understand why so many important sites for nature, archaeology, landscape, and people are managed without plans, and why some organizations with specific activities and goals often stop short of producing their own comprehensive management plans: many managers have had direct or indirect experience in the past of management plans that are produced at great cost but deliver nothing. But it is more than that. Often, protected area staff are constantly busy and stretched to capacity so that there is little time to think, and other work pressures always have a higher priority than the need to sit down quietly and plan. Monitoring often suffers a similar fate.

Despite this, management planning is essential. It is the “thinking” part of the management process, and if a manager has not thought through what he or she is doing and why, this can often result in wasted resources and wasted opportunities to really make a difference in the very important work that the manager is doing. Planning is also important for continuity, ensuring that agreed ideas about management are recorded for future managers to understand and to take forward. This improves efficiency and helps create a stronger sense of ownership and commitment among staff who, after all, want to be productive and to achieve more in the work that they do. Anyone who has a responsibility for managing sites needs to produce meaningful plans that are easily understood by anyone.

It is these key principles that the lead management planner must understand if the management plan process is to be successful. Even though he or she may have been designated to do much of the work, a participatory approach that relieves other staff of much of the burden involved in plan preparation, but uses their knowledge to the full and gives them plan ownership, is essential.

The Inclusive Management Planning Process

Joint Site Visit

The very first step is for the principle planner to visit the site in the company of some key people, effectively the management team, including the site manager(s), and advisors on nature conservation, archaeology, and recreation from the site owner organization and important partner organizations. It is important to have as many people present as possible without making the meeting too unwieldy. It is often effective to have tiered representation, where one person is effectively representing many others. This ensures that all stakeholders are represented at this crucial first meeting. In this way, the collective knowledge and experience of the site is brought together in a very useful, two-way process which, if managed correctly, can tell the principle planner everything he or she needs to know about how the management plan should be constructed.

Collection of Site Information

Having seen and discussed the main site features and issues affecting the features of the site, the principle planner will then require further written information with which to prepare a first draft of the plan, including maps, surveys, and any other literature that describes the site and main features of interest. This is then sent to the management team for consultation.

Recognition of Plan Structure and Content

At this point, if the first two steps have been carried out well, even though no-one may have seen a written management plan for this site before, there should be a collective recognition and familiarity with the structure and contents. This in turn means that excessive amounts of time are not wasted with comments, amendments, counter comments, and counter amendments. Often at this stage the management planning process can become completely bogged down and can fail because of a collective confusion or failure to understand or accept the key objectives within the plan. This is why the first two steps are so important.

Ownership by Managers and Stakeholders

If the principle planner has done the job well to this point, there is often a growing sense of excitement among the site managers, who can see how this process and this plan is going to enable them to do a better job of managing the site and that it is effectively empowering them. It is going to help them to justify and access resources (both time and money), it is going to help them to monitor the impact of their work on the features of the site, and it is going to help them to report on and communicate this information to a very wide audience in a way that would not have been possible previously.

In addition to this the other stakeholders, from partner organizations, government agencies, users of the site, and interest groups gain a sense of collective confidence in the work that the site managers are doing and a sense of ownership of the shared objectives. This makes their work easier.

A management planning process that has communicated effectively from the beginning will diminish conflicts of interest and controversy over time and will lead to a much clearer understanding, by all those with an interest, of the long-term objectives of the site. This will only be good for the key features of the site which are, after all, the most important things.

Implementation

This kind of approach to management planning more often than not results in the full implementation of the plan. This may sound like a strange statement but, as we noted at the beginning of this chapter, there are thousands of management plans around the world that remain un-used and un-read. The process and the plan can only be called successful if the plan is actually implemented by the site managers and then begins to evolve and develop as the management and monitoring informs the necessary adaptation required in an ever changing environment. Following this inclusive approach gives us a better than even chance of success.

Wetland Management Planning and Computers

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David Mitchel

Contents

Abstract

The process of writing a management plan is a complicated one that can be helped by the use of software tools like CMSi. Software management systems help in that they can provide a consistent structure for the plan, allow rapid access to other existing management plans where similar issues have already been addressed encouraging a sharing of management experience and they can integrate mapping with the management plan thus helping readers visualise where work should be undertaken. An additional bonus is that different report outputs can simply created from the same data source thus from the same "document" an simple overview management plan can be produced for the public whilst a more detailed technical plan can be produced for the site manager. Even more beneficial of a software approach to management planning is the integration of work

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recording with the plan. This transforms the plan into a dynamic working tool, recording actions and targets achieved against objectives allowing both local site reporting and corporate reporting on all the sites managed by that organisation.

Keywords
Conservation Management System · CMSi · LibraryLink · Integrated mapping · GIS · Dynamic management tool · Corporate reporting · Work recording

Introduction: Why Use Computers for Management Planning?

Computer software can support the management planning process and integrate management plans and data derived from work recording into wider corporate use. Management plans can be written using word processing software. In principle, they do not require anything else. However, the use of custom-made software can turn the plan into a dynamic working tool. This will ensure that it is implemented and thus improves the efficiency of management and ultimately help conserve the features through more effective management. The use of such software is particularly effective when it is used on an organizational basis where all levels of management engage in the process. It provides the detail required by site mangers and the overview that senior management needs to supervise and coordinate the management of sites (Alexander 2005, 2013). Management software can help address the following issues:

- Lack of consistency of approach to planning from site to site leading to difficulties in comparing and reporting on work across sites
- The planning process taking too long because the successes and failures of comparable plans are not readily available
- Management plans becoming out of date and not being revised due to lack of time
- The loss of crucial site information, especially regarding work done by staff and volunteers who subsequently leave
- Difficulties in finding information about a site, its management, and management outcomes
- Reporting taking up excessive time due to the difficulty of compiling the data
- Difficulties identifying and defending budgets without readily available data to justify them

Effective management planning software can address these issues by:

- Producing staff work plans so that all work undertaken is derived from the plan and the plan becomes a working tool.
- Using a standardized planning structure consistently on all sites managed by an organization.
- Always including the latest planning revision in the work plans, thus facilitating “adaptive management.”

- Having all the corporate plans at a planner's fingertips, making the planning process easier and preventing the planner from working in isolation.
- Allowing the planner to copy projects or complete features and their associated projects from another site.
- Allowing annual work plans to be rolled forward to the next year so that a plan never becomes out of date.
- Recording activities and outcomes, thus building a complete history of site management which is invaluable to maintain continuity.
- Facilitating reporting using readily available and accessible site data.
- Involving all members of staff and volunteers in the management plan as they are using it for their work plans and recording.
- Integrating management planning data with other areas of work like mapping; the buying, selling, and leasing of land; legal agreements; buildings and asset management; site designation and administration; and species and habitat monitoring. In this way, management plans become a central core function of the organization and are no longer seen as a "nicety."

Case Study: CMSi

History of the CMS Consortium and CMSi

Conservation Management System International (CMS*i*; www.software4conservation.com) is the most commonly used software system for site management in the Netherlands and the UK. CMS*i* is owned by the CMS Consortium (CMSC), a not-for-profit company whose members include large governmental and nongovernmental conservation organizations from the United Kingdom and the Netherlands, including Natural England (UK), Staatsbosbeheer (NL), Natural Resources Wales (UK), Natuurmonumenten (NL), De12Landschappen (NL), Wildfowl and Wetlands Trust (UK), and Welsh Wildlife Trusts (UK).

The first version of CMS was written in 1988 by James Perrins and Mike Alexander, and in 1993, the CMS Consortium (then partnership) was established to take over the running of the program. In 2004, the CMSC signed a commercial agreement with exeGesIS SDM Ltd to run the system on a day-to-day basis. In 2010, the Dutch members joined the CMSC, and a complete rewrite of the software began, leading to the current version, CMS*i*. The software is currently used by over 1500 users in over 50 organizations in eight different countries in their own language as the program includes tools to translate all labels and messages.

How the Management Plan Looks in CMSi

It can be easy for someone to become "lost" in a management plan and forget how the different elements of the plan relate to each other. In CMS*i*, the management plan is entered in a simple, visual format. This makes it much easier to view and understand

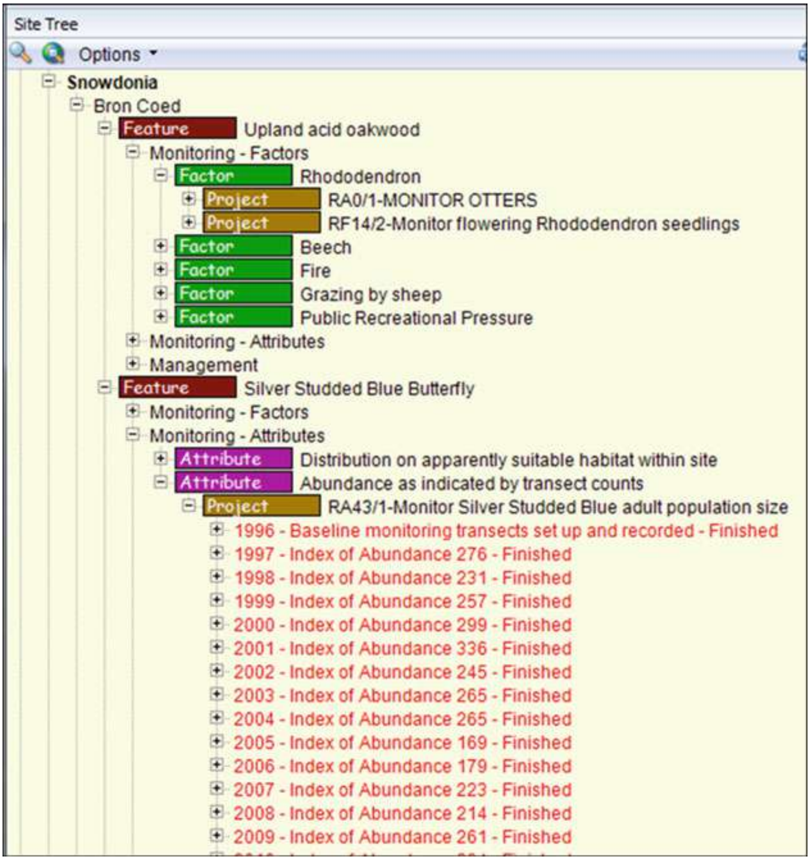


Fig. 1 A management plan in CMS*i*

the plan. The user can double click on any level (site, feature, attribute, project, or annual project) to view full information on that record. Dragging and dropping to reorganize the plan or even copy data to another site is supported. Figure 1 shows how recording is integrated with the plan with annual outcomes shown in the tree.

Integrating Mapping with the Plan

A large part of the information related to a management plan is spatial. Without maps, the plan is difficult to make and understand. CMS*i* allows users to see relevant GIS datasets alongside the maps related to management (Fig. 2). Free GIS mapping is integrated into the software, and relevant mapping objects can be opened and edited in full GIS programs like ArcGIS, MapInfo, or Quantum GIS to avail of additional functionality.

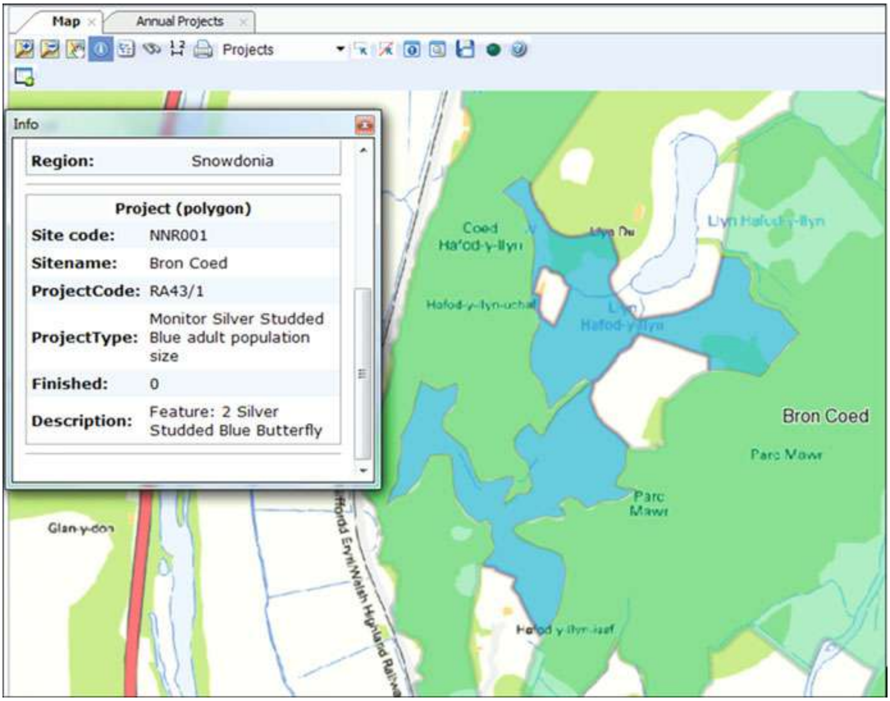


Fig. 2 Integrated mapping in CMS*i*

Integrating Images and Documents with the Plan

A management plan does not have to include every photograph taken or document written about the site as this would make the plan large, unwieldy, and difficult to read. CMS*i* comes with a module, LibraryLink, which can link different parts of the plan to other documents for detailed information such as policy documents, annual projects, and detailed monitoring results held in spreadsheets or to photographs taken as part of a photo-monitoring exercise. It can also allow linked images to be inserted into reports (Fig. 3).

Linking Planning to Recording

The integration of planning and recording, which is impossible to achieve using word processing alone, is probably the single most important benefit of using computers for management planning and transforms the plan into a dynamic management tool. There are different levels of recording, from the "day-to-day" recording of work undertaken to annual project summaries. Both are very important in the adaptive management process as they are used to assess if the management is

Fig. 3 LibraryLink

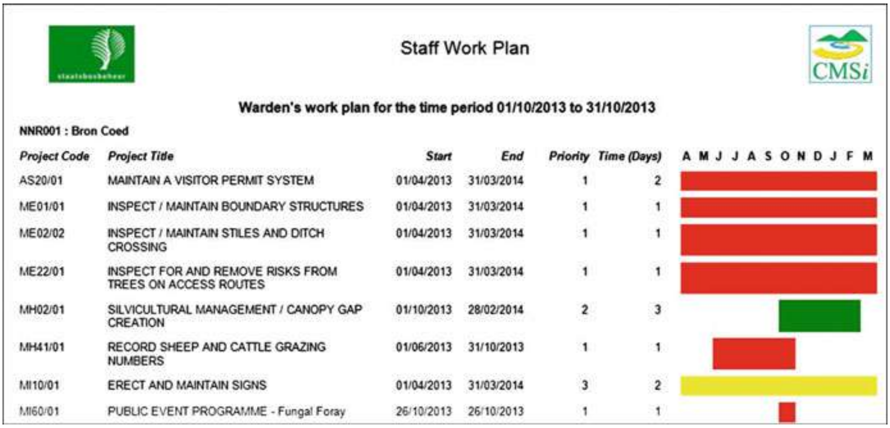
delivering the required outcomes and if planning needs to be altered. Computerized tools make this simple and quick to do.

Using CMS*i*, work records can be added quickly for multiple projects, but it is the ability to add work remotely via a website or mobile app that is revolutionizing how conservation organizations work. Through a web interface, external users can obtain access to the management plan. For example, external partner organizations or volunteers can add work records, view what work has been planned for them, or edit data and view project methodologies. Or the general public can view management objectives and work planned to be undertaken on a site.

Integrating Management Planning with Ownership and Asset Data and Species Recording

A common problem with site management planning is that it can be seen as something done by a site manager just to help themselves. Hence if it doesn't happen, it is not too important. Meanwhile, site managers need to submit their budget planning, invoice records, time recording, asset inventories, and maybe even species recording data and monitoring results all to different and sometimes competing systems. By integrating all of these into a single central system, the management plans become central to the corporate entity which leads to huge benefits in terms of efficiency and efficacy of the work.

Integration can be achieved by data transfer between different systems or by setting up one large core system. CMS*i* does both of these. It includes functionality that can transfer data out of CMS*i* to financial and contact databases. It can also store all the work undertaken by land agents and legal staff along with specialist monitoring teams in the same database, using the additional modules for property and agreements and species and habitat monitoring. This extends the use of the database to different work areas with the advantage that site managers can readily consult legal records about, e.g., property transactions and grazing rights. Land agents can



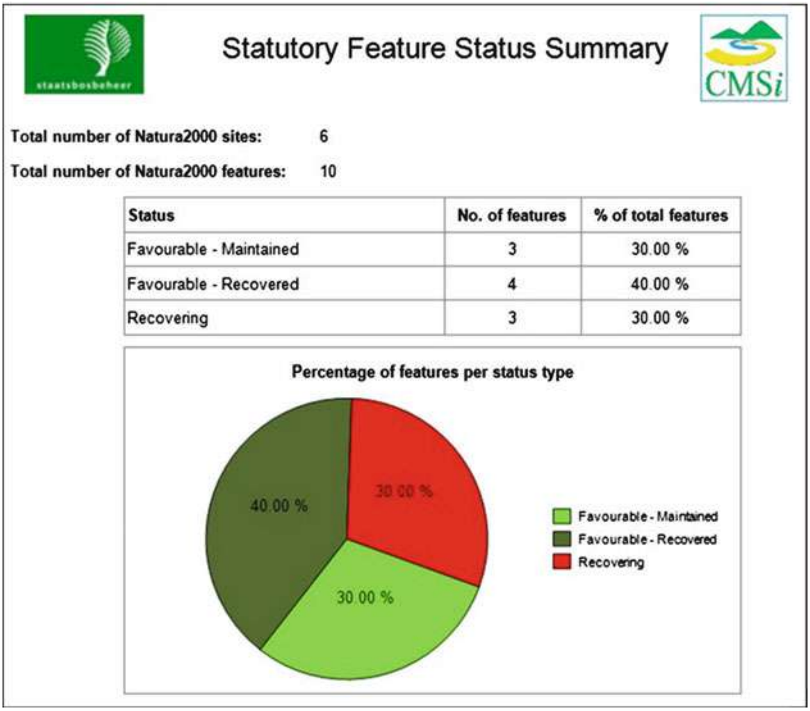
Staff Work Plan

Warden's work plan for the time period 01/10/2013 to 31/10/2013

NNR001 : Bron Coed

Project Code	Project Title	Start	End	Priority	Time (Days)	A M J J A S O N D J F M
AS20/01	MAINTAIN A VISITOR PERMIT SYSTEM	01/04/2013	31/03/2014	1	2	
ME01/01	INSPECT / MAINTAIN BOUNDARY STRUCTURES	01/04/2013	31/03/2014	1	1	
ME02/02	INSPECT / MAINTAIN STILES AND DITCH CROSSING	01/04/2013	31/03/2014	1	1	
ME22/01	INSPECT FOR AND REMOVE RISKS FROM TREES ON ACCESS ROUTES	01/04/2013	31/03/2014	1	1	
MH02/01	SILVICULTURAL MANAGEMENT / CANOPY GAP CREATION	01/10/2013	28/02/2014	2	3	
MH41/01	RECORD SHEEP AND CATTLE GRAZING NUMBERS	01/06/2013	31/10/2013	1	1	
MI10/01	ERECT AND MAINTAIN SIGNS	01/04/2013	31/03/2014	3	2	
MI60/01	PUBLIC EVENT PROGRAMME - Fungal Foray	26/10/2013	26/10/2013	1	1	

Fig. 4 Work plan report generated by CMSi



Statutory Feature Status Summary

Total number of Natura2000 sites: 6

Total number of Natura2000 features: 10

Status	No. of features	% of total features
Favourable - Maintained	3	30.00 %
Favourable - Recovered	4	40.00 %
Recovering	3	30.00 %

Fig. 5 Site status report generated by CMSi

also rapidly put their work in context by relating it to site objectives, and this integration of work areas is beneficial to the whole organization.

CMSi as a Local and Corporate Reporting Tool

The key to effectiveness and acceptance of a database like CMS*i* is that it must "work" for the people who use it. It must be functional at the local level as a simple to use tool for the site staff to plan and record their work and relieve them of some of the reporting burden (Fig. 4). For senior management, it must be able to rapidly answer questions like "How much money and time are we spending on invasive species work on all our sites?" or "On all our sites, how many statutory features are in favorable conservation status?" Both levels of questions are easy to answer using CMS*i* (Fig. 5). When resources are stretched and money short, it is important to know that every penny is being spent wisely and effectively. A management plan within a database like CMS*i* helps keep track of the financial situation easily and effectively.

Conclusion

The use of good software tools for management planning delivers significant benefits to any organization of any size. The most significant gains arise when the planning process is recognized as an essential working tool, when it is in constant use and made relevant to all levels of management. Further detailed information on CMS and illustrations of the issues covered by this article, including access to videos which demonstrate the effective use of software, can be seen at: www.software4conservation.com.

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Capacity Development for Wetland Management

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Ingrid Gevers, Esther M. J. Koopmanschap, Kenneth Irvine, C. Max Finlayson, and Anne A. van Dam

Contents

Abstract

Despite increasing awareness of the importance of wetland ecosystem services and an increase in the number of countries with policies aimed at preventing degradation and destruction of wetlands, effective protection and restoration is often

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constrained by the limited capacity of governmental and non-governmental organizations responsible for wetland management. This article defines capacity development in terms of the knowledge, skills and attitudes of people, and highlights the levels at which capacity development takes place: individual, organisational and institutional. Capacity development is presented as a long-term, integrated process of collaborative and experiential learning by all stakeholders. A number of structured steps in the capacity development process is outlined: assessment, during which gaps in capacity are identified; vision development, which describes the future goal of the capacity development effort; strategy development, which focuses on the specific interventions needed to achieve the vision; action planning and implementation; evaluation of impact, sustainability, relevance and effectiveness; and monitoring of the progress made. The article concludes with some recent examples of wetland capacity development initiatives.

Keywords
Capacity development · Wetland management · Wise use · Stakeholder participation · Experiential learning

Introduction

Increased awareness of the importance of wetlands for biodiversity and people in the 1960s led to the signing of the Ramsar Convention on Wetlands in 1972 and an increased investment in wetland conservation and wise use. From the outset, the Convention addressed the "wise use" of wetlands which later came to embody the general principles of sustainable use (Finlayson et al. 2011). In response, a wide range of methods and tools for managing wetlands have been developed, many of which are documented in the Wise Use Handbooks produced by the Convention (Ramsar Convention Secretariat 2010). The importance of wetlands for people and livelihoods was further emphasized by the Millennium Ecosystem Assessment (MEA 2005) which led to ecosystem services being included within the concept of wise use (Finlayson 2012).

Nevertheless, wetland degradation and destruction are still occurring globally (MEA 2005) with an unsubstantiated estimate that 50% of wetlands globally have been lost over the last century (Davidson 2014). At the same time, a number of reviews and analyses have shown that river/stream restoration projects are often not successful, particularly in reestablishing the processes and functions that characterize wetlands and provide benefits to people. Some countries have developed policies for the wise use of wetlands but successful implementation has proven difficult, as shown by the continuing loss and degradation of wetlands and their ecosystem services.

One of the reasons given for the continued degradation of wetlands is the lack of capacity of governmental and nongovernmental organizations and communities to effectively formulate and implement wetland policies and management plans. To improve wetland protection and management, this capacity needs to be developed or enhanced. In this contribution, capacity for wetland management is defined and ways in which capacity can be developed are described. It is argued that this concerns not only the technical capacity to strengthen wetland management (such as monitoring equipment, GIS systems, and infrastructure) but also the capacity of people and institutions to innovate and collaborate in planning and implementing wetland management.

What Is Capacity Development for Wetland Management?

In the context of capacity development (or "capacity building" as it is often called) a broad understanding of what is meant by "capacity" is presented below (Morgan 2006; UNDP 2009; Gevers and Koopmanschap 2012).

- When it comes to the capacity of people, the concept of "competency" is important. Competency consists of knowledge, skills, and attitude. Often, the emphasis in capacity development is on transferring formal/explicit *knowledge* like data, information, models, and theories. While this is important, it is often not enough, especially as knowledge is often informal or tacit, which requires other modes of transfer (Alaerts 2009). People also need *skills* to convert knowledge into action and perform tasks. Skills can be related directly to technical knowledge (e.g., how to regulate water levels in a wetland) but can also be more general (e.g., communicating with other people). Finally, a certain *attitude* is needed to be effective in performing tasks. This is related to people's beliefs about the importance of what they are doing, their motivation to perform these tasks and collaborate with others, and their determination to complete their tasks successfully. Together, knowledge, skills, and attitude define someone's competency to contribute effectively to wetland management.
- Capacity development happens at three levels. At the *individual* level, it is focused on the competencies of individuals to perform specific tasks and often consists of training and educational activities. At the level of *organizations* (such as ministries, departments, institutes, NGOs), capacity development can focus on the equipment and infrastructure ("hardware") needed for performing certain tasks, but also on the competencies of the people in that organization ("software") and the collective ability of a group of people to initiate and/or engage in a change process.
- At the level of *institutions*, capacity means the enabling environment for wetland management as determined by the policies, strategies, traditions, laws, and regulations that define and support wetland protection and management. It also includes less tangible things such as accountability, transparency, coordination, and cooperation among the people and organizations involved. Capacity development programs ideally address capacity at all three levels, taking into account that the individual, organizational, and institutional levels are closely linked to each other.
- Capacity development in all its stages is a process of learning by all stakeholders. If capacity development works well, stakeholders (including local communities, researchers, government officers, funding agencies, etc.) start acting, monitoring, adapting, and learning together while involved in managing wetlands. In doing

this, stakeholders apply both their own experience and knowledge, and the new competencies they acquire through the capacity development program. This is called "experiential learning."

- Capacity development requires a long-term, integrated effort. The combination of wetland ecosystem complexity, diversity of stakeholder groups involved, and multilevel development efforts (individuals, organizations, institutions) require a prolonged effort to achieve success. It takes time to learn new concepts and skills, to change attitudes and behaviors that have existed for years, to influence policy processes, and to achieve supportive institutions and effective collaboration.

Steps in Capacity Development for Sustainable Wetland Management

A capacity development program consists of a number of steps, as shown in the framework for capacity development presented by Gevers and Koopmanschap (2012). In general, a capacity development program is an integral part of an overall wetland management process. Importantly, the steps presented here do not always have clear boundaries and are not necessarily taken in the order presented. Often, earlier steps are revisited in the light of the results of subsequent steps, and the process can be followed in an iterative manner as a part of an adaptive management approach.

Step 1. Capacity Assessment During this step, gaps related to individual, organizational, and institutional capacity are identified and assessed. This includes reviewing earlier reports on the wetland, its river basin, and the region or country(-ies) concerned. It also includes a review of the stakeholders involved to determine the target groups. If a stakeholder analysis was not done beforehand, it will need to be done here as part of the capacity assessment phase. Another component of the assessment phase is a problem analysis, which is usually conducted as part of the overall management planning.

It is important to define the purpose of the capacity assessment (e.g., for formulating a national policy, or site management plan, or implementing a plan). The outcome of the capacity assessment should be an indication of which capacities need to be strengthened, i.e., of which stakeholder group(s), and at which level. It forms the basis for the learning objectives of the capacity development program or intervention.

Step 2. Developing a Vision A vision is a statement which describes a future state and provides an answer to the question: "where do we want to go?" Visioning for capacity development for wetland management consists of asking questions like "what should the status of the wetland be in the future?" "how will decisions be made to achieve that status?" and "who will be involved in those decisions and in achieving that status?" The time period for which a vision is developed is generally between 15 and 50 years from now.

Visioning ideally is a collective process in which all stakeholders participate. This helps to build a feeling of ownership and motivation among stakeholders; allows them to share their interests, values, and ideas; and can empower stakeholder groups that are not traditionally involved in planning processes for wetland management. The resulting vision should not be limited by the current problems and challenges faced, but rather paint a picture of the positive future that the stakeholders would like to shape collectively.

In the capacity development process, it is possible to reverse the capacity assessment and visioning phases, such that the process starts with positive and constructive thinking about the desired future state of the wetland and the people involved, followed by the assessment of what capacity is (or is not) currently available.

Step 3. Capacity Development Strategy Where the vision is formulated in very general terms, the strategy focuses more on the capacity development intervention at hand. Often this concerns a particular project or program for capacity development for a defined period of time. For each stakeholder group, an overall capacity development (learning) objective can be formulated that is based on the vision formulated in Step 2. This is often done in the form of a matrix which specifies, for each stakeholder group, the capacity gaps in terms of knowledge, skills, and attitudes; supporting and limiting factors; and the overall learning objective.

Based on the overall learning objective for each stakeholder group, operational objectives for each group should be defined according to the SMART principles – Specific – Measurable – Achievable – Realistic – Time-bound. SMART objectives are formulated in a very specific manner, defining exactly and quantitatively (often using indicators) what will be achieved when. A SMART objective could, e.g., say something like: "By July 2015, 20 local government staff of District D will be able to facilitate a stakeholder analysis exercise." The SMART objectives for capacity development are often closely linked to the objectives of the overall wetland management plan.

The final result of the strategy step is a capacity development strategy with clear overall (long-term) learning objectives for each stakeholder group and specific, operational, SMART objectives for each stakeholder group that are tightly linked to the wetland management plan. Based on these operational objectives, an action plan for capacity development can be defined.

Steps 4 and 5. Action Plan and Implementation The action plan (Step 4) translates the learning objectives for each stakeholder group into clear activities for capacity development. If necessary, the action plan can provide more detail about the learning objectives for each stakeholder/target group. For each learning objective, it details delivery mechanisms (learning activities), timing of activities, resources needed (e.g., materials and equipment), and costs. The action plan also assigns responsibilities to the personnel involved (preparation, facilitation, etc.) and links with people in other organizations that need to be made for implementation of the program. The plan should include intermediate targets or "milestones" that can be used to monitor progress during the implementation of the program.

Implementation of the action plan (Step 5) represents the actual capacity building program. This can take many forms, and a more detailed discussion of this Step is beyond the scope of the present paper.

Step 6. Evaluation Evaluation is defined as "the occasional assessment of the overall value and progress of a project" (OECD/DAC 2000). It is done to determine if the project was implemented adequately and according to plan, and also if it achieved its preset objectives. In a multistakeholder setting as often found in capacity development programs, evaluation can also address questions about the common vision underlying the program and the process of participation and collaboration.

Effective evaluation is based on evaluation criteria. Possible criteria for evaluating a capacity development effort are:

- Impact: were the learning objectives for the different stakeholder groups achieved? Were competencies improved as intended? Were the goals achieved at the different levels of intervention (individual, organization, institutions)? Did the interventions lead to a change in wetland management practice?
- Sustainability: is it likely that the impact of the intervention will persist after the capacity development project is finished?
- Relevance: are the results of the program consistent with the original priorities of stakeholders and donors?
- Effectiveness: did the program activities achieve what they aimed for?
- Efficiency: were the resources (funds, time, expertise, etc.) used efficiently in view of the results?

In each capacity development program, stakeholders can jointly identify the success criteria that they want to use to evaluate their program. Various methods for evaluation can be used, varying from discussion sessions (plenary or with separate stakeholder groups) to formal questionnaires. The results of the evaluation phase are "lessons learned" that can be used to improve existing strategies or formulate new ones.

7. Monitoring, Facilitation, and Adaptation Monitoring is the periodic reviewing of the progress made in achieving the objectives of the capacity development program in the light of the circumstances in which the program is implemented. If necessary, monitoring can lead to adaptation of objectives and activities to accommodate changes in circumstances or to deal with outcomes that are different from initial expectations. To achieve effective monitoring and adaptation, facilitation is needed to encourage stakeholders to reflect on the process and to foster a flexible, critical, and responsive atmosphere. In time, monitoring cuts across all other phases of the capacity development program.

Initiatives in Capacity Development for Wetland Management

Capacity building is a continuous process through which management of wetlands develops and the communities and other stakeholders involved can build new knowledge and understanding. This can be helped through dedicated wetland networks that enable sharing of knowledge and experiences. The widespread

Table 1 List of internet resources on capacity building for wetland management

Initiative	URL
Australian Wetland Network	www.wetlandcare.com.au
CapNet	www.cap-net.org
Kenyan Wetland Forum	www.kenyawetlandsforum.org
Organisation for Economic Cooperation and Development (OECD)	www.oecd.org/dac/governance-development/capacitydevelopment.htm;
Ramsar Convention	www.ramsar.org
South Africa Water Research Commission	www.wrc.org.za
US Society of Wetland Scientists	www.sws.org
United Nations Development Program (UNDP)	www.undp.org/content/undp/en/home/ourwork/capacitybuilding/overview.html
Wetlandforum.net (community of practice for Africa)	www.wetlandforum.net
WetlandsforLife	www.worldwetnet.org

availability of internet, even in the most remote places, provides a rich resource of knowledge. Examples include the material available on the website of the Ramsar Convention and through knowledge platforms such as provided by the South Africa Water Research Commission or capacity building and training networks such as CapNet, which also provides opportunities for establishing networks. Dedicated network sites for wetland management include regional networks such as the Australian Wetland Network and the Kenyan Wetland Forum, international initiatives such as WetlandsforLife, a global network of 500 NGOs, as well as Wetlandforum.net, an initiative to develop an internet-supported Community of Practice for Africa. Scientific support and networking is provided by organizations such as the US Society of Wetland Scientists. Linking these various communities together provides a high potential for developing a global wetland community from local to international scales. International organizations that provide support for capacity development initiatives include the UN Development Program (UNDP) and the Organisation for Economic Cooperation and Development (OECD). Table 1 lists the internet resources of these various initiatives.

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265 Wetland Management Planning, “Nieuwkoopse Plassen” (The Netherlands)

Martijn van Schie

Contents

Abstract

The “Nieuwkoopse plassen” area is situated in the western part of the Netherlands. This lowland peat bog is one of the most valuable of this type in the Netherlands but is threatened by isolation, environmental change, and nitrogen deposition. In the area 170ha of vegetations commonly referred to in the Natura 2000 network as H7140 “Transition mires an quacking bogs” occur. This article gives insight into the adaptations in management that targets bog orchid, crested wood fern, root vole and sedge warbler, and the quality of the transition mires and quaking bogs (peat moss reed-land).

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Introduction

The “Nieuwkoopse plassen” area is situated in the western part of the Netherlands, right between the three major Dutch cities Amsterdam, Rotterdam, and Utrecht. This lowland peat bog is one of the most valuable of this type in the Netherlands but is threatened by isolation, environmental change, and nitrogen deposition.

The diversity that is found in the “Nieuwkoopse plassen” is, like most other wetland areas in the Netherlands, not only the result of natural processes. Peat cutting in the seventeenth and eighteenth century transformed the natural peat bog into an open-pit mining ruin with lots of water. The impact of hard labor, mostly done by hand for two centuries, resulted in a suitable habitat for all the species in this area. The combination of reed bed mowing and natural processes transformed lots of the open water into quaking bog and peat moss reed-land. These types of vegetation are commonly referred to in the Natura 2000 network as H7140 “Transition mires and quaking bogs.” This habitat covers almost 170 ha of the reed bed area and is surrounded and inhabited by other rare habitats and species.

Some of the key species in the “Nieuwkoopse plassen” are bog orchid (*Hammarbya paludosa*), fen orchid (*Liparis loeselii*), an endemic subspecies of root vole (*Microtus oeconomus ssp. arenicola*), green hawker (*Aeshna viridis*), crested wood fern (*Dryopteris cristata*), black tern (*Chlidonias niger*), European great bittern (*Botaurus stellaris*), and sedge warbler (*Acrocephalus schoenobaenus*).

The management of the area is focused on the long-term maintenance of large areas of different suitable habitats for all these species under conditions of environmental change, specifically of nitrogen deposition. This article gives insight into the adaptations in management that targets bog orchid, crested wood fern, root vole and sedge warbler, and the quality of the transition mires and quaking bogs (peat moss reed-land). The article does not consider the management of buffered types of reed-land and grasslands.

The Ecological System

To achieve the goals and to focus the management, knowledge of the ecological system is essential. The “Nieuwkoopse plassen” is a polder system in which the surface water levels are fixed within a 4 cm margin. At the most southern part of the area, where water from the canalized river “the Oude Rijn” is let into the “Nieuwkoopse plassen,” ferric chloride is added to remove the phosphates from the river water. Because of the narrow margin for water fluctuation, water is added to the polder during almost the entire summer season. While most of the phosphates are removed from the water, sulfates and other nutrients enter the polder from this place. During rainy periods, the water inlet is also the place where nutrient-rich water is drained from the polder. Because of the length of the system – the eastern part is about 10 km from the inlet point – the water becomes more nutrient-poor towards the east. This hydrological mechanism is very important for the ecological system. Because of water movements caused by wind, this mechanism is more effective in small ditches than in large water bodies. The ecological effects of this mechanism are clearly shown on the map (Figure 1). The aquatic Natura 2000 habitats H3150 "Natural eutrophic lakes with Magnopotamion or Hydrocharition-type vegetation" and H3140 "Hard oligo-mesotrophic waters with benthic vegetation of *Chara spp.*" become more abundant in the eastern part of the "Nieuwkoopse plassen." The fen orchid only occurs in an area far from the inlet point, where the landscape is dominated by small ditches. This is where the water quality is at its best, with almost no available nutrients and very species-rich vegetation.

Another important mechanism that influences water quality is a colony of breeding cormorants (*Phalacrocorax carbo*). Their feces contribute approximately a quarter of all the yearly added phosphorus to the area. The effects of the phosphorus on water quality and vegetation development in the most eastern part of the system, where the cormorants breed, are significant (see Fig. 1). There are no submerged aquatic plants there and algae dominate the water.

The last very important landscape feature is peat holes. When peat is dug out and the soil consists of young ($\leq$300 years) peat soil, it is likely to be strongly acidic and very nutrient-poor. When the soil is older, it is richer in nutrients and lime. The contact zone between the water and the acidic, nutrient-poor reed lands is very important. When the water is mesotrophic, a very species-rich contact zone will develop containing all sorts of species that will benefit from this gradient. When the water is nutrient-rich, this contact zone will also be nutrient-rich and in a very short range two completely different vegetation types will exist next to each other. The influence of the surface water will never be more than two meters, because of the strong hydraulic resistance of the peat soil.

Values and Relations

To target the key species and habitats, knowledge of their ecological needs is necessary. For this article we will only assess the young peat soils, which have formed after the peat digging in the seventeenth and eighteenth century. These soils represent a very specific and rare environment and are home to all of the key species.

Bog orchid is a very small orchid found in mesotrophic nonacidic vegetation when the upper layer starts to acidify due to natural leaching of ions. In peat moss reed-lands, this phase is, in the Netherlands, always temporary. The natural acidification process eventually leads to complete acidification of the soil. In the "Nieuwkoopse plassen," this phase can last for many decades at the contact zone between the peat moss reed-land and mesotrophic water, which is only available far from the water inlet point. The timing of reed-bed cutting is crucial (this can be seen on the map, Fig. 1). Only where reed beds are cut between the first week of August and the last week of October, consistently for many decades, and mesotrophic water is available, bog orchid finds suitable habitat. This has a direct relation with the high rate of nitrogen deposition (70 mol N/ha/year). Purple moor-grass (*Molinia caerulea*) will profit from this, it can grow very large and outcompete all other species. By mowing, in the specified period, purple moor-grass and other potentially

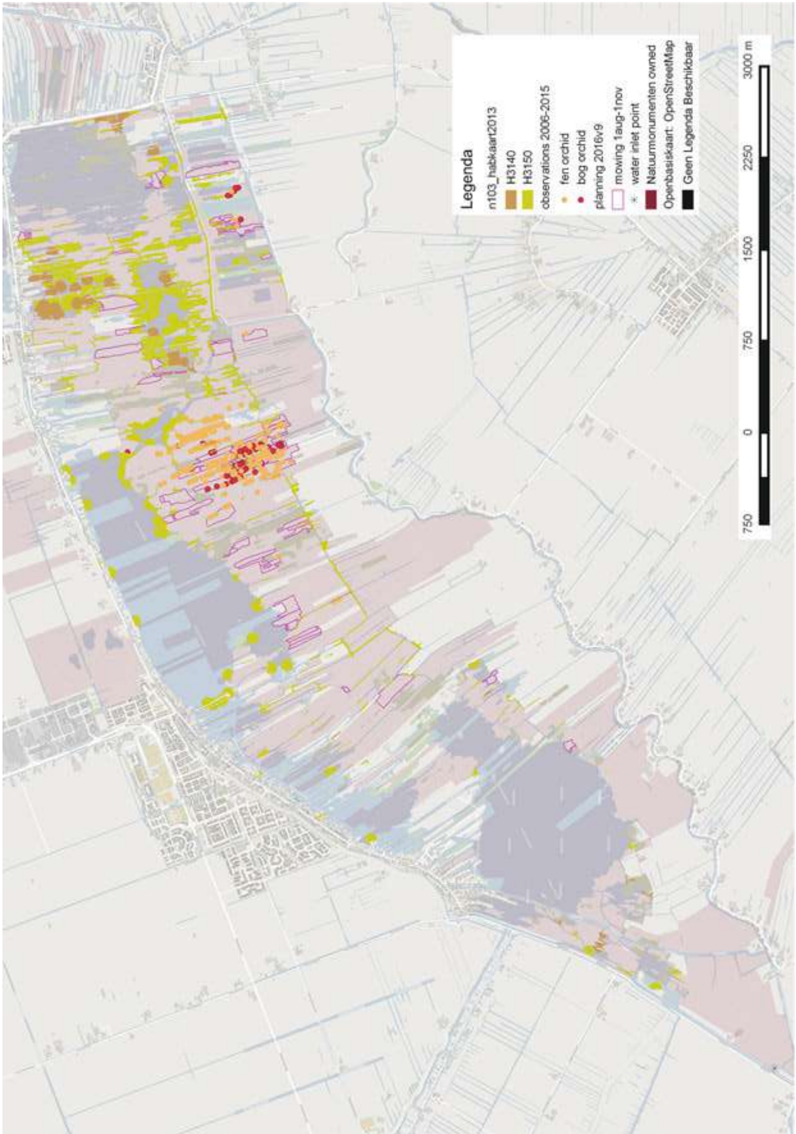

Fig. 1 Map of the Nieuwkoopse Plassen

large species will be prevented from translocating all their energy reserves into their roots for the winter period. This ensures that these plants stay smaller and will form a mosaic with small species like the bog orchid.

The crested wood fern finds its habitat in much more acidic vegetation. Because this habitat disappears in totally acidified reed-beds in the “Nieuwkoopse plassen,” this species is also temporary in this area. Although it can last much longer than bog orchid and for example round-leaved sundew (*Drosera rotundifolia*), it suffers from the same problems caused by nitrogen deposition.

Root vole (*Arenicola ssp.*) is an endemic species in the Netherlands. It finds his habitat in dynamic wet and moist conditions but is rapidly disappearing from large parts of the Netherlands. This is mostly due to drainage and disappearance of natural dynamics. In the “Nieuwkoopse plassen,” with its dynamics of yearly reed cutting and the opportunity to find shelter in uncut parts of the reed-lands, the root vole finds a sustainable habitat. The species seems to benefit from the presence of areas of unmown vegetation. This unmown vegetation also provides sedge warblers with an advantage. Although this bird is a ground breeder, vegetation which is rich in structure is essential for the maintenance of good population of this species.

The Natura 2000 habitat “Transition mires and quaking bogs” in the “Nieuwkoopse plassen” is mostly formed out of peat moss reed-land of a low quality. Quaking bogs have been replaced mostly by this type due to natural acidification in combination with sulfate (mostly in the period 1970–1990) and nitrogen deposition. Presently, it is especially nitrogen deposition that affects the quality of the reed lands because of the ability of purple moor-grass to benefit from this deposition. This species will outcompete all other species when mown in the wintertime. To prevent the disappearance of the quaking bogs, landscape management needs to aim at providing sites with enough mesotrophic, nonacidic water. At these locations, the open water will have the potential to become quaking bogs, which in time will develop into peat moss reed-lands. The quality of these reed-lands will depend on the management as described earlier.

To make sure all the species and habitats find the best possible place in the system, the right choices in management are very important. When vegetation is mown in the summer, vegetation will benefit, but root vole and sedge warbler will find a strongly degraded habitat. When the surface water is nutrient-rich, the banks of the reed lands (the contact zone between most of the acidified reed lands and the surface water) will produce large reed plants in mosaic with other common plant species and lots of insects. These banks are the best breeding habitat for the sedge warbler and other birds that depend on reed beds, but the vegetation coverage also provides perfect spots for root vole. However, these banks will always be nutrient-rich, regardless of the management. As a result, bog orchid will never grow in these reed lands because of a competitive disadvantage to other larger species.

The quality of the peat moss reed lands strongly depends on the timing of management. When mown in summer, species like round-leaved sundew, common tormentil (*Potentilla erecta*) and crested wood fern will be abundant but the root vole will disappear. When mown in winter, the root vole finds a suitable habitat but the quality of the vegetation will diminish.

Choices

There are approximately 400 ha of reed-lands in the "Nieuwkoopse plassen." Almost all of these reed beds are cut every year to prevent a rapid change in vegetation and to prevent trees from colonizing the reed-beds. To make sure that insects find a winter habitat and that there is enough vegetation coverage and structure, approximately 5% of the vegetation is not mown each year. The timing and methods of management are essential and depend on the quality of the adjacent water and of the peat moss reed land.

This leads to four management scenarios in the "Nieuwkoopse Plassen," which Natuurmonumenten is now incorporating into the work planning of the reed cutters and contractors.

Management Type 1

Good-quality peat moss reed-land with adjacent mesotrophic water quality.

The absence of dominant purple moor-grass gives other smaller species a chance to dominate the vegetation. The banks of the ditches are home to a large number of rare species like fen orchid and bog orchid. In these reed-lands, mowing takes place from August 1 to November 1 just before the reed plants and purple moor-grass translocate nutrients to their root system for the winter period. Five percent of the vegetation area remains uncut, but never in the same place as the year before and never within a distance of three meters from a ditch. This prevents unnecessary nutrient enrichment of water near the banks by deposition of dead leaf material and ensures that these vulnerable habitats remain intact.

Management Type 2

Poor-quality peat moss reed land with adjacent mesotrophic water quality.

The domination of purple moor-grass defines the vegetation, and the banks of the ditches are home to lots of rare species like fen orchid. Bog orchid is not present in these reed lands. Mowing takes place from August 1 to November 1 just before the reeds and purple moor-grass translocate their nutrients to their root system for the winter period. Five percent of the vegetation area remains uncut, but never in the same place as the year before and never within a distance of three meters from a ditch. This management will change the vegetation. The dominant large species will suffer from the mowing, and with time crested wood fern, fen orchid, bog orchid, and the overall botanic quality of the habitat will improve.

Management Type 3

Good-quality peat moss reed land with adjacent nutrient-rich water quality.

The absence of purple moor-grass gives smaller species the chance to dominate the vegetation, and the banks of the ditches are home to large flowering herbs and very strong reed plants. In these reed-lands, mowing takes place from November 1 to April 1. Five percent of the vegetation area remains uncut but never in the same place as the year before and always at least partly adjacent to a ditch. It is likely that the acidified parts of the reed lands will suffer from vegetation change due to nitrogen deposition, but the banks of the reed lands provide a perfect habitat for the root voles and sedge warblers. The management is adaptable, and when the vegetation changes unacceptably other choices can be made.

Management Type 4

Poor-quality peat moss reed-land with adjacent nutrient-rich water quality.

The dominance of purple moor-grass defines the vegetation, and the banks of the ditches are home to large flowering herbs and very strong reed plants. In these reed-lands, mowing takes place from August 1 to November 1. Five percent of the vegetation remains uncut but never in the same place as the year before and always at least partly adjacent to a ditch. This management will change the vegetation. The dominant purple moor-grass will suffer from mowing, and with time crested wood fern, round-leaved sundew and the overall quality of the habitat will benefit. The banks of the reed-lands provide a perfect habitat for the root voles and sedge warblers.

The Future of Management

Implementation of this management approach has been implemented by Natuurmonumenten since 2010 in order to make sure that all the target species and target habitats have a sustainable future. During the implementation process, existing agreements with reed cutters are taken into consideration and intensive coordination with the province of Zuid-Holland is necessary. The provincial government is responsible for the Natura 2000 targeted measures and for a large part of its funding.

The reed cutters have managed large parts of the area owned by Natuurmonumenten with a minimum of labor and other expenses. After the reeds were cut for thatching in winter, the residual material was burnt. This kind of management is cheap, but not sustainable. The reed cutting companies are equipped to do this, but only this kind of work. The change in management brings new business opportunities, but also the challenge of change for these companies. Investment in new machinery and changes in work planning are needed. This takes a few years. Some companies change quickly, others need some more time.

At the beginning of 2014, approximately 70% of all the reed beds were managed in accordance with the described management scenarios. For the remaining 30% of the area, new management still has to be implemented. Natuurmonumenten plans the completion of this transition for the end of 2018. During this time, the reed-cutting companies are given the time to reschedule their planning and targets.

Wetland Management Planning: Okavango Delta (Botswana)

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C. Max Finlayson

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Abstract

The Okavango Delta Management Plan was developed by the Botswana Department of Environmental Affairs in cooperation with a large number of stakeholders and partners. The department is responsible for the overall coordination of environmental activities in Botswana. It also coordinates environmental research, undertakes environmental education, and ensures implementation of environmental impact assessments, among many other duties. The management plan provides a framework for cooperative management of a diverse and highly important wetland complex. It addresses the role of local communities in management processes and recognises the multiple values and benefits that acrue.

Keywords

Management plan · Biodiversity · Ecosystem services · Stakeholders · Local communities

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Introduction

The Okavango Delta Management Plan was developed by the Botswana Department of Environmental Affairs (DEA 2008) in cooperation with a large number of stakeholders and partners. The DEA is responsible for the overall coordination of environmental activities in Botswana. It also coordinates environmental research, undertakes environmental education, and ensures implementation of environmental impact assessments, among many other duties.

The Management Plan was based on the planning system of the Government of Botswana and complemented by components of the Ramsar Convention's guidelines for management planning (Ramsar Convention 2010) and took into account the principles of the Convention on Biological Diversity's Ecosystem Approach (see http://www.cbd.int/ecosystem/ accessed 18 September 2016). It was a multidisciplinary planning exercise based on participatory planning, integrated management, database management, hydrological modeling, economic valuation, scenario planning, feedback mechanisms, piloting, and joint planning (DEA 2008). The details of the management plan are given below, as derived from the formal version published by the Department of Environmental Affairs (DEA 2008).

Stakeholders

Organizations involved in developing the plan included the Departments of Water Affairs, Wildlife and National Parks, Tourism, Forestry and Range Resources, Animal Health and Production, Town and Regional Planning, as well as relevant authorities in the district, namely, North West District Council, District Administration, Tribal Administration, and Tawana Land Board. In addition, nongovernmental and community-based organizations in the district were involved. The Harry Oppenheimer Okavango Research Centre (now the Okavango Research Institute) played a major role throughout the design and development of the plan. The plan was subject to external review before being finalized.

Vision and Aim

The vision for the Okavango Delta is "A carefully managed, well functioning ecosystem that equitably and sustainably provides benefits for local, national and international stakeholders."

It has an Overall Goal "to integrate resource management for the Okavango Delta that will ensure its long-term conservation and that will provide benefits for the present and future well-being of the people, through sustainable use of its natural resources."

The Overall Goal is separated into three Strategic Goals, each with several Strategic Objectives:

Strategic Goal 1. *To establish viable institutional arrangements to support integrated resource management in the Okavango Delta*

Strategic objective 1.1: *To establish viable management institutions for the sustainable management of the Okavango Delta*

Strategic objective 1.2: *To improve the planning and regulatory framework for sustainable management of the Okavango Delta*

Strategic objective 1.3: *To raise public awareness, enhance knowledge, and create a platform for information exchange and learning about the Okavango Delta*

Strategic Goal 2. *To ensure the long-term conservation of the Okavango Delta and the provision of existing ecosystem services*

Strategic objective 2.1: *To conserve the ecological character (biotic and abiotic functions) of the Okavango Delta and the interactions between them*

Strategic objective 2.2: *To maintain or restore the wetland habitats and ecosystems of the Okavango Delta*

Strategic Goal 3. *To sustainably use the natural resources of the Okavango Delta in an equitable way and support the livelihoods of all stakeholders*

Strategic objective 3.1: *To sustainably use the wetland resources of the Okavango Delta for the long-term benefit of all stakeholders*

Strategic objective 3.2: *To develop socioeconomic opportunities to improve livelihoods of the Okavango Delta stakeholders*

The strategic objectives were subjected to a SWOT analysis (Strength, Weakness, Opportunities, and Threats) to determine SMART operational objectives (Specific, Measurable, Achievable, Realistic, and Timebound). These were in turn developed into a series of action plans.

Development of the Management Plan

The development of the management plan was driven by issues and based around extensive consultation with stakeholders from local communities, government departments, the private sector, nongovernmental and community-based organizations, and policy makers. The diverse interests and expectations were recorded and, as possible, incorporated and considered during the planning processes. While the Ramsar Planning Guidelines were used to guide the development of the Plan, they were adjusted in order to accommodate local structures and processes.

The focus of the Planning was on demonstrating that integrated resource management was possible with integration being treated as a process that would be realized as the Plan was implemented and feedback obtained. A purposeful effort was made to engage with stakeholders and ensure there was wide participation – processes that will continue throughout implementation to ensure there are ample opportunities for stakeholder involvement in decision-making. Similarly, capacity building is given a high priority in order to enhance understanding and dispel misconceptions and, importantly, to create further opportunities to ensure the vision and goals are obtained.

The DEA will continue to play a major role with a presence in the main local center of Maun and by seeking opportunities for focused environmental planning and research on important issues. The latter will be extended through collaboration with the Okavango Research Institute and other institutions and experts. The importance of consultation and collaboration are emphasized throughout the Plan and are expected to be a mainstay of implementation. Similarly, data gathering, updating, storage, and processing for more informed decisions are key activities. As new information is acquired through focused research, specific action plans will be adjusted and the Plan kept adaptable and flexible. Annual sectoral reviews should ensure the Plan remains relevant.

The importance of basin-wide collaboration has been recognized given the influence that other jurisdictions have for the water that feeds the delta. In this sense, the Permanent Okavango River Basin Water Commission (OKACOM) is an important organization with mechanisms for partnership building and participation in regional projects. This level of activity will complement, that at a local level, where multiple stakeholders have a vested interest in the success of the Plan that is expected to provide benefits for the present and future well-being of the people, through the sustainable use of the bountiful natural resources of the delta.

Structure and Content of the Management Plan

The Plan was based around the implementation of an integrated resource management approach that covered three important subsystems:

1. The institutional subsystem which encompassed the management infrastructure and associated tools.
2. The biophysical subsystem which included the biotic and abiotic components of the delta ecosystem.
3. The socioeconomic subsystem covered the diverse uses (based on the ecosystem services) that people make of the biophysical subsystem.

These were addressed through the following structure:

Chapter 1 – Introduction – sets out the goal of the Plan and the policy framework within which it was developed. It also provides the background to the development of the Plan, such as the need for the plan, and the planning approach.

Chapter 2 – Site Description – describes the ecological character and the institutional and socioeconomic characteristics of the site which is listed as internationally important under the Ramsar Convention on Wetlands. It provides a collation and synthesis of existing data and information about the site and acknowledges that this needs to be regularly reviewed and updated with new data and information. A separate series of documents provides a comprehensive and detailed inventory.

Chapter 3 – Evaluation of Status and Condition – provides an evaluation of the status and condition of the ecological character, as described in Chap. 2 in order to determine the necessary management interventions and objectives. The evaluation involved the development of a set of criteria that were applied to each of the key features of the ecological character. The evaluation focused on the entirety of the ecological character, including the ecological components and processes, as well as the ecosystem services provided by the wetland to support livelihoods. This also drew on the information obtained from the stakeholder analysis.

Chapter 4 – Goals, Objectives, and Action Plans – describes the objectives for the management interventions identified during the evaluation process and confirmation of action plans. Four levels of objectives have been used, namely, the overall goal of the Plan, the long-term strategic goals, strategic objectives which were developed in order to reach the three strategic goals, and the operational objectives derived from the strategic objectives.

Chapter 5 – Management Plan Implementation Strategy – describes the strategies as well as the necessary institutional and financial arrangements required to implement the Plan. The chapter also provides a framework within which future management interventions should operate and presents the planning horizon as well as the procedures for reviewing the implementation of the plan.

Chapter 6 – Monitoring and Evaluation Plan (M&E) – describes how implementation of the plan will be monitored and evaluated against the operational objectives. It provides information about how the project is performing to help decision makers and other stakeholders by providing information to track implementation and achievements. It is a source of information for justifying changes in management strategies and budgets through an adaptive management approach.

Chapter 7 – Conclusions – describes in brief the processes and principles used in the plan and the expectations arising from its implementation.

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Wetland Management Planning: Lake Chilika (India)

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C. Max Finlayson

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Abstract

The Chilika Development Authority in collaboration with Wetlands International – South Asia developed a management framework for the conservation and wise use of Lake Chilika in Odisha State in eastern India. The management framework represents the commitment of multiple stakeholders to support the wise use of the Lake that was listed as a wetland of international importance in 1981. The development of the framework followed several decades of successful restoration and on-going management involving governmental and nongovernmental organizations and local stakeholders. In 2002, the Authority received the triennial Ramsar Wetland Conservation Award in recognition of its impressive work and outstanding achievements in restoring the Lake.

Keywords

Management plan · Ecological character · Restoration · Stakeholders

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Introduction

The Chilika Development Authority (CDA) in collaboration with Wetlands International – South Asia (WISA) developed a management framework for the conservation and wise use of Lake Chilika in Odisha State in eastern India (Kumar and Pattnaik 2012). The management framework represents the commitment of the CDA, the Governments of Odisha State and India, WISA, and the Ramsar Convention on Wetlands to support the wise use of the Lake that was listed as a wetland of international importance under the Convention in 1981. The development of the framework followed several decades of successful restoration and on-going management led by the CDA and involving governmental and nongovernmental organizations and local stakeholders. In 2002, the CDA received the triennial Ramsar Wetland Conservation Award in recognition of its impressive work and outstanding achievements in restoring the Lake.

Following acceptance that an integrated management plan was needed to address the complex ecological, social, and economic dimensions of the Lake, the CDA obtained financial support from the Small Grants Fund of the Ramsar Convention to develop the planning framework. Expert review of the framework and analysis of the status and trends of the ecological character of the Lake was carried out under a climate vulnerability assessment project funded by the International Development Research Centre (IDRC) and implemented by the CDA and WISA.

Objective and Purpose

The management planning framework outlines a pathway for the wise use of Lake Chilika through a socioecological systems approach whereby the ecological character is influenced by and influences the livelihood system associated with the lake. The long term of the management planning is the "conservation and wise use of Lake Chilika integrating catchments and coastal zones for ecological security and livelihood improvement of local communities."

The framework represents the commitment of the management authorities to support the conservation and wise use of Lake Chilika by:

- Outlining a management strategy to identify specific objectives for site management
- Describing the management actions to achieve the objectives
- Determining the factors that affect, or may affect, various features of the lake
- Defining monitoring requirements for detecting any change in the ecological character of the lake
- Supporting efforts to obtain resources for implementation
- Enabling communication within and between organizations and stakeholders
- Ensuing compliance with local, national, and international policies

The purpose of the management plan is to establish effective management practices for the lake ecosystem at river basin and coastal zone scales for ecological and economic security of people dependent on the wetland resources for their sustenance.

Development of the Management Plan

The development of the management plan was driven by ecological and livelihood issues and based around extensive consultation with stakeholders from local communities, government departments, the private sector, nongovernmental and community-based organizations, and policy makers. The diverse interests and expectations were recorded and, as possible, incorporated and considered during the planning processes. While the Ramsar Planning Guidelines (Ramsar Convention 2010) were used to guide the development of the Plan, they were adjusted in order to accommodate local structures and processes.

The planning processes took into account the interconnectedness of coastal and freshwater processes, the implications of climate change, and the need to maintain the ecological character of the lake while supporting sustainable use of the lake resources by stakeholders, in particular, local communities. An Integrated Water Resources Management (IWRM) approach was used to bring these components together with planning being undertaken at a river basin scale. The broad approach for management planning was characterized by the following:

- Adoption of a river basin approach integrating catchment and coastal processes for conservation and sustainable management
- Integration of biodiversity into regional planning
- Participatory approaches involving local communities, scientists, and nongovernmental and citizen organizations
- Adoption of preventative measures rather than merely curative measures
- Revival of indigenous knowledge and traditional practices for managing biodiversity
- Application of knowledge-based techniques for restoration
- Periodic monitoring and evaluation

The individual components of the planning approach are described in detail in the comprehensive documentation that accompanied the planning (Kumar and Pattnaik 2012).

Structure and Content of the Management Framework

The framework was organized in four sections: (i) the management planning background that describes the approach and method, (ii) description and evaluation of the ecological character and identification of key threats, (iii) a review of institutional arrangements, and (iv) the management planning framework. The planning process included the involvement of experts with specialization in catchment conservation, water resource management, biodiversity conservation, community livelihoods, and institutional development. Importantly, in terms of obtaining local acceptance and a basis for implementation, it included extensive stakeholder consultation throughout the process.

A particular strength of the framework was the extensive documentation of the information available about the ecological character of the lake, including the ecosystem services that provide many benefits for local people. Extensive efforts have been made over many years to record and analyze a vast amount of information about the ecology and use of the lake ecosystem, particularly in relation to the changes that have occurred since an artificial channel was made to enable greater exchange of water between the lake and the ocean, with subsequent large and sustained increases in productivity and fish catches.

The key management strategies to be adopted under the framework include:

- Ensuring hydrological connectivity of freshwater and coastal processes
- Establishing hierarchical and multiscalar inventory to support management planning and decision-making
- Promoting sustainable catchment management to manage the inflow of silt and nutrients
- Establishing environmental flows as basis for water allocation for conservation and development activities
- Developing ecotourism for enhancing awareness, income generation, and diversification of livelihoods
- Promoting sustainable fisheries for maintaining food security, maintenance of biodiversity, and sharing of benefits
- Reducing poverty through sustainable use of resources and diversification of livelihoods
- Promoting institutional arrangements to enable integration of wetland management planning, river basin and coastal management
- Capacity building for implementation of integrated management planning
- Communication, education, participation, and awareness to support management planning
- Results-oriented monitoring and evaluation

Action plans including performance indicators and quantitative targets where possible, and research needs and activities, have been developed for the three components of the framework – institutional development, ecosystem conservation, and sustainable resource development and livelihood improvement. The framework concludes with an estimate of the relative budgets for the action plans for each component with 5% for institutional development, 58% for ecosystems conservation, and 37% for sustainable resource development and livelihood improvement.

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Section XIX

Restoration and Creation of Wetlands

David Moreno Mateos

Wetland Restoration and Creation: An Overview

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David Moreno Mateos

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Abstract

Despite providing 40% of the global annual renewable ecosystem services generated on the planet, while only occupying ~3% of the emerged surface of the Earth, more than 50% of the world's wetland ecosystems have been heavily modified or destroyed by humans since the early twentieth century. The recognition of their importance for provisioning, regulating, cultural and supporting ecosystem services, and the highest value for restoration investment of all ecosystems has also led to many attempts at restoration, supported by global commitments for ecosystem restoration. Frequently, however, attempts at wetland restoration fail to restore ecosystem structure and functions to preimpact levels, and the research and practice to improve results are of major interest across three major categories of wetland restoration projects: (1) highly artificial systems receiving high inputs of energy and focused on optimizing the delivery of one or a few ecosystem services to society, which are usually referred to as "constructed wetlands"; (2) creating new wetlands in a location where there were no wetlands and allowing spontaneous ecological succession. These are referred to as "created wetlands" for usually multifunctionality; and (3) assisting

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wetlands to recover from impacts following with minimal intervention and maintenance, usually referred as "restored wetlands." Improving practice and understanding of wetland restoration is an ongoing need that depends not only on technical knowhow but a greater socioeconomic perspective and involvement of local communities. Such a multidisciplinary approach to wetland restoration and creation is likely to be of one of increased focus in the future and bring greater ecological and societal benefits.

Keywords
Constructed wetlands · Created wetlands · Restored wetlands · Ecosystem services · Stakeholders · Multidisciplinary

Introduction

Despite the fact that they occupy only ~3% of the emerged surface of the Earth, wetlands provide 40% of global annual renewable ecosystem services generated on the planet (Zedler and Kercher 2005) and the highest value for restoration investment of all ecosystems (de Groot et al. 2013). Wetlands provide provisioning (e.g., food, water, fiber, fuel), regulating (e.g., carbon storage, nutrient retention, flood control, erosion control), cultural (e.g., recreational, aesthetic, educational), and supporting (sediment retention, organic matter accumulation, nitrogen cycling) ecosystem services to humans. However, more than 50% of the world's wetland ecosystems have been heavily modified or destroyed by humans since the early twentieth century, especially in North America, Europe, Australia, and China (Millennium Ecosystem Assessment 2005).

Over the last century, restoration of degraded wetlands and creation of new ones have been attempted in efforts to recover physical, chemical, and biological processes and entities lost due to wetland destruction or degradation. Frequently however, this approach does not restore ecosystem structure and functions to preimpact levels (Moreno-Mateos et al. 2012). A partial estimation of restoration investments showed that only in North America (including Canada, United States, and Mexico), over $70 billion (US) have been spent attempting to restore more than 3,000,000 ha of wetlands in the last 20 years (Moreno-Mateos et al. 2012). At a global scale, ecosystem restoration is now becoming a global priority, and the UN Convention on Biological Diversity (CBD) in December 2012, called upon its parties to commit to restore – or at least to begin to restore – no less than 15% of every ecosystem type within their territories by 2020, including at least 10% of the oceans. Notably, the IUCN, the UN Convention for Combating Desertification, the UN Framework Convention on Climate Change, the European Commission, and many other major agencies and governments have also endorsed or adopted this ambitious goal. Concurrently, the UN Environment Program has called restoration one of the most profitable public investments for sustainable economic growth and overcoming poverty. In 2011, IUCN launched the "Bonn Challenge" to restore 150 million ha of degraded lands and forest by 2020.

Offset policy schemes are also increasing the demand for ecosystem restoration and creation, with particular emphasis on wetlands. Offset policies require both ecosystem restoration and creation to compensate for damage and loss of ecosystems caused by human activities (e.g., agricultural transformations, development). The first offset policies started in the 1970s in the USA under the name of "no-net-loss" policies and referred specifically to wetlands (i.e., Clean Water Act in 1977). However, no-net-loss of wetlands has never been achieved in the more than 30 years since the regulation was established. Today, offset regulations are required in 45 countries and are being developed in another 27. The need for successful wetland restoration and creation keeps growing (but see Box 2).

Definitions and Approaches

The practice of both wetland restoration and creation began more than 40 years ago. Since then, three main approaches have emerged. The first focuses on highly artificial systems receiving high inputs of energy and focused on optimizing the delivery of one or a few ecosystem services to society. These intensively managed systems are usually referred to as "constructed wetlands". A second approach has focused on creating wetlands *de novo* in a location where there were no wetlands before (primarily within the context of offset policies), but allowing spontaneous ecological succession, and involving low or no maintenance. These wetlands are referred to as "created wetlands" and usually are designed to achieve multifunctionality rather than focusing on specific ecosystem services. The third approach has focused on assisting wetlands to recover from past anthropogenic impacts following spontaneous ecological succession with minimal intervention and maintenance (Society for Ecological Restoration 2004). This third type of wetlands is usually referred as "restored wetlands".

Most constructed wetlands today are engineered to remove nutrients from both agricultural and urban wastewater or organic compounds and heavy metals as efficiently as possible (Moreno et al. 2007) (Comin, this volume). By manipulating substrates, flow speed, oxygenation, plant species, and microbial communities under highly controlled conditions, constructed wetlands can yield high long-term removal rates of nitrogen, phosphorus, and organic compounds (Mander, this volume). It has been estimated that between 1.5% and 4% of semiarid Mediterranean agricultural catchments must be dedicated to wetland creation or restoration to remove nitrates coming from agricultural fertilization (Moreno-Mateos et al. 2010), and 1.2% to degrade half of the pesticides present in agricultural run-off. However, managing constructed wetlands to obtain high phosphorus retention rates usually requires important maintenance efforts, involving periodical substrate or sediment replacement or removal (Langman and Craft, this volume). Another major purpose of constructed wetlands is to provide flood control which, when adequately managed, may also provide other services like fisheries, wildlife habitat, and reclaimed land for agriculture.

Multifunctional engineered wetlands providing various services at various scales are starting to be commonly conceived, either by adapting wetlands with other purposes, or by specifically engineering multifunctional wetlands. These services include water quality improvement, soil desalinization, and biodiversity enhancement (Moreno-Mateos and Comin 2010) (Comin, this volume, Mander, this volume). This is one important aim of wetlands created under mitigation schemes. In this context, many wetlands are created in places that never were wetlands before to compensate for loss of existing wetlands. Once the creation project is finished, spontaneous succession usually leads the trajectory towards the biodiversity and functionality of created ecosystems. These kinds of manipulations carried out in created wetlands are similar to those used in restored wetlands, and thus, the outcomes are commonly comparable (Moreno-Mateos et al. 2012). In the case of restored wetlands, the main aim is to restore their historical continuity, that is, their ability to recover from damage following spontaneous succession once all degrading factors have been removed. In this case, the presence of nonnative invasive species, for example, could be considered a degrading factor preventing restored wetlands to follow their historical continuity.

Managers, designers, and practitioners of wetland restoration and creation should also be aware that service-based wetland restoration or creation is based on current market demands that might change in the future. More importantly, ecosystems have functions that might become services in future markets. Their loss today, as a consequence of a narrow focus on particular services now in demand, might incur increased restoration and creation costs in the future. For this reason, restoring and creating multifunctional wetland ecosystems is insurance in the context of fluctuating and partly unpredictable markets.

Current Limitations

The practice of wetland restoration and creation is still evolving and its ability to recover biodiversity and ecosystem functions has been proved to be currently limited. Restored and created wetlands of all types (including depressional, tidal, and riverine wetlands) from all over the world frequently are ~25% less biodiverse and functional than reference systems in the predisturbance state during decades or centuries after they are restored or created (Box 1; Moreno-Mateos et al. 2012). But wetland recovery rates were affected by the size and the environmental setting. For example, large wetlands (especially those over 100 ha) recovered better than small ones. Also, wetlands restored or created in temperate climates recovered faster than those in cold climates. This could be explained by the fact that ecological processes may need decades or centuries to recover. For example, although the ability of wetland soils to store carbon may recover after a few years, it might take many decades to restore the complete carbon and nitrogen cycling processes and the amount of carbon stored in soils (Craft et al. 2003) (Langman and Craft, this volume). Ballantine et al. (2009) found that after 55 years only 50% of the soil organic matter existing in reference wetlands in New York State had recovered.

During the long period in which wetlands are recovering their biodiversity and functions, their positive effects on other ecosystems and their provision of ecosystem services are diminished.

Box 1 Reference Wetlands

Reference wetlands orient practitioners to set their goals. They are a tool to inform the process of decision making in wetland restoration and creation. Restoration objectives should not be rigidly based on reference wetlands but need to be flexible and adapted to currently changing conditions. References could come from adjacent "undisturbed" wetlands, from wetlands located far from the area to be restored or created but with similar biophysical settings, or from old – written or oral – descriptions of areas that have been degraded with real examples no longer existing. Occasionally, no wetlands similar to the wetland we want to restore exist anymore. In this case, restorationists should look for the most similar wetland existing to obtain some guidance.

The present level of global alterations is such that wetlands used as reference systems may not be the same as before degradation. Nitrogen deposition, climate change, or invasive species are widespread human impacts that cannot be reversed at the local scales at which restoration projects normally take place. Restoration practitioners must be aware that the biophysical conditions in reference wetlands are fundamentally different from the conditions existing in areas to be restored or created. For this reason, the assembling communities of areas being restored or created should not be expected to be similar to communities found in "undisturbed" systems (Galatowitsch, this volume). For example, some restored parts of the Florida Everglades have high amounts of phosphorus in their soils as a result of an agricultural legacy. Species in these newly restored areas (mostly *Typha* sp.) must be tolerant to high phosphorus concentrations and are not found in less disturbed areas with low phosphorus contents in their soils.

Another relevant question is if and how the practical tools used in wetland restoration and creation are helping to increase wetland biodiversity and functionality. Studies show that ecosystem response to varying intervention approaches, under similar habitat and environmental conditions, may be markedly different. Multiple examples show the need for direct biological manipulations or interventions (e.g., revegetation), in efforts aiming to restore or create wetland ecosystems (e.g., in salt marshes and mangroves; Galatowitsch, this volume). In other cases, spontaneous recovery happened without interventions in coastal wetlands. In other wetlands, results of biological manipulations were inconclusive.

A global study of restored and created wetlands found that physical interventions (i.e., hydrological restoration) by themselves facilitated, on average, recovery of the biodiversity and biogeochemical functionality within the first 10 years (Moreno Mateos, unpublished data). However, the study also found that biological interventions aiming to accelerate the spontaneous succession process (involving organisms and functionality) might in fact delay it by about 20 years. Differences in the recovery rate of plant and animal assemblages between revegetated and non-revegetated wetlands were largest, and extended over the longest periods of time. Overall, the use of revegetation did not provide extra benefits or added value compared to physical interventions used alone. In many cases environmental factors such as invasive species or extreme climatic events forced recovery trajectories to converge over short (~10 years) periods of time, regardless of the restoration or creation approach used (Collinge and Ray 2009). A combination of ecological and anthropogenic factors appears to limit or delay the successful recovery of restored and created ecosystems. Even if full ecosystem restoration and creation were obtainable today, given the widespread need to buffer biodiversity and ecosystem functionality losses, we must accelerate the assisted recovery processes, which currently take many decades or centuries.

Policy and Governance

There is widespread consensus about the urgent need to convert research results in conservation and restoration into real action to reduce species and habitat loss and ecosystem degradation, and contribute to global initiatives to achieve “no net loss”. Ecosystem restoration projects, including wetland restoration projects, habitually lack a socioeconomic perspective, which may have negative effects on their success (Box 2; Aronson et al. 2010). Specifically, limited involvement of local communities is known to reduce restoration success (Suding 2011) (Barbier, this volume). On the other hand, the socioeconomic value of restoration is high. The benefits of restoration normally outweigh its cost (Bullock et al. 2011) and are in fact high-yielding investments considering the amount and range of ecosystem services that restored ecosystems provide to societies (de Groot et al. 2013). Of particular interest are coastal and inland wetlands that return the highest value for restoration investment in absolute terms compared to any other ecosystem type (de Groot et al. 2013). Based on these premises, international and global regulations and initiatives mandate the use of wetland restoration and creation. However, they do not necessarily take cognizance of the current limitations to our ability to restore or mimic natural ecosystems which, over the long term, may increase an accumulated loss of biodiversity and ecosystem functionality (Fig. 1).

In this context, expanding compensation or no-net-loss policies that do not internalize unachieved outcomes might be promoting long-term loss of biodiversity and ecosystem services of wetlands (Moreno-Mateos et al. 2012). For example, Gutrich et al. (2004) estimated that the cost lag of mitigation wetlands in Ohio and Colorado was on average 50% of the total restoration investments over a period of 13–33 years. This suggests that society is currently incurring significant wetland restoration costs due to recovery time lags of mitigation sites. No-net-loss policies strive to balance biodiversity loss with gains in alternative ecosystems, but offsets are rarely adequate for achieving no net loss of biodiversity alone, and some development effects may be too difficult or risky, or even impossible, to offset. Over time, higher investment might be necessary to recover that fraction of biodiversity and functionality lost. Estimation of socioeconomic losses, usually measured in terms of ecosystem services losses, of unsuccessful ecosystem restoration and creation policies will be essential to improve current wetland management policies (Barbier, this volume).

Box 2 Wetland Restoration Performance

Performance, many times also called "success", in wetland restoration and creation measures the similarity between restored or created wetlands and reference systems (Box 1), or how close they are to our goals of providing specific ecosystem services. However, how to measure performance is highly debated and controversial. In most cases, restored and created wetlands recover or develop some of the desired features but not others. Also, a reference might not be achievable anymore due to major environmental change that prevents reversing ecosystem temporal trajectories. For these reasons, the performance of wetland restoration and creation can be measured in terms of overall performance, regardless of references; and in terms of historical continuity, i.e., to what extent do wetlands follow their own recovery trajectories when all possible degrading factors are removed? In created wetlands with particular objectives (e.g., water quality improvement), performance can be more easily measured (e.g., how much nitrogen and phosphorus is the wetland removing? Comin, this volume). Performance can also be viewed differently by different wetland restoration stakeholders. For example, hunters will define as successful a restoration project which brings ducks back to the wetland. In the same sense, success for some farmers is when wetlands remove some of the nutrients that contaminate their rivers. However, scientist could argue that a wetland that does not have a similar diversity and functionality as an undisturbed wetland has not been "successfully" restored. As pointed out in the main text of this chapter, full restoration in this sense may take centuries, millennia, or even never be realized. Within this logical framework, most wetland restoration projects and many creation projects will never be recovered or fully developed within a few human generations (Fig. 2).

Future Challenges

To accelerate the wetland recovery process, the traditional perspective of ecosystem restoration and creation focused on self-organization could also include other approaches. An example is engineering of the community assembly processes, as suggested by food-web and community ecologists. Evidence supporting this approach suggests that when phylogenetically distant species are used in restoration projects, the survival and density of neighboring species increases (Verdú

Fig. 1 Coastal wetlands restoration in the eastern Coast of USA. Revegating with *Spartina* sp. is one of the most common practices in coastal wetland restoration (Photo credit: Ecosystem Restoration and Manangement, Inc.; used with permission)

et al. 2012). This may happen because functional traits of unrelated taxa increase niche overlap and, thus, raise the overall efficiency of the community in the use of energy (i.e., resources). Other approaches focused on restoring ecological networks (e.g., food-webs, mycorrhizal networks, spatial fluxes of resources) at the ecosystem and the landscape scale should also be developed. In parallel, approaches to recover the historical continuity, through restoring the evolutionary processes that shaped current communities and species, and that can act quickly (<100 years is quick at an ecological scale) must be also restored.

A better understanding of how the relationships between biodiversity, functions, and services evolve during the recovery process could also help to find tools to accelerate the wetland recovery process. In all types of ecosystems, the effects of biodiversity on ecosystem functionality are responsible for major changes when ecosystems are altered because of the strong, positive relationships between biodiversity and ecosystem functions (Doherty and Zedler, this volume). However, in some cases, negative relationships between biodiversity and functionality can appear (Doherty and Zedler, this volume). In other cases, ecosystem functionality can be recovered even when species cannot. This could be the case in wetlands where recovering the predisturbance biodiversity is unachievable within the context of the restoration or creation projects and due to ecological, technical, or economic factors. However, functional restoration must be always considered as a last option, because it may involve both unexpected outcomes from newly forming communities, and

Fig. 2 Before and after a large scale wetland restoration project (>800 ha) took place in Rancho Humo (Guanacaste, Costa Rica). Agricultural land was restored to enhance biodiversity and develop the local industry of ecotourism (Photo credit: Rancho Humo Private Reserve; used with permission)

long-term loss of some species which recovery may be ever harder in the future. Finally, another way to understand limitations caused by anthropogenic factors will be to quantify the effect of the historical legacies, that is, the response of wetlands to the intensity and duration of the degrading phase (Galatowitsch, this volume). Jones et al. (2009) found that agricultural transformations and the impact of multiple stressors on ecosystems (both aquatic and terrestrial) had the largest effects on the recovery time of restored ecosystems. The duration and intensity of the degrading activity could force regime shifts into alternative states that prevent recovery (Box 3). For example, restored and created wetlands that become eutrophicated and exhibit persistently low biodiversity and functionality.

Box 3 Alternative States of Restored Wetlands

Once wetlands are damaged, their recovery to predisturbance states might not be feasible in all cases. In this circumstances, restored wetlands might enter into alternative states during varying periods of time or permanently. Compared to the original state of the wetland, alternative states are characterized by the presence of different communities of plants, animals, and microorganisms

(*continued*)

Box 3 Alternative States of Restored Wetlands (continued)

and by different functions and services. They are stable over time, and reversing the alternative state to the predisturbance state involves large amounts of energy or will not be possible at all.

Alternative states can be less functional and diverse than predisturbance states. This is the case for example in eutrophized wetlands, where high loads of nitrogen only allow for the establishment of reduced numbers of aquatic macroinvertebrates and plants as compared to "undisturbed" wetlands. In other cases, alternatives states might be highly functional and the main difference is the species composition or relative abundance of the communities present in the wetland.

Traditionally, alternative states have been considered as stable. However, recent evidence suggests that alternative states are essentially transient states over long time periods that lead periodically to other alternative transient states. The duration of alternative states is frequently debated, since many ecosystem functions may take centuries to recover, e.g., carbon storage in soils. In other cases, there is no actual change, e.g., the cycling of phosphorus in soils. Phosphorus has a conservative cycle without gaseous phase that barely affects the total amount and the relative abundance of the different fractions of phosphorus in soils (Langman and Craft, this volume). Wetlands usually take centuries or millennia to reach their present conditions. For this reason, severe damage will take centuries or millennia to be repaired. During that time wetlands will, in many cases, be less diverse and functional.

Forces that lead wetlands into alternative states can be environmental, ecological, or anthropogenic. For example, wetlands whose species are at their range limits may be less resilient to impacts. Once some species are removed, their return is highly unlikely. Several ecological factors can bring wetlands to alternative states. For example, priority effects (the effects of the species that arrives first to the restored wetland on the subsequently arriving species) during the plant and animal assembly history of restored wetlands might change the fate of those wetlands. The species that arrives first will change the environmental conditions (in the case of plants, through its litter, its ability to use nitrogen from the soil, or it interaction with herbivores) for the other species that will arrive later, thus affecting the resulting community. An example of anthropogenic factor leading to alternative states is the high nitrogen load explained above.

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Restoring and Creating Wetlands for Water Quality Improvement in Agricultural Territories

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Contents

Abstract

Wetlands play a key role in improving the quality of the water that flows through them. As such, loss or degradation of wetland area can have severe consequence on water quality of receiving rivers, lakes, and coastal habitats. This often has catastrophic and expensive costs for society and provides many lessons to provide for better integration and management of wetlands in, particularly, agricultural landscapes. Using wetlands for water quality improvement is most successful when based on restoration of the hydro-geomorphologic characteristics of existing wetlands and when avoiding exceeding their loading capacity. This includes restoring riparian zones and river morphology in accordance with their spatial (longitudinal, transversal, and vertical) and temporal dynamics. Implementing restoration projects at a sufficiently large scale to mitigate nutrient

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impacts is major future challenge for wetland integration in agricultural landscapes, as this also necessitates resolving stakeholder socioeconomic and often cultural concerns that can consider restoration of wetlands as contrary to agricultural development. These challenges can be overcome by using experience of how restored and created wetlands can be integrated into both small-scale and large-scale socioeconomic development plans.

Keywords
Water quality improvement · Agricultural uses · Restoration · Wetlands creation

Introduction

Despite the large number of wetlands that have been restored and constructed all around the world during the last two decades (Mitsch and Gosselink 2007; see also the journal Aquatic Sciences 75(1), January 2013), the present wetland area is still much smaller than their original extent. There is also a difference between the present benefits people obtain from wetlands compared to their potential provision of ecosystem services. This has occurred in great part because the water quality of natural aquatic ecosystems is still below acceptable standards in many parts of the world (Russi et al. 2013), particularly in areas with intensive agricultural development where the strong links between ecological processes and biodiversity were not considered. In many cases, desiccation associated with subsidized agricultural development caused wetlands to collapse (e.g., the Eden marshes, Richardson and Hussain 2006; the Aral Sea, Micklin 1988). Lessons learned from these and similar experiences around the world should prevent plans for large land transformations that do not take into account the integration of natural systems, particularly wetlands, in agricultural landscapes. However, rural land use and socio-economic developments based on this short-sighted approach with catastrophic and expensive costs for society still continue (e.g., in China; Luan and Zhou 2013), despite the fact that numerous examples exist of how restored and created wetlands can be integrated into both small-scale and large scale socio-economic development plans (Brandeis 2010).

Wetlands play a key role in improving the quality of the water that flows through them, and for this reason they have been called the 'kidneys of nature' (Mitsch et al. 1998). This is particularly important in areas with agriculture, livestock and fish farming. In these areas fertilizers, pesticides, and pharmaceuticals are used to increase and ensure food production. Restoring and creating wetlands in these areas can play a key role in improving the water quality and providing other ecosystem services (Finlayson 1993). Wetlands provide multiple ecosystem services, even if managed or constructed to provide specific services (Moreno-Mateos and Comin 2010). However, while a natural wetland is the result of centuries or millennia of hydro-bio-geomorphological evolution and provides a suite of combined services in accordance with its ecological functioning, a restored wetland may take 30–100 years to resemble natural wetlands (Moreno-Mateos et al. 2012). In contrast, a wetland that is restored or created to optimize one specific function will need a short time to provide the services associated to that function. Developing additional functions might lower its performance (Van der Valk and Jolly 1992).

Alternative plans can be performed to restore or create wetlands at a regional scale in accordance with the objectives to be achieved. If the major objective is buffering impacts of pollutant discharges and improving water quality discharged into the natural environment, wetlands should be created purposely for this objective. A combination of undisturbed and restored wetlands, and also wetlands intentionally constructed for wastewater treatment, will provide more services than just restoring and creating wetlands with a single objective (Russi et al. 2013).

Priorities for Wetland Management, Restoration, and Creation for Water Quality Improvement at Landscape Scale

Priority Establishment

The conservation of existing natural wetlands is always preferred because it provides more ecosystem services, at a lower cost, than restoring or creating wetlands. Remains of wetland biota and structural components (e.g., soil) help the restoration process by contributing autochthonous components and facilitating the recovery of their ecological processes. Wetland creation should be complementary to restoration, rather than used as mitigation or compensation for past, present or future wetland destruction. The diversion of a river mouth and elimination of marshlands in the delta of River Llobregat south of Barcelona (NE Spain), as proposed for the expansion of Barcelona harbor and airport, respectively, is a paradigmatic case of wetland creation against wetland restoration favoring short term economic interests (Marshall 1994). Restoring wetlands as part of adaptive planning for anthropogenic global environmental change is frequently proposed as an effective method to prevent natural disasters and steering long-term socio-ecological development, including the improvement of water quality at basin scale (Mitsch and Day 2006). This approach requires long-term and large-scale integration of economic, social and environmental aspects of development (Comín et al. 2005).

Priority Definition

Based on the principles mentioned above, using wetlands for water quality improvement in a certain area should be based on restoration of the hydro-geomorphologic characteristics of existing wetlands and should avoid surpassing their loading capacity. This includes restoring riparian zones and river morphology in accordance with their spatial (longitudinal, transversal, and vertical) and temporal dynamics (Stutter et al. 2012).

Creating wetlands upstream of, and in parallel with natural wetlands contributes to preventing overload of existing wetlands with pollutants. Treatment wetlands can be constructed for intensive point source pollution from small human populations and farms (Kadlec and Wallace 2008; Vymazal 2010). These treatment wetlands can be designed in a compact form to remove specific pollutants or with shapes adapted to land geomorphology (Harrington et al. 2011). Reducing and controlling point source wastewater is relatively easy with treatment wetlands designed for specific pollutants.

However, using wetlands to reduce non-point source pollution requires planning the spatial distribution of wetlands in accordance with their efficiency to improve the water quality of natural systems. Decisions must be made about where wetlands must be restored or created for buffering pollutant discharge. A number of modeling tools (e.g., Soil and Water Assessment Tool: SWAT) can be used to obtain information about the amounts of pollutants that are discharged from every sub-watersheds or part of the area. Then, different wetlands can be designed to intercept and treat as much of the pollutant discharge flows as possible. However, economic and social aspects are also involved in the decision-making about restoring and creating a single large wetland to treat a large discharge collected from a relatively large territory, or several small wetlands distributed among different discharge points. In contrast to the SLOSS (Single Large One or Several Small) debate in ecology for establishing biodiversity conservation reserves in fragmented habitats (Wilcox and Murphy 1985), the solution for restoring and creating wetlands to improve the water quality discharged from non-point sources in agricultural areas is to restore and create those wetlands which treat most of the pollutant discharge. This may be one large wetland in a place which accumulates most of the pollutant discharge of a certain area. Alternatively, where pollutant discharges are highly dispersed, multiple small wetlands can be restored or created distributed in different sub-watersheds.

Dimensioning and Designing Wetlands for Water Quality Improvement at Landscape Scale

One of the most frequent pollutants in agricultural areas is nitrate, which is commonly used as fertilizer for stimulating plant growth and crop production. Constructed wetlands are very efficient systems for nitrate removal from non-point source agricultural wastewater, achieving 50–99% removal of the nitrate discharged into the wetlands in 1–5 years. This is due to the denitrification activity of microbial communities at the aerobic-anaerobic interphase of wetland sediments, assimilation by the biological community and other biogeochemical processes (Mitsch and Gosselink 2007).

Simple empirical models of removal rates in treatment wetlands, which relate wetland area and other physical characteristics to the decrease of nitrate concentration, are useful tools for dimensioning wetlands for nitrate removal from non-point source pollution in agricultural areas. However, site-specific parameters and seasonal temperature variability and soil and plant characteristics must be considered as well in more sophisticated dimensioning tools, particularly if not much previous information exists on the pollutant of interest (Kadlec and Wallace 2008).

One further step is designing the wetland characteristics for water quality improvement of non-point source pollution in agricultural areas. The most realistic approach is to define the morphological characteristics and structural (including biological) components of the wetlands in accordance with the social and economic possibilities of each project. Social aspects include land availability and official regulations, which may differ by country. If enough land is available, preserving some areas next to the main restored or created wetland will be useful for future land use changes and maintaining operations. Economic restrictions may limit the extent of the wetland restoration/creation project. Thus, a strategy combining all these aspects must be adopted for every restoration project, at the adequate spatial scale (Comin et al. 2014). The watershed is the most appropriate scale for planning the recovery of functions and services provided by natural and constructed wetlands, because it is the geographic area where most intensive hydrological and biogeochemical relationships between parts of a territory take place. (Dunne et al. 2005).

Future Challenges

Implementation of projects at a large scale is the challenge ahead for wetland restoration and creation, both to recover the wetlands lost during the last century and, in particular, to improve the water quality in agricultural areas. The European Commission has sponsored the restoration of wetlands for 20 years through its Life Program (http://ec.europa.eu/environment/life); the US Environmental Protection Agency provides general guidelines for large-scale wetland restoration that was implemented at watershed and state scales (http://water.epa.gov/aboutow/owow/org.cfm); and the Australian Government develops several wetland restoration projects at watershed scale including the large Murray-Daling river basin (http://www.mdba.gov.au/what-we-do/environmental-water/restoring-rivers-wetlands). A coordinated global restoration strategy is still missing for those purposes. In Asia, Africa, and South America, where huge wetland areas have been destroyed or degraded and plans for extending agricultural cultivation will have a negative impact on wetlands, large-scale restoration is very limited, despite efforts by the Ramsar Convention of Wetlands to incorporate more and more wetlands into conservation regulations all around the world. A more integrative approach for large-scale planning and implementation of wetland restoration and creation (preferably at large watershed scale) for improving the water quality of agricultural areas is needed. This, along with a rational use of fertilizers, pesticides, and water, and a food production strategy based on and adapted to the environmental and socio-economic characteristics of the landscape will enhance the socio-ecological development of people and their environment.

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Denitrification in Constructed Wetlands for Wastewater Treatment and Created Riverine Wetlands

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Ülo Mander, Julien Tournebize, and William J. Mitsch

Contents

Abstract

Human activities have altered the nitrogen (N) cycle substantially at both local, regional and global levels. As a result, the availability of reactive N in the environment has greatly increased, causing the leaching of nitrogen surface waters, groundwater and oceans and creating eutrophication of freshwater ecosystems, hypoxia in coastal waters and pollution of groundwater. Constructed wetlands (CW) can be used to remove the excess reactive N from water environment, whereas nitrate, the most common component of nitrogen in the aquatic environment, is normally removed during the denitrification. Opportunities for this approach are presented, offering examples from various parts of the world. The problems of controlling emission of nitrous oxide, which is a byproduct of denitrification and being a dangerous greenhouse gas and destroyer of ozone layer, are also discussed.

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Introduction

Agriculture and food processing, the combustion of fossil fuels, and other human activities have altered the cycle of nitrogen (N) substantially, generally increasing both the availability and the mobility of N over large regions of Earth (Vitousek et al. 1997). As one of the key characteristics of these alterations, the availability of reactive nitrogen in the environment has greatly increased, considerably changing the nitrogen cycle locally, regionally, and globally. In the beginning of the past century, the Haber– Bosch process was developed, which produces ammonia (NH_3) from N_2 and hydrogen (H_2). As a result, global anthropogenic production of reactive N increased dramatically and exceeded all natural terrestrial production and it is projected to increase in the future as human populations and per capita resource use rise (Galloway et al. 2003). This leads to leaching of N to surface waters through seepage of nitrate-polluted groundwater and surface runoff. In addition, domestic and industrial waste disposal contribute to nitrogen pollution. Consequently, riverine N fluxes have increased 2–20 fold in the past century (Howarth et al. 1996).

The almost exponential production of global reactive N has resulted in its accumulation in the environment contributing to a loss of biodiversity and habitat degradation in coastal and terrestrial surface waters (Galloway et al. 2003). On the other hand, increased fossil-fuel combustion is a major source of N pollution through the emission of reactive nitrogen (NO) into the atmosphere as a waste product.

In surface waters, increased N concentrations contribute to acidification and eutrophication effects, such as altered plant productivity, harmful phytoplankton blooms, floating plant cover, temporal anoxia and consequently fish kills, biodiversity loss, and losses of ecosystem services. Many freshwater ecosystems in these areas are therefore highly eutrophic and contribute to the eutrophication of receiving waters, namely rivers, lakes, and reservoirs, and finally also coastal zones and oceans.

The most evident effects of eutrophication in shallow freshwater ecosystems are shifts in the dominant vegetation, altered biogeochemistry, and the loss of biodiversity. Eutrophication can stimulate decomposition and alter the coupling of biogeochemical cycles, with cascading effects on water quality (Howarth et al. 2011). For example, nitrate addition can couple sulfide oxidation to denitrification, resulting in a release of sulfate from the sediment, and consequently iron reduction and the release of iron-bound phosphorus into the water column.

Treatment Wetlands

Natural wetland ecosystems act as sinks and transformers of nutrients and carbon (Mitsch and Gosselink 2007). This ability of wetlands has led to a widespread use of natural and constructed wetlands for water quality improvement (Vymazal 2007).

Constructed wetland (CW) systems are fully human-made wetlands for wastewater treatment, which apply various technological designs, using natural wetland processes associated with wetland hydrology, soils, microbes, and plants. These systems are used to improve the quality of water polluted with various point and nonpoint sources of water pollution, from stormwater to domestic, agricultural, and industrial wastewater. The most commonly used classification of CWs, which is also used in this chapter, is based on their flow regime: free water surface (FWS) or surface flow, horizontal subsurface flow (HSSF), and vertical subsurface flow (VSSF) CWs (Vymazal 2007).

Created wetland ecosystems that are located in the riparian zone or in alluvial plains and are at least temporarily connected with the river are known as created riverine (riparian) wetlands (CRW) (Mitsch and Gosselink 2007). The CWs and CRWs together are usually called "treatment wetlands" (Kadlec and Wallace 2008).

Denitrification

Denitrification, as the microbial reduction of NO_3^--N to NO_2^--N and further to gaseous forms of NO, N_2O, and N_2, is a key process in nitrogen cycling and closes the entire cycle (Fig. 1). Numerous studies have found denitrification to be a significant process in nitrogen removal in treatment wetlands (Kadlec and Wallace 2008).

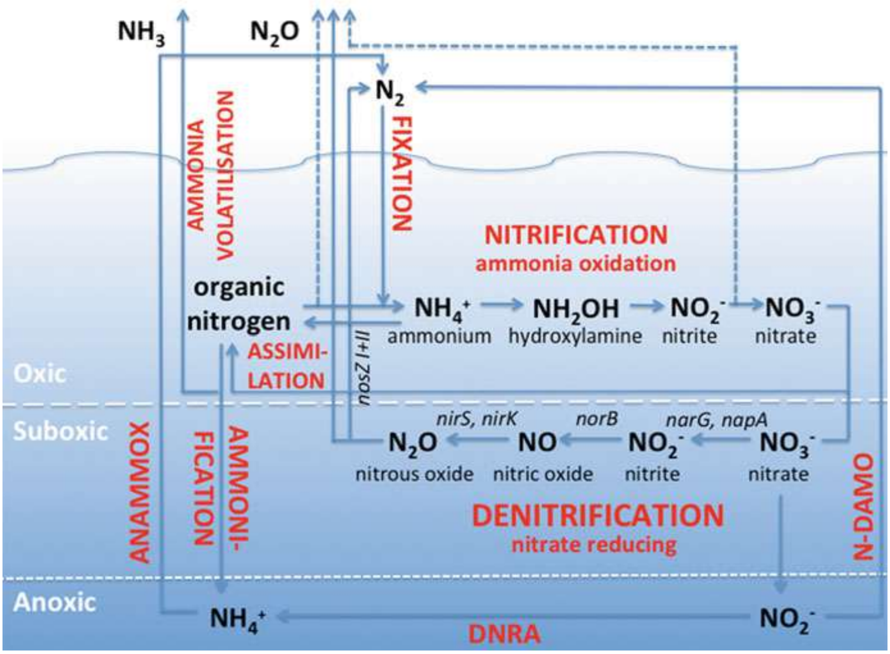

Fig. 1 The role of denitrification in the nitrogen cycle. Denitrification enzymes and relevant functional genes (in brackets): nitrate reductase (*narG*, *napA*) which reduces nitrate to nitrite, nitrite reductase (*nirS*, *nirK*), nitric oxide reductase (*norB*), and nitrous oxide reductase (*nosZ*) (Figure developed based on Fig. 1 in Francis et al. 2007)

Heterotrophic denitrification (also known as respiratory denitrification) is carried out by a wide array of facultative anaerobic microorganisms, including bacteria, archaea, and eukaryotes. It requires easily degradable organic carbon ($>$ 1.07 g C per 1 g NO3-N denitrified) as the electron donor and nitrate as electron acceptor. The overall reaction-equation can be described as follows:

$$5(CH_2O) + 4NO_3^- + 4H^+ \Rightarrow 5CO_2 + 2N_2 + 7H_2O \quad (1)$$

In the case of limited carbon resources, the denitrification is noncomplete and nitrous oxide is formed:

$$4(CH_2O) + 4NO_3^- \Rightarrow 4HCO_3^- + 2N_2O + 2H_2O \quad (2)$$

Some denitrifiers are chemolithoautotrophs and use other electron donors, such as sulfide and iron, at the expense of nitrate reduction:

$$5HS^- + 8NO_3^- + 3H^+ \Rightarrow 5SO_4^{2-} + 4N_2 + 4H_2O \quad (3)$$

Heterotrophic denitrification is considered to be the dominant form of denitrification in nonsulfidic systems with high carbon loads, such as wetlands, ditches, streams, and shallow lakes. Abiotic or chemodenitrification can also occur, when nitrite reacts with reductors present in the environment. Most microorganisms preferentially denitrify under anoxic or suboxic conditions (Knowles 1982), because aerobic oxidation of organic carbon yields more energy. However, aerobic denitrification has been found to occur as well (Reddy and DeLaune 2008).

Total denitrification from nitrate to dinitrogen gas is performed in four different reactions (Fig. 1), each catalyzed by different enzymes: nitrate reductase (*Nar, Nap*) which reduces nitrate to nitrite, nitrite reductase (*Nir*), nitric oxide reductase (*Nor*), and nitrous oxide reductase (*Nos*). Nir reduces nitrite to nitric oxide. The last step in the denitrification pathway is the reduction of nitrous oxide, which is catalyzed by nitrous oxide reductase (*Nos*), encoded by the *nosZ* gene. Under certain conditions, the last denitrification reaction is not performed, either because the denitrifiers do not possess the necessary *Nos*-enzyme or because it is blocked by environmental factors (Philippot and Hallin 2005). In this case, N_2O is the end-product of denitrification. Possibly, the lack of *nosZ* gene copies is a reason of relatively low denitrification intensity in the treatment wetlands (Batson et al. 2012).

Nitrous Oxide

Nitrous oxide, a key compound in the N cycle, is an intermediate of biological denitrification and a byproduct of biological nitrification. However, both biotic and abiotic processes are involved in the production of nitrous oxide (N_2O) in soil environments.

N_2O traps radiant energy 296 times more efficiently than CO_2, thereby contributing about 6% to the overall anthropogenic global warming trend (IPCC 2007). Despite its minor contribution to global warming, a small percentage of increase in emissions can lead to a large accumulation of N_2O in the troposphere, a phenomenon resulting from the long residence time of N_2O, approximately 120 years. In addition, N_2O is one of the most dangerous reagents disturbing the stratospheric ozone layer. The global emission of N_2O to the atmosphere is estimated to be 17.7 Tg N_2O-N year^{-1}, and the observed increase in emission is 5% per year (IPCC 2007).

Denitrification in Treatment Wetlands

The denitrification process plays a significant role in N removal in FWS and HSSF CWs (Kadlec and Wallace 2008). In VSSF CWs, this process can happen in anaerobic microsites and during the loading phase of VSSF beds, making it in combination with nitrification, a countable source of N_2O emissions (Teiter and Mander 2005).

Following characteristics are often used for the characterization of denitrification performance of treatment wetlands: area (A; ha or m^2), hydrological loading rate (HLR; mm d^{-1}), hydrological retention time (HRT; d), inflow (C_{in}) and outflow concentrations (C_{out}) of NO_3^--N (mg L^{-1}), inflow and outflow NO_3^--N loading rates (g m^{-2} year^{-1} or mg m^{-2} d^{-1}), NO_3^--N percent removal (%), NO_3^--N removal rate (LR; g m^{-2} year^{-1} or mg m^{-2} d^{-1}), and the removal rate coefficient k (m year^{-1} or m d^{-1}) (Kadlec and Wallace 2008).

The removal rate coefficient k comes from a first-order area based equation with temperature correction (Kadlec and Knight 1996), which is often used to describe the nutrient retention kinetics in constructed wetlands:

$$(\mathrm{C_{out}} - \mathrm{C^*})/(\mathrm{C_{in}} - \mathrm{C^*}) = \mathrm{e}^{(\mathrm{k} * \mathrm{A/Q})} \tag{4}$$

and the modified Arrhenius relationship for temperature impact:

$$\mathrm{k} = \mathrm{k_{20}} * \theta^{(\mathrm{T-20})} \tag{5}$$

where Q is the mean water flow (m^3 year^{-1} or m^{3}d^{-1}), k is the area-based first-order rate coefficient (in m year^{-1} or m d^{-1}), k_{20} is the temperature corrected rate coefficient (in m year^{-1} or m month^{-1}), and T is the air temperature (°C). The value of θ is often taken to be 1.08, although it varies between 1.00 and 1.21in various systems (Kadlec and Wallace 2008).

In general, in the annual N budget of CWs and CRWs the denitrification can make 50–80% of the nitrate removal, followed by plant uptake of 5–30% (Mitsch and Gosselink, 2007; Kadlec and Wallace 2008).

Carbon availability (the C:N or COD:N ratio in inflow; COD means chemical oxygen demand) and the hydrological retention time (HRT) are the most significant

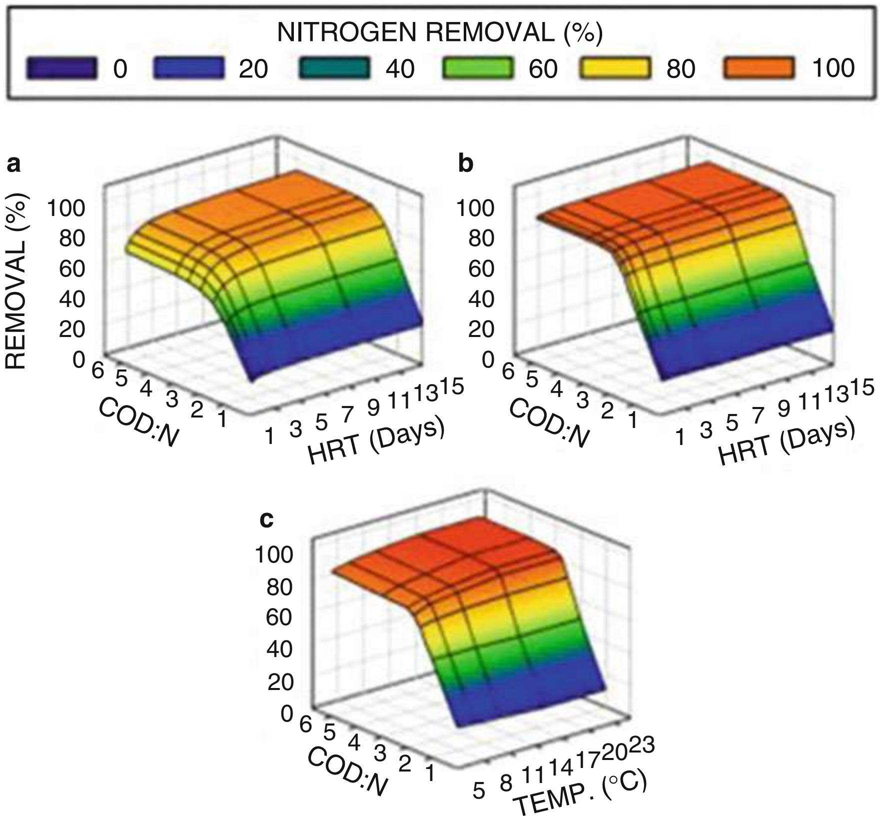

Fig. 2 A model simulation of the effect of COD:N ratio and HRT on nitrogen removal in a continuous-flow system at (**a**) 5 and (**b**) 15 °C; (**c**) effect of COD:N ratio (1–6) and temperature (5–22 °C) on nitrogen removal at an HRT of 5 days (75% denitrifying biomass retention is assumed in all simulations) (Reprinted from Misiti et al. (2011) with permission from Elsevier)

factors influencing intensity of denitrification in both CWs (Kadlec and Knight 1996) and CRWs (Song et al. 2014). The C:N ratio is itself highly predictive of nitrate removal efficiency. An example of a model simulation effectively illustrates the relationship between the COD:N ratio, HRT, temperature, and NO_3^--N removal (%) (Fig. 2).

Nitrous Oxide Emission in Treatment Wetlands

Constructed wetlands and created riverine wetlands can be significant sources of N_2O. An analysis of about 25 CWs and 3 CRWs shows that N_2O-N emissions vary in a wide range between the individual systems of certain construction types as well between the different types, showing significantly lower values in

Table 1 Average ± standard error values of N_2O emission and emission factors (EF) in treatment wetlands. TN_{in} – inflow total nitrogen loading (mg N m^{-2} d^{-1}). The letters x-y and a-d indicate significant ($p < 0.05$) differences between EF values and N_2O emission values in different types of CWs, respectively. SE – standard error. Based on data from Mander et al. (2014a, b)

Type of wetland	Emission (mg N_2O-N m^{-2} h^{-1})	EF (N_2O-N/TN_{in}) (%)
Free water surface (FWS) wetlands	$^{x}0.11 \pm 0.02$	$^{a}0.13 \pm 0.024$
Horizontal subsurface flow (HSSF) wetlands	$^{x}0.24 \pm 0.10$	$^{b}0.79 \pm 0.38$
Vertical subsurface flow (VSSF) wetlands	$^{x}0.14 \pm 0.025$	$^{c}0.023 \pm 0.005$
Created riverine wetlands (CRW) without plants	$^{x}0.10 \pm 0.05$	$^{d}2.0 \pm 1.0$
With plants	$^{y}0.021 \pm 0.005$	$^{b}0.51 \pm 0.08$

vegetated CRWs (Table 1). Average N_2O emission factors (EF) differed significantly in all three CW types being highest in HSSF CWs and CRWS without vegetation (Table 1).

The EF values for N_2O in FWS, CWs, and CWRs (0.13–2.0%) are lower than those found on many agricultural soils, where annual EFs varied from 0.4 to 6.5% of the N applied, and were comparable with the present IPCC default EF for N_2O of 1.25 ± 1.0% of the N applied (IPCC 2007).

Among the physical-chemical factors and processes influencing N_2O emission in constructed wetlands for wastewater treatment and created riverine wetlands, water/soil/air temperature, moisture in the soil or filter material and water table depth, wastewater loading rate, the presence of plants, and pulsing hydrological regime (intermittent loading) have been found to be most important.

Findings from studies on GHG emissions in HSSF and VSSF CWs which show high N_2 and low N_2O emission – indicators of almost complete denitrification – suggest denitrification might be the main source of nitrous oxide (Mander et al. 2008).

Future Challenges

Among further investigations that would contribute to a better understanding of C and N cycling, the continuous measurement of CO_2, CH_4, and N_2O fluxes using transparent automatic chamber method and eddy covariance technology can be highlighted. In order to distinguish between different sources of nitrous oxide (denitrification vs. nitrification), N_2O isotopomer studies have yielded promising results and must be used in further investigations in wetlands. Also, wider use of methods of the direct measurements of N_2 emission and adequate estimations of N_2:N_2O ratio in wetlands is foreseen. Better understanding of the role of microorganisms in denitrification and N_2O emissions is another important challenge for further investigations. Analysis of the structure of microbial communities and functional gene abundance and diversity studies using next-generation sequencing and quantitative PCR techniques are already widely used and promising results are expected.

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Biodiversity-Ecosystem Function (BEF) Theory and Wetland Restoration

James Doherty and Joy B. Zedler

Contents

Abstract

Biodiversity-ecosystem function (BEF) theory was founded on the idea that levels of ecosystem functions (e.g., productivity, nutrient cycling, decomposition) and the stability of those functions depend directly on levels of biodiversity, including diversity of all biota at the level of genotypes, species, and functional groups (sets of physiologically or morphologically similar species). Ecosystem functions are typically estimated from measures of stocks, e.g., plant biomass or nutrient crop, in response to vascular plant diversity (which can be easily manipulated in experiments). To date, the vast majority of experimental tests indicate that, on average, diversity increases productivity. Experimental outcomes have prompted BEF researchers to call on restoration ecologists to apply BEF theory by establishing more diverse biotic communities to increase ecosystem function. Using BEF theory relevant to its application, outcomes from experiments testing that theory with wetland plants and outcomes from experiments for wetland restoration projects suggest that a major challenge in applying BEF theory in wetlands is to establish and maintain plant diversity. However, applying BEF theory in wetland restoration does not simply mean adding more species to

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plantings, but to further explore how plantings can help achieve target ecological functions. Field experiments that vary plant composition and diversity, or that indirectly increase bird, insect, and bacterial diversity, would advance the practice of wetland restoration while ground-truthing BEF theory.

Keywords
Ecosystem functions · Wetland restoration · Field experiments · Biodiversity

Introduction

Biodiversity-ecosystem function (BEF) theory was founded on the idea that levels of ecosystem functions (e.g., productivity, nutrient cycling, decomposition) and the stability of those functions depend directly on levels of biodiversity, including diversity of all biota at the level of genotypes, species, and functional groups (sets of physiologically or morphologically similar species). Ecosystem functions are typically estimated from measures of stocks, e.g., plant biomass or nutrient crop, in response to vascular plant diversity (which can be easily manipulated in experiments). To date, the vast majority of experimental tests (414 of 479 reviewed by Cardinale et al. 2011) indicate that, on average, diversity increases productivity. Experimental outcomes have prompted BEF researchers (e.g., Naeem 2006) to call on restorationists to apply BEF theory by establishing more diverse biotic communities to increase ecosystem function.

A seminal model of ecological restoration by Bradshaw (1984) promoted a similar approach: recover biodiversity to develop ecosystem function. In restored wetlands, both biodiversity and ecosystem function are often lower than in natural sites that restorationists use as references or targets (Moreno-Mateos et al. 2012). It is not yet clear if functional outcomes in wetland restoration projects can be improved with more diverse plantings, even though this is often the case in small-scale experiments.

Following are elements of BEF theory relevant to its application, outcomes from experiments testing BEF theory with wetland plants, and outcomes from experiments testing BEF theory in wetland restoration projects. The authors then suggest future challenges for BEF theory in wetland restoration.

BEF Theory in Applied Settings

Evidence for BEF theory comes from "BEF experiments," hundreds of similarly designed experiments carried out in the last two decades (Cardinale et al. 2011). Among the best-known BEF experiments are large grassland field experiments, which showed that species-rich plantings can increase productivity (Tilman 1999; Hector et al. 1999). The same experiments also showed that species-rich plantings had more consistent levels of productivity over time (Tilman et al. 2006) and higher rates of nutrient uptake and decomposition (Hector and Bagchi 2007). Thus, BEF experiments suggest that more functions or higher levels of functions would be found where more plant species are restored.

However, benefits found in BEF experiments are not readily transferrable to restoration projects. BEF researchers have designed their experiments to vary only plant diversity. The resulting experimental plantings are unlike nonexperimental plant communities in several ways: (1) planted assemblages are composed at random, (2) species evenness is very high because all species are initially planted at equal abundance, (3) all species are planted as monotypes, even though some are unlikely to grow alone in nature, and (4) plants that colonize assemblages (weeds) are removed. Each of those features can diminish the effects of individual species and dominance, both of which influence ecosystem function in nonexperimental systems.

Early on, BEF researchers recognized that particular species and functional groups were affecting experimental outcomes. In grassland experiments, the inclusion of legumes (N-fixing plants) seemed to account for higher productivity in diverse assemblages. Researchers identified two mechanisms that could explain how diversity increases function: (1) selection effects (SELs), in which the number of species in an assemblage is proportional to the likelihood that a high-function species (e.g., a legume) is present, or (2) complementarity effects (COMs), in which the number of species in an assemblage increases the capacity for complementary use of limiting resources (resource segregation, e.g., interspecific variation in rooting depths allowing more efficient nutrient uptake). In over 200 experiments, diversity effects have been attributed almost equally to SELs and COMs (Cardinale et al. 2011). COMs strongly support the argument that many species affect function and should be planted, while SELs suggest that restorationists focus on restoring plant species with desirable traits, rather than maximizing diversity.

Restorationists do not choose species for plantings at random, especially when planting just one. In contrast, BEF researchers have typically compared the average level of function across all monotypes vs. the average level of function across all assemblages. Most of those experimental monotypes would never be planted in a restoration. The review by Cardinale et al. (2011) instead compared the highest-function monotype to the highest-function assemblage and found that the former had higher productivity in 237 of 375 experimental tests and higher rates of nutrient uptake in 44 of 47 experimental tests. The top-performing monotype is the appropriate functional benchmark in practical applications of BEF theory. For restorationists to increase ecological function via high diversity plantings, such plantings must, in some way, surpass a lower-diversity, practitioner-selected alternative.

BEF Theory in Wetland Experiments

Relatively few BEF experiments involve wetland vegetation (7 of the 479 experimental observations of diversity-productivity relationships reviewed by Cardinale et al. 2011). Wetland BEF experiments have shown positive diversity-productivity relationships, with the exception of two mesocosm experiments where no trends were detected (Engelhardt and Ritchie 2001; Doherty and Zedler in press). Other wetland experiments have shown that planting more species or functional groups increased emergent macrophyte biomass (Bouchard et al. 2007), seagrass biomass (Gustafsson and Bostrom 2011), and salt marsh plant biomass (Callaway et al. 2003; Sullivan et al. 2007).

Like other BEF experiments, those in wetlands suggest that diverse plantings can increase multiple ecosystem functions. Some functions likely increase along with productivity in diverse plots, e.g., plant N accumulation (Callaway et al. 2003). Other functions might depend more directly on vascular plant diversity, including macroinfauna abundance and diversity (Gustafsson and Bostrom 2011), algal biomass and P-removal (Engelhardt and Ritchie 2001), and reduction in methane emission (Bouchard et al. 2007). Wetland plant diversity has the potential to support multiple functions and biotic structure and function at other trophic levels.

In some cases, diversity effects were driven by particular wetland species or functional groups. For example, the presence of a particular submergent macrophyte favored high-algal biomass and P-removal (Engelhardt and Ritchie 2001), and a particular emergent macrophyte functional group substantially increased levels of denitrification potential (Bouchard et al. 2007). Among researchers who could partition diversity effects into SELs and COMs, results were mixed. For Gustafsson and Bostrom (2011) COMs consistently exceeded SELs, whereas Sullivan et al. (2007) found no COMs but strong SELs (assemblage productivity was largely determined by whether or not the most productive species had been planted).

BEF Theory in Wetland Restorations

Naeem (2006) recommended embedding BEF experiments in restoration projects as a first step to applying BEF theory. The previously mentioned experiment by Callaway et al. (2003) did that, with plantings of one, three, or six species in a newly excavated salt marsh plain and regular monitoring as functions developed. Keer and Zedler (2002) documented plant community structure while the experiment was being weeded to maintain species-richness treatments (within 18 months of planting); at that time, planted species were nearly equal in frequency and abundance, as expected for a weeded, immature BEF experiment (Fig. 1a, b). However, without weeding for several years, the majority of the planted species became rare, and the tallest and most productive of those species became the most frequent and abundant (occupying 100% of plots and contributing 52% of all biomass); those changes resulted in a shift toward negative diversity-biomass correlations (Doherty et al. 2011).

The authors observed similar outcomes in two additional BEF experiments involving wetland plantings: (1) three swales (each 0.15-ha created wetlands) were planted with three or nine species of wet prairie herbs and (2) the edge of a 2.4-ha created pond planted with one, three, or six species of native emergent macrophytes. In both cases, tall and highly productive *Typha* spp. invaded and became by far the most abundant taxon (Fig. 1c, d). In the swales and in a nearby mature restored wetland, the authors found negative species richness-biomass correlations (Doherty and Zedler in press). Several other authors have noted similar negative patterns in

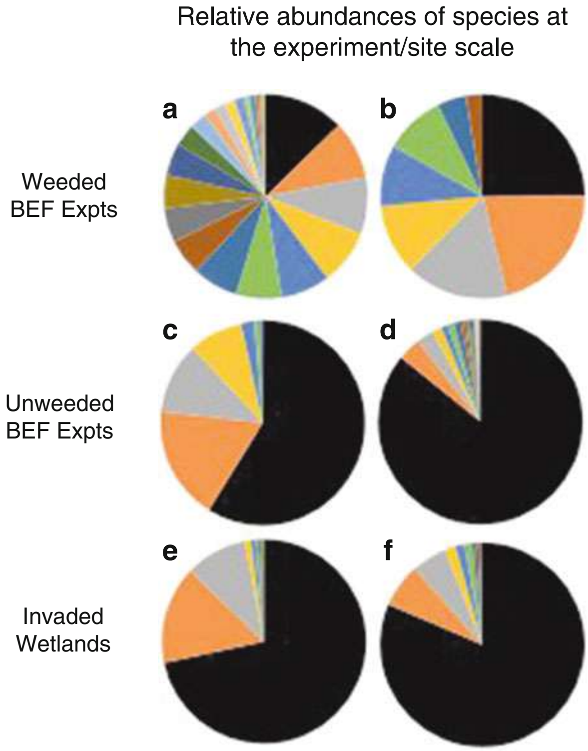

Fig. 1 Relative abundances of wetland plants were more similar in weeded than unweeded plantings, as indicated by: shoot biomass in weeded wet prairie mesocosms (**a**; Doherty and Zedler in press), point-intercept sampling in a weeded salt marsh field experiment (**b**; Keer and Zedler 2002), cover in experimental pond plantings (**c**; Doherty, unpubl. data), shoot biomass in experimental swale plantings (**d**; Doherty and Zedler in press), and cover in wetlands invaded by *Phalarisarundinacea* and *Glyceria maxima* (**e** and **f**, respectively; Olson and Doherty 2012)

highly productive wetlands (e.g., Schultz et al. 2011). Researchers have generally attributed negative species richness-biomass correlations to dominance by highly productive species that excluded other species.

Invasion of restored or existing wetlands by highly productive species, such as *Typha* spp., *Phragmites australis*, *Phalaris arundinacea*, *Glyceria maxima*, etc., is relatively common and can greatly reduce diversity (Fig. 1e, f). Such invasion can (1) overwhelm the effect of diverse plantings where wetland vegetation is established de novo and (2) thwart efforts to increase biodiversity in already-invaded sites. Thus, while species-rich plantings have the potential to increase ecosystem function in weeded experiments, the benefits of diversity are unlikely to be sustainable in nonexperimental systems after highly productive species invade.

Future Challenges

A major challenge in applying BEF theory in wetlands is to establish and maintain plant diversity. Where highly productive species become dominant, BEF theory is not applicable unless stressors (e.g., excess nutrients, invasive species) can be curtailed and diversity restored. Where high diversity can be established, BEF theory should be field-tested. In such field sites, restoration ecologists should pursue basic

BEF experiments (e.g., varying plant diversity parameters alone) to test whether the results of small-scale experiments are reproducible in the field. Further experiments should incorporate realistic wetland community composition and dominance to identify species or groups of species that strongly support wetland functions that are targets for restoration.

Applying BEF theory in wetland restoration does not simply mean adding more species to plantings. Instead, the role of BEF theory in wetland restoration is to prompt ecologists and practitioners to devise plantings that help achieve target functions. Field experiments that vary plant composition and diversity, or that indirectly increase bird, insect, and bacterial diversity, would advance the practice of wetland restoration while ground-truthing BEF theory.

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Economics of Wetland Restoration and Creation

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Edward Barbier

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Abstract

As the overall aim of wetland restoration, enhancement, and creation is to recover valuable ecosystem goods and services, assessing these benefits is vital. In addition, as wetland restoration and creation are not "costless" activities, they must be assessed in terms of their cost effectiveness and their impacts on different stakeholders. Evaluating these various benefits and costs, as well as the distribution of their impacts, is critical in determining the success or failure of wetland restoration and creation of policies and investments.

Keywords

Cost-benefit analysis · Ecosystem services · Wetland creation · Wetland restoration

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Introduction

As the overall aim of wetland restoration, enhancement, and creation is to recover valuable ecosystem goods and services, assessing these benefits is vital. In addition, as wetland restoration and creation are not "costless" activities, they must be assessed in terms of their cost effectiveness and their impacts on different stakeholders. Evaluating these various benefits and costs, as well as the distribution of their impacts, is critical in determining the success or failure of wetland restoration and creation of policies and investments.

Limitations

One of the major limitations to successful wetland creation or restoration is the assumption that all wetlands create "equivalent" economic benefits. For example, since the Clean Water Act of the 1970s, the US government has instigated a variety of policies to encourage wetland creation or restoration as compensation for wetlands damaged or lost through development. This policy of "compensatory wetland mitigation" to achieve "no net loss" of wetlands in the United States has assumed that both the structure and functions of destroyed wetlands can be adequately reestablished elsewhere by the new wetlands. However, this critical assumption has been challenged by a number of studies, which have found that too much emphasis has been placed on recreating the acreage of wetland area lost rather than ensuring that the restored or created wetlands provide an equivalent ecological structure and functions necessary to generate valued wetland goods and services (Barbier 2011a; Bendor 2009; Dale and Gerlak 2007; Zedler and Kercher 2005). Reviewing compliance performance for 76 restored wetland sites in Illinois, Matthews and Endress (2008) found that the monitored indicators of success were based exclusively on the quantity and quality of the plant communities established in restored wetlands, and that the performance standards were often unclear, did not set measurable targets, varied considerably from site to site, were poor indicators of site performance and wetland functions, and were set arbitrarily without reference to similar natural or restored wetlands. Similar findings have been noted for restoring coastal and estuarine wetlands in the United States, where poor site location with respect to the surrounding landscape and lack of consideration of the appropriate hydrological regime have been common factors in restoration failure (Barbier 2011a; Lewis and Gilmore 2007; Simenstad et al. 2006).

Landscape-Scale Restoration

The increasing focus on landscape level considerations in the restoration of wetlands has meant that economic valuation of alternative land use scenarios and their impacts on conservation versus development trade-offs is becoming a priority (Barbier 2013). For example, Posthumus et al. (2010) examine six alternative floodplain management scenarios to reflect different priorities for land use in lowland floodplain areas of England. Although obvious conflicts emerge (such as between agricultural production and water quality, carbon sequestration, and habitat and species conservation), some development and environmental benefits are complementary (such as agricultural production and flood storage and control). Because the financial returns to different land uses are sensitive to farm input and output prices, there is scope to develop combined floodplain restoration and agricultural regimes that provide a balance of land and water management benefits that appeal to a wide range of stakeholders, including farmers and local communities, and conservationists and flood managers.

Jenkins et al. (2010) conduct a more conventional cost-benefit analysis of restoring over 200,000 hectares (ha) of forested wetlands in the Mississippi Alluvial Valley, a floodplain area below the confluence of the Mississippi and Ohio Rivers in the United States. The authors were able to provide lower bound estimates on the total ecosystem value of the wetland restoration by quantifying the benefits from three ecosystem services: carbon sequestration, nitrogen runoff abatement, and waterfowl recreation. The total social value of these services amounts to between $1,435 and $1,486 per ha annually, which exceeds the full costs of wetland restoration after only 1 year and indicates a high social return on the public investment. In comparison, given existing markets that generate actual payments for these ecosystem services, their market value is only $70 per ha. But when fully accounting for potential markets, this value rises to $1,035 per ha annually. This potential market value suggests that payments to private landowners to restore wetlands could also be profitable for individual landowners.

Large-scale wetland restoration projects need also to be assessed for their appeal to different stakeholder groups, especially when there are several alternative restoration options. Milon and Scrogin (2006) analyze three distinct groups, who vary significantly in socioeconomic characteristics and in their preferences for ecosystem restoration of the Greater Everglades in Florida, to assess their willingness-to-pay (WTP) for different restoration options. The results of the stakeholder analysis suggest that public support and WTP for Everglades restoration is more likely to favor plans that emphasize conserving key populations of native fauna than hydrological regime restoration and management, which are currently stressed by wetland scientists and the US Army Corps of Engineers as the proposed restoration plan. In concluding their analysis, Milon and Scrogin (2006, p. 172) make an important observation: “Policy analysis for wetland ecosystems is especially difficult because these systems provide multiple, interdependent services that vary by type of wetland, location, ecohydrological management, and other factors.”

Community Involvement

Too often, policies for wetland restoration and creation focus exclusively on the rehabilitation of these natural systems without adequate attention to the economic incentives determining local communities’ attitudes and assistance in these efforts.

In developing countries especially, unless local communities have more of a say in the control, use, and protection of wetlands, restoration projects will fail to have any lasting results. Several studies of replanting mangroves in Thailand illustrate this connection.

An analysis of four coastal communities reveals that awareness of community conservation efforts, of community-imposed utilization rules, and of the mangrove destruction inflicted by shrimp farms are key motivating factors in the decision by male and female members of mangrove-dependent households to participate in replanting activities (Barbier 2008). As a consequence, there may be more willingness to participate in mangrove rehabilitation as a means to combat mangrove loss due to shrimp farm expansion and other developments, but equally important is the degree of control the community has over managing the mangroves and their utilization. Similarly, in a mangrove forest rehabilitation project in Pattani Bay, local ownership of the project and effective community participation by three surrounding villages were crucial to the successful restoration of degraded mangroves (Erftemeijer and Bualuang 2002). Community surveys throughout Thailand have confirmed that where local villages have been allowed to design and maintain well-defined governance structures over mangroves, stand structure was superior in these community-managed forests than in open-access state forests (Sudtongkong and Webb 2008). The lessons learned from these and similar studies of the motivation of local communities to participate in mangrove replanting projects should be applied to designing the correct institutions and incentives for the larger mangrove rehabilitation schemes planned for many coastal areas across Asia, since the Indian Ocean Tsunami in 2004 (Aswani et al. 2012).

Future Challenges

A review of the peer-reviewed literature on ecosystem restoration found that socioeconomic benefits are generally not adequately quantified and assessed and that aquatic ecosystems (including wetlands) are poorly represented (Aronson et al. 2010). A comprehensive review of nonmarket value estimates for US coastal and marine environments suggested that our knowledge of key habitat values was insufficient to support effective policy-making and management, including restoration efforts (Pendleton et al. 2007). Although valuation of key ecosystem services of freshwater and coastal wetlands has improved in recent years, there are many services and wetland systems that still need assessing (Barbier 2011b; Barbier 2013). Considerable progress in this area is essential for adequate economic assessment of wetland restoration and creation efforts.

Wetland restoration must also take into account the wider policy and development context that led to the loss of the natural systems in the first place. This calls for broader economic assessment of the costs and effectiveness of wetland restoration, enhancement and creation projects, stakeholder preferences, and the potential role of payments for wetland services.

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Plant Community Reassembly in Restored Wetlands

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Susan Galatowitsch

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Abstract

When wetlands are restored to reverse ecosystem degradation caused by anthropogenic change, the hope is that plant and animal communities will efficiently reassemble once stressors on the ecosystem have been minimized. In many situations, however, anthropogenic change is so severe or widespread that reassembly occurs slowly, if at all, without active seeding or planting. A common need for wetland restoration projects is to anticipate how much active intervention is needed for plants and animal communities to recover. Studies of plant reassembly in restored wetlands indicate that recolonization potential depends on three main factors: the level of site degradation, the extent of anthropogenic change to wetlands in the surrounding landscape, and the kind of wetland being restored. Which species actually become established depends on a fourth factor, the array of environmental conditions in the restored wetland. The longest lags of

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recolonization will likely occur in restored wetlands that lack remnant vegetation and seedbanks and that are isolated from extant wetlands. In these situations, actively seeding or planting dominant species is necessary to restore wetland vegetation.

Keywords
Wetland restoration · Wetland revegetation · Seedbanks · Seed dispersal

Introduction

Wetlands are restored to reverse ecosystem degradation caused by anthropogenic change, such as land conversion, water withdrawal, or infrastructure development. A wetland ecosystem cannot recover unless the main causes for degradation are addressed. For example, if groundwater withdrawals have depleted freshwater supplies to a desert oasis, causing it to become saline, ecosystem recovery will only be possible if withdrawals are reduced. So, the primary concern of any restoration project is to identify the drivers of change and work to correct them. In some cases, once these problems have been corrected, the environmental conditions return to normal and plant and animal communities quickly rebound. In other situations, though, ecosystem stress caused by anthropogenic change is so severe or widespread that biotic recovery occurs very slowly (if at all), unless additional restoration actions are taken. A common need for wetland restoration projects is to anticipate how much active intervention is needed for plants and animal communities to recover. Doing more than is necessary, wastes limited resources and potentially can do more harm than good. However, not doing enough can result in a restoration not achieving its desired results for the foreseeable future.

In contrast to terrestrial ecosystem restoration, most restored wetlands are not planted; their plant communities are allowed to reassemble through natural processes. A key reason for this difference is that many wetland plants produce seeds that can lie dormant in the soil for many years, surviving through unfavorable periods, such as prolonged high water or drought, until conditions are suitable for germination and growth. Thus, restorationists often assume that those stores of seeds in wetland soils (seedbanks) have withstood the period of degradation and will serve as a source of colonists once abiotic problems have been corrected. Many also assume that other wetland plants will arrive on their own, either by water or transported by animals. In reality, though, the likelihood that a plant community, similar to what existed on a site prior to degradation, will efficiently recolonize is quite variable. Studies of plant reassembly in restored wetlands indicate that recolonization potential depends on three main factors: the level of site degradation, the extent of anthropogenic change to wetlands in the surrounding landscape, and the kind of wetland being restored (e.g., Matthews et al. 2009; Mulhouse and Galatowitsch 2003). Which species actually become established depends on a fourth factor, the array of environmental conditions in the restored wetland.

Seedbank Presence

If some portions of a wetland have escaped severe degradation, these areas may harbor remnant patches of the original vegetation. Vegetative spread from these remnants can be very rapid, since many wetland plants are clonal. For example, emergent aquatic plants, often found along the margins of earthen ditches and canals in drained agricultural wetlands, may vegetatively spread from these corridors if hydrology is restored (Galatowitsch and van der Valk 1995). Seeds stored in the soils of these remnants (i.e., seedbanks) may also be transported to colonize other parts of the wetland. The role of seedbanks is, however, likely to be most important for recolonization where they exist throughout a site. Seedbanks are most likely to have persisted if the duration of degradation has been relatively short and the surface of the wetland is intact (Wienhold and van der Valk 1989). If the surface is removed, for example, by excavation or sod-cutting, the seedbank is removed (Grootjans et al. 2001). Dormant seeds of a seedbank are likely to persist through unfavorable periods that are within the natural range of conditions typical of that type of wetland. For example, prairie wetlands in the mid-continental US typically experience droughts that cause drawdowns every 5–10 years that trigger emergence of emergent marsh seedlings from seedbanks. Not surprisingly, wetlands that were drained for agricultural production 70 or more years ago have diminished seedbank diversity and density. So, on-site sources of propagules will be of limited or no importance to plant community reassembly in wetlands with very long histories of degradation (especially drainage) and degradation that has affected the entire wetland.

Seed Dispersal

Plant species without on-site propagules sources can colonize newly restored wetlands, if they are dispersed there. Seeds are carried from one wetland to another by water, wind, or animals. The abundance and diversity of seeds and vegetative propagules greatly depend on the level of connectivity between restored and extant wetlands in the surrounding landscape. Restored wetlands with surface water connections have the highest connectivity and so will likely receive the greatest influx of seeds. Water is an especially important mode of transportation because the seeds of many wetland species are buoyant. Even if they lack special adaptations, many seeds can be carried in moving water. For example, of 125 plant species that dispersed into a restored tidal freshwater marsh, 89 arrived via water, whereas only 10 were carried by geese and 39 by wind (Neff and Baldwin 2005). Restored wetlands lacking surface water connections to other wetlands depend on seeds arriving by wind or animals, which is much less efficient for most species. Restored prairie potholes which depended on overland seed dispersal received seeds of very few species (i.e., 6 native perennials over two years) and few seeds overall (i.e., 20 seeds m-2 per year) (Kettenring and Galatowitsch 2011). However, restored wetlands that are near but not connected to other wetlands in the landscape can accumulate more species over time, compared to those that are more distant, even though dispersal is inefficient (Galatowitsch 2006). So, wetland loss that removes connections between wetlands or increases their isolation likely diminishes the rate of plant recolonization during restoration.

The makeup of seeds arriving via dispersal, called seed rain, reflects the species composition of wetlands serving as propagule sources. Species that are most abundant and produce the most seeds are likely to comprise a large fraction of arriving seeds. This phenomenon is called propagule pressure. Where introduced, invasive species have spread through extant wetlands, propagule pressure will favor the arrival of these species into restored wetlands. For example, 93% of the seeds dispersing into restored prairie potholes were introduced (i.e., nonnative) species (Kettenring and Galatowitsch 2011). Species assemblages in dispersal-dependent restored wetlands may have little similarity to what occurred there prior to degradation.

Disturbance Effects

The frequency and magnitude of natural disturbances varies among different types of wetlands, and so the plant communities also vary in their capacity to recolonize after disturbances. Wetland plant communities vary in seedbank development: species that form long-lived seedbanks are most likely to be found were there are multiyear phases unfavorable to plant regeneration, interrupted by natural disturbances. Natural disturbances remove adult vegetation and change environmental conditions potentially encouraging seedling emergence and growth. Species with long-lived seeds stored in the soil can capitalize on these events and so will be well-represented in the plant community. Extant emergent marshes are widely reported to have well-developed seedbanks, with densities of $>10{,}000\ m^{-2}$ not uncommon; in contrast, wet meadows seedbanks more typically have densities of $<2{,}000\ m^{-2}$ (Galatowitsch and Biederman 1998).

The extent to which alterations to wetlands, such as drainage, diminishes seedbanks could also differ among wetland types, affecting recolonization potential. For example, of three wetland types found on the Kissimmee River Floodplains in Florida that had been drained, only one type (wet prairie) contained seedbanks with many species characteristic of that habitat; broadleaf marshes and wetland shrub communities, in contrast, contained no more than one such species (Wetzel et al. 2001). In this situation, broadleaf marsh and wetland shrub communities will depend more on dispersal of propagules for plant community reestablishment after restoration, than would wet prairies. Species capable of rapid dispersal and colonization are likely to be from wetland plant communities that experience frequent disturbances, such as annual flooding and drawdowns. These species often produce abundant seed, can germinate quickly, and have relative short-times to maturity.

Within a restored wetland, plant colonization is potentially most rapid for species with on-site remnant populations that can vegetatively spread and are already producing seed. If no remnant vegetation has survived degradation, plant colonization may still be relatively efficient if many species are well-represented in the seedbank. If seedbanks are no longer present because the duration or intensity of degradation has been too great, then many colonists will need to disperse to the site. The longest lags of recolonization will likely occur in restored wetlands that are isolated from extant wetlands. In these situations, actively seeding or planting dominant species is necessary to restore wetland vegetation. Dispersal limitations have been reported from restored wetland ecosystems, including salt marsh plains (Morzaria-Luna and Zedler 2007) and prairie meadows and marshes (Galatowitsch 2006).

Seed Survival

Not all species in the seedbank or seed rain will establish populations in restored wetlands. Conditions must be suitable for germination, establishment, and growth. Ecologists use the analogy of a filter or sieve to describe how site conditions regulate which species become part of a plant community (van der Valk 1981). All of the species with seeds on a particular site are referred to as the local species pool. Key environmental factors, such as water levels, salinity, or light act as filters (or sieves), selecting subsets of species that will form the initial vegetation of restored wetlands (Keddy 1999). So, for a wetland plant community to resemble what was present prior to degradation, these environmental conditions must be similar. If some aspect of the wetland, such as the water regime, is very different than what previously existed on the site, the plant community that establishes will not be the same, even if many of the seeds (or other propagules) were present. In landscapes where many wetlands are dominated by introduced species, these species will likely arrive most rapidly. When this occurs, later arriving species may be unable to establish if the initial colonists create unsuitable conditions.

Future Challenges

For wetland restoration to be a useful conservation strategy, capable of reversing adverse impacts at the scale at which they occur, it is critical to be able to pursue projects that are much larger than are feasible today. Because wetland vegetation reassembly through natural processes is often inefficient or incomplete, especially in human-modified landscapes, active intervention (i.e., seeding or planting) is often necessary for full restoration. Project size is frequently limited in situations where humans must compensate for the lack of wetland plant colonization: commercial seed and plant supplies are often minimal or nonexistent, the logistics of acquiring and installing seeds and plants is often very expensive, and seed technology information is lacking for wetland species. Thus, in most parts of the world, research on the seed technology and horticulture of wetland plants is needed to improve the efficiency of producing wetland plants and seeds in large quantities. This kind of research has been essential for expanding agricultural opportunities worldwide and, more recently, for facilitating terrestrial ecosystem restoration (especially grasslands). In some cases, methods from terrestrial species for production, handling, storage, and installation can be adapted for wetland species; however, new approaches will frequently be needed since these species are adapted to life in wet environments.

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Carbon and Nutrient (N, P) Cycling of Created and Restored Wetlands

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Owen Langman and Christopher Craft

Contents

Abstract

Wetlands are created or restored for a number of purposes, including flood control, water purification, sediment and nutrient retention, and biodiversity. Restoration of wetlands is the process of recreating former conditions at a site that once contained a wetland. Restoration and creation of wetlands involve the creation or restoration of hydrology, but may also include activities that alter soil composition, manipulate vegetative communities, or promote target wetland functions. Carbon accumulation in restored and created wetlands is dependent on hydrology, vascular vegetation, and microbial communities. Once hydrology is restored, consistent periods of soil inundation reduce rates of microbial decomposition by creating anoxic soils. Wetlands prior to restoration and terrestrial soils prior to development of wetland hydrology are usually carbon poor, and created wetlands often retain soil properties from the soils from which they are created, potentially taking tens of decades to build up carbon reserves similar to natural wetlands. Initial nitrogen pools reflect past land use. Agricultural sites are increasingly being converted into wetlands, resulting in many restored or created wetlands with large initial nitrogen pools due to legacy effects of fertilizer use.

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Since nitrogen removal via denitrification is often desirable in restored and created wetlands, increasing the duration of inundation in restored or created wetlands may be used to encourage denitrification, but microbial community biomass often is still limited by available organic carbon. Unlike carbon and nitrogen, phosphorus cycling in wetlands is primarily controlled by abiotic processes. Increasing water residence time and depth have been shown to increase soil phosphorus sorption. However, in restored or constructed wetland phosphorus release may occur in sites that initially contain elevated phosphorous levels in the soil, such as sites with agricultural legacies. Overall, the retention and recycling of carbon, nitrogen, and phosphorus are controlled by a variety of biotic and abiotic processes, themselves influenced by ambient oxygen and for restored wetlands initial nutrient concentrations that become less important over time.

Keywords
Carbon · Nitrogen · Phosphorus · Recycling · History · Chemistry · Restoration

Introduction

Wetlands are created or restored for a number of purposes, including flood control, water purification, sediment and nutrient retention, and biodiversity (Mitsch and Gosselink 1993). Restoration of wetlands is the process of recreating former conditions at a site that once contained a wetland. Wetland creation involves manipulating a site without a historical wetland so that it functions as a wetland. Success of wetland creation or restoration is usually determined by comparing structural (hydrology, vegetation structure) and functional (carbon and nutrient cycling) elements of the new wetland with a natural system. The new wetland will ideally be a natural, functioning, self-regulating system that is integrated within the ecological landscape in which it occurs (Palmer et al. 1997).

Restoration and Creation of Wetlands

Restoration and creation of wetlands involve the creation or restoration of hydrology but may also include activities that alter soil composition, manipulate vegetative communities, or promote target wetland functions. Developing wetland hydrology often involves the removal of dams, dikes, levees, or tide gates to restore the periodic inundation required to establish wetland vegetation and soils. Vegetative plantings, seeding, or manipulation of soil nutrient and carbon pools may be used to try to accelerate the return of wetland structure and function, although many elements of wetland structure and function only develop after prerequisite structure or function has developed.

Once hydrology is reestablished, wetlands theoretically follow general developmental trajectories that approach the structure and function of natural wetlands over several decades. In practice, wetland development is rarely smooth or rapid, and many restored and created wetlands fail to reach structural or functional equivalency

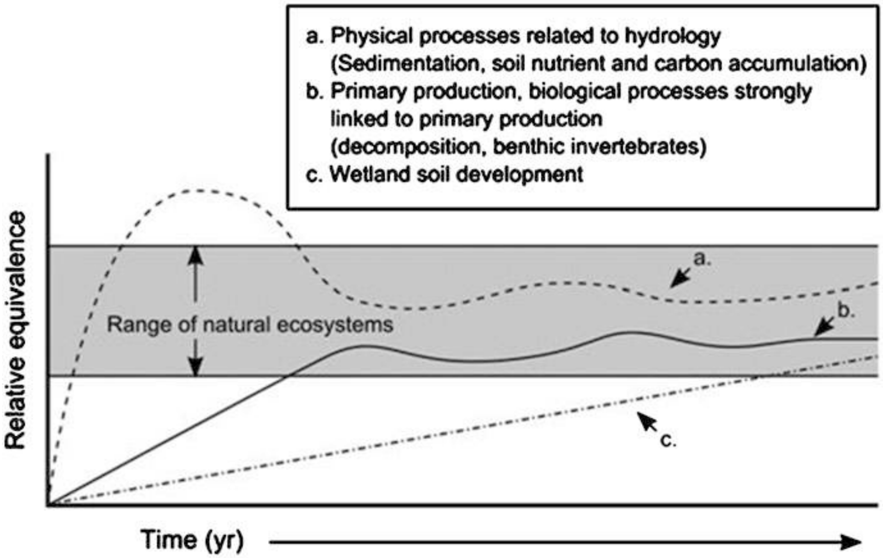

Fig. 1 Trajectories describing changes in functional and structural attributes during salt marsh ecosystem development (Figure recreated from Craft et al. 2003; used with permission from the Ecological Society of America and Wiley)

(Moreno-Mateos et al. 2012). Despite these limitations, general developmental patterns can be observed in restored and created wetlands. Processes that depend on wetland structure return relatively quickly. Vegetation establishes rapidly after the restoration or creation of wetland hydrology, within 1–3 years. Processes that depend on vegetation including sedimentation, surface water hydrology, and the rates of soil carbon and nitrogen accumulation achieve equivalence with natural marshes soon after vegetation establishment, generally within 5 years. These measurements are still not sufficient, however, to conclude that a restored or constructed marsh has achieved equivalence with a natural system. Vegetation structure, higher trophic level diversity, and benthic infauna abundance depend on carbon and nutrient pools which take significantly longer to reach parity with natural systems (Craft et al. 1988; Craft 2000). Wetland nitrogen pools begin to develop within a few years of wetland creation, but may require as long as 30–40 years for soil pools to reach equivalence, and soil carbon pools may only approximate a natural system after 70 years or longer (Fig. 1, Craft et al. 1999; Craft et al. 2003; Moreno-Mateos et al. 2012).

Carbon and Nutrient Cycling

Carbon accumulation in restored and created wetlands is dependent on hydrology, vascular vegetation, and microbial communities. Once hydrology is restored, consistent periods of soil inundation reduce rates of microbial decomposition by creating anoxic soils. Increased duration of inundation further decreases decomposition rates, which can create heterogeneity of decomposition rates and carbon pools within or among wetlands (Serna et al. 2013). Other factors that can affect decomposition rates include the quality and quantity of detritus additions, soil characteristics, and temperature which can vary both with season and with latitude. Since carbon input is highly dependent on vascular vegetation detritus, rates of carbon accumulation in restored marshes are similar to natural marshes within 3–5 years of the restoration of wetland hydrology, coincident with the development of aboveground biomass production (Craft et al. 2003).

The vertical gradient of carbon quality (C:N ratio) in restored and created wetlands is initially exacerbated relative to natural wetlands. In all wetlands, labile carbon fractions are decomposed more rapidly than recalcitrant forms resulting in a gradient in the soil profile of carbon where shallow, more recently deposited soil carbon is more decomposable than deeper carbon. In created or restored wetlands, this pattern is exaggerated. Initial detrital inputs are higher in quality compared to natural systems. In constructed and restored wetlands, live roots make up a larger proportion of vegetative material relative to natural wetlands where decomposing roots predominate. The increased quality of carbon inputs to the soil organic carbon pool drives the initial development of decomposer activity.

Initial carbon pools are smaller than natural sites and may take tens of decades to reach equivalency. Restored and created wetlands often retain soil properties from the soils from which they are created. Wetlands prior to restoration and terrestrial soils prior to development of wetland hydrology are usually carbon-poor, resulting in wetlands that have smaller carbon pools relative to natural sites. Evidence from long-term studies suggests that carbon pools are one of the last features to develop and may require 70 years or longer to achieve equivalency (Craft et al. 2003; Ballantine and Schneider 2009; Moreno-Mateos et al. 2012).

Nitrogen retention in wetlands is tied to the vegetation community. Nitrogen retention is initially higher in restored or created wetlands due to high demand from developing vegetation. The nitrogen retention during this period can be 2–4 times greater than retention in natural wetlands. After the establishment of vegetation biomass, nitrogen retention slows, eventually returning to rates similar to those observed in natural marshes.

Initial nitrogen pools reflect past land use. Agricultural sites are increasingly being converted into wetlands, resulting in many restored or created wetlands with large initial nitrogen pools due to legacy effects of fertilizer use. Despite increased nitrogen availability, former agricultural sites still exhibit increased nitrogen demand during the vegetation development period, although areal nitrogen retention may be depressed (Ardón et al. 2010). Conversion of other terrestrial lands or restoration of former wetlands generally results in a small initial nitrogen pool relative to natural wetlands.

Rates of nitrification and denitirification are controlled by the microbial community and nitrogen loading. The total microbial biomass is limited by the availability of soil organic matter, in turn limiting the amount of nitrogen that can be transformed. Since nitrogen removal via denitrification is often desirable in restored and created wetlands, increasing the duration of inundation in restored or created wetlands may be used to encourage denitrification, but microbial community biomass often is still limited by available organic carbon. Rates of denitrification approach the rates of natural wetland after some buildup of soil organic matter in the soil, usually between 5 and 15 years after creation or restoration (Craft et al. 2002).

Equivalency of soil nitrogen pools between created or restored wetlands and natural wetlands may take decades. The buildup of soil nitrogen pools is slowed by denitrification losses of nitrogen to the atmosphere and relies on sequestering organic nitrogen in soil organic matter. Equivalency of soil nitrogen pools in restored or created wetlands with natural wetlands requires approximately 30–40 years (Craft et al. 2003).

Unlike carbon and nitrogen, phosphorous cycling in wetlands is primarily controlled by abiotic processes. Phosphorous inputs into wetlands are dissolved in surface water in both inorganic and organic forms or bound to sedimentary particles. Retention of soil-bound particulate phosphorus is controlled by sediment deposition, a process closely tied to hydrology. Initially, sediment deposition in restored or created wetlands is higher than in natural wetlands. This is partly due to the establishment of vascular vegetation which alters surface water hydrology, slowing flowing waters and allowing more suspended particles to deposit. Over 1–3 years, the rates of sedimentation of slow and available binding sites for phosphorous are reduced, slowing the rate of phosphorous sequestration. Long-term maximum rates of phosphorus retention across wetland types are around 1 g P m^{-2} $year^{-1}$ (Richardson and Qian 1999).

Retention of soluble phosphorus involves several geochemical and biological mechanisms. Geochemical mechanisms include complexation of phosphorus with iron, aluminum, and calcium present in wetland soils, all of which are characteristics that are dependent on the parent soil type. Increasing water residence time and depth have been shown to increase soil phosphorus sorption (Hansson et al. 2005). Biotic uptake from vascular vegetation, microbes, and periphyton occur, but are responsible for relatively little long-term storage. In systems with high-surface water, algal productivity can further increase phosphorous retention by altering the carbonate equilibrium causing phosphorous precipitation (Vymazal 2007). In these systems, high algal productivity can deplete aqueous CO_2 concentrations, increasing the pH causing precipitation of calcium phosphate.

Restored or constructed wetland phosphorus release may occur in sites that initially contain elevated phosphorous levels in the soil, such as sites with agricultural legacies. Flooding a previously drained soil may release phosphorus to surface waters for 1–2 years after flooding (Aldous et al. 2005). Additional phosphorus loss can occur over the lifetime of the wetland if the wetland experiences regular cycles of wetting and drying, due to either microbial cell lysis following osmotic shock or release of soil-bound phosphorus due to ion exchange (Chacón et al. 2005).

When properly restored, nutrient cycling functions of created and restored wetlands develops relatively quickly. Comparison of C sequestration and N and P removal by created and restored wetlands suggests that C sequestration and N and P burial in soil develop instantaneously with reintroduction of hydrology and establishment of good vegetative cover (Fig. 1). For example, after 1 year, created tidal salt marshes sequestered more C than natural reference salt marshes

Table 1 Carbon sequestration and N and P burial in soils of created and natural salt marshes (Table constructed from data in Craft et al. 2003)

	Age	Carbon	Nitrogen	Phosphorous
Constructed marsh	1	99	12.5	5
	3	39	5.5	0
	8	27	1.9	0.6
	11	18	1.3	0
	13	62	5.7	0.8
	24	21	1.9	0
	26	34	2.7	0
	28	39	2.6	0.2
Mean	–	42	4.3	0.8
Natural marshes	>100	40–43	2.8–3.0	0.5–0.6

[1]Mean of eight natural references marshes based ^{137}Cs and ^{210}Pb dating of soil cores

(Table 1). Nitrogen and P accumulation in soil also was greater. As the marshes aged, rates of C sequestration and nutrient accumulation slowed to levels consistent with natural salt marshes in the region. The predictable hydrology of tidal inundation was key to the rapid development of nutrient cycling functions of these marshes. However, this may not always be the case for wetland restoration projects, especially those where hydrology is less predictable such as precipitation-driven wetlands.

The Effects of Marsh Restoration Techniques

Many of the techniques used in the restoration and creation of marshes may accelerate the development of wetland function or structure. These techniques primarily take the form of hydrological design, soil amendments, and vegetative plantings, all of which are applied prior to the reestablishment of wetland hydrology.

Amendments generally take the form of added carbon or nutrients in an attempt to more rapidly develop processes that are dependent on carbon or nutrient pools. Amending soils is usually performed by casting material onto the wetland soils prior to the establishment of vegetation. Carbon amendments include compost, sawdust, or other high-carbon material which drive the development of soil microbial communities in wetlands. Increased microbial activity increases the availability of both nitrogen and phosphorous in wetland soils and accelerates rates of denitrification (Sutton-Grier et al. 2009). Despite increased availability of nitrogen and phosphorous, vegetation establishment is not significantly affected by carbon amendments alone. Nitrogen and phosphorus amendments increase soil fertility by ameliorating low-nutrient conditions in the soils (Zedler 2000). Adding nutrients accelerates the development of vascular vegetation biomass and the denitrifying soil community.

Mobilization of soil phosphorus from flooding may be ameliorated via several amendments including calcite, dolomite, alum, or iron chloride which bind soluble forms of phosphorous (Ann et al. 1999). The amount of each amendment needed increases with the amount of soil organic matter present due to complexation with amendment cations. While ultimately effective at preventing remobilization of the initial pool of phosphorous, long-term effects on phosphorous soil cycling and pools are minimal.

Planting of wetland vegetation may accelerate the return of surface water hydrology and increase the initial rate of nitrogen accumulation in soils. Restored or created wetlands distant from seed sources or with little or no surface water connectivity cannot be colonized readily and exhibit reduced diversity which may persist decades after creation (Galatowitsch et al. 1999). Reduced vegetation diversity in wetlands with low nutrient inputs reduces both biomass and the rate of soil nitrogen accumulation (Callaway et al. 2003). Conversely, the highest rates of both biomass and nutrient retention occur in highly eutrophic systems with low species diversity. Depending on the nutrient load to the wetland, plantings may provide an opportunity to more rapidly establish carbon and nutrient cycling in created or restored wetlands.

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Section XX

Environmental Impact Assessment for Wetlands

Roel Slootweg

Environmental Impact Assessment for Wetlands: Overview

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Roel Slootweg

Contents

Abstract

Environmental Impact Assessment (EIA) is a process of evaluating the likely environmental impacts of a proposed project or development, taking into account interrelated socioeconomic, cultural, and human-health impacts. The main objectives of EIA are to incorporate environmental considerations into the decision-making process, and to minimize or offset the adverse effects of development proposals. EIA initially developed in the USA in the late 1960s in response to growing concern about environmental degradation and was later adopted more widely by various international agencies and the EU. EIA consists of various stages: screening, scoping, impact assessment and evaluation, reporting, review, decision-making, and monitoring/enforcement. The effectiveness of EIA is constrained by the capacity of the organizations responsible for its implementation, which varies in relation to the political and governance context of different countries. The importance of EIA for wetlands is recognized by important international conventions such as the Convention on Biological Diversity (CBD) and the Ramsar Convention on Wetlands.

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Environmental impact assessment · Sustainable development · Decision-making · Envirionmental policy · Governance

Introduction

Environmental Impact Assessment (EIA) is a process of evaluating the likely environmental impacts of a proposed project or development, taking into account interrelated socioeconomic, cultural, and human-health impacts, both beneficial and adverse. Participation of relevant stakeholders, including indigenous and local communities, is considered as a precondition for a successful EIA.

Wetlands are complex and dynamic ecosystems where human interventions may result in significant changes of the system. Wetlands have throughout history provided humanity with important ecosystem services, ranging from harvestable food and water, to flood control and coastal protection, and more recently to climate change mitigation and adaptation. The demand for these services changes over time but definitely increases with increasing population and development pressure, often leading to unsustainable and undesirable changes in wetland systems. The maximization of one ecosystem service, for example, provision of irrigation water by damming a river, may go at the cost of many other services (e.g., fish productivity, prevention of saline water intrusion, sediment transport, etc.). EIA is an instrument to predict the potential negative and positive consequences of development proposals, prior to construction. Good EIA thus is in the interest of wetlands, wetland managers, and people depending on wetlands.

EIA has been around since the end of the 1960s and is practiced in some form or other in most countries. The principle behind environmental assessment is deceptively simple: it directs decision makers to "look before they leap." An EIA should bring into focus what the likely environmental effects of a project or plan could be, before decisions on that project or plan are made. With a clear insight into the environmental consequences, decision makers are in a better position to direct development towards a more sustainable course.

EIA is defined as "the process of identifying, predicting, evaluating and mitigating the biophysical, social, and other relevant effects of development proposals prior to major decisions being taken and commitments made" (IAIA 1999). The objectives of "good practice" EIA are:

- To ensure that environmental considerations are explicitly addressed and incorporated into the development decision-making process in a participatory way
- To anticipate and avoid, minimize, or offset the adverse significant biophysical, social, and other relevant effects of development proposals
- To protect the productivity and capacity of natural systems and the ecological processes which maintain their functions
- To promote development that is sustainable and optimizes resource use and management opportunities

Of course, decision makers do not direct development on their own. Most plans or projects concern a range of actors, from government to the business sector and the public arena. For this reason, EIA does not merely provide information but brings the various parties together to discuss this information, come to a shared understanding of the possible effects, and determine what this knowledge should mean for the project at hand.

This overview entry describes the origins and early development of EIA briefly, sets out some basic concepts, and identifies principles of best practice drawn from practice and research into the effectiveness of EIA. Subsequently, recent trends in thinking and application will be discussed. This entry provides general views on EIA and its relevance for wetlands. In subentries, the procedural steps of EIA are elaborated further from a wetlands perspective. The closely related fields of social impact assessment, health impact assessment, and public participation are introduced in separate subentries.

Origins and Early Development of EIA

The history of EIA can be traced back to the 1969 US National Environmental Policy Act (NEPA) which is credited with first institutionalizing EIA, in response to growing concern about environmental degradation. Several other countries followed suit in the 1970s, predominantly those in the western world. Then, in the 1980s, EIA application began to spread. The European Union instituted EIA legislation in member states, and EIA became part of the World Bank's operating procedures. In the 1990s, other international finance institutes such as the Asian Development Bank and the European Bank for Reconstruction and Development adopted EIA, and it became legally embedded in developing countries as well. By 1997, over 100 countries had an EIA system in place (Wood 2003; Sadler 1996).

The emphasis of early EIA in industrialized countries was on the biophysical environment. In recent years, this narrow biophysical focus has evolved into a broader scope with inclusion of different types of effects such as economic and social impacts, the assessment of effects on human health (health impact assessment), transboundary effects, and applications in postconflict or postdisaster situations.

Generally Accepted Procedural Framework for EIA, with Existing Variations

Although legislation and practice vary around the world, the fundamental components of an EIA involve the following stages (see Fig. 1):

1. **Screening,** to determine which proposals should be subject to EIA, to exclude those unlikely to have harmful environmental impacts, and to indicate the level of assessment required. Types of existing screening mechanisms include:

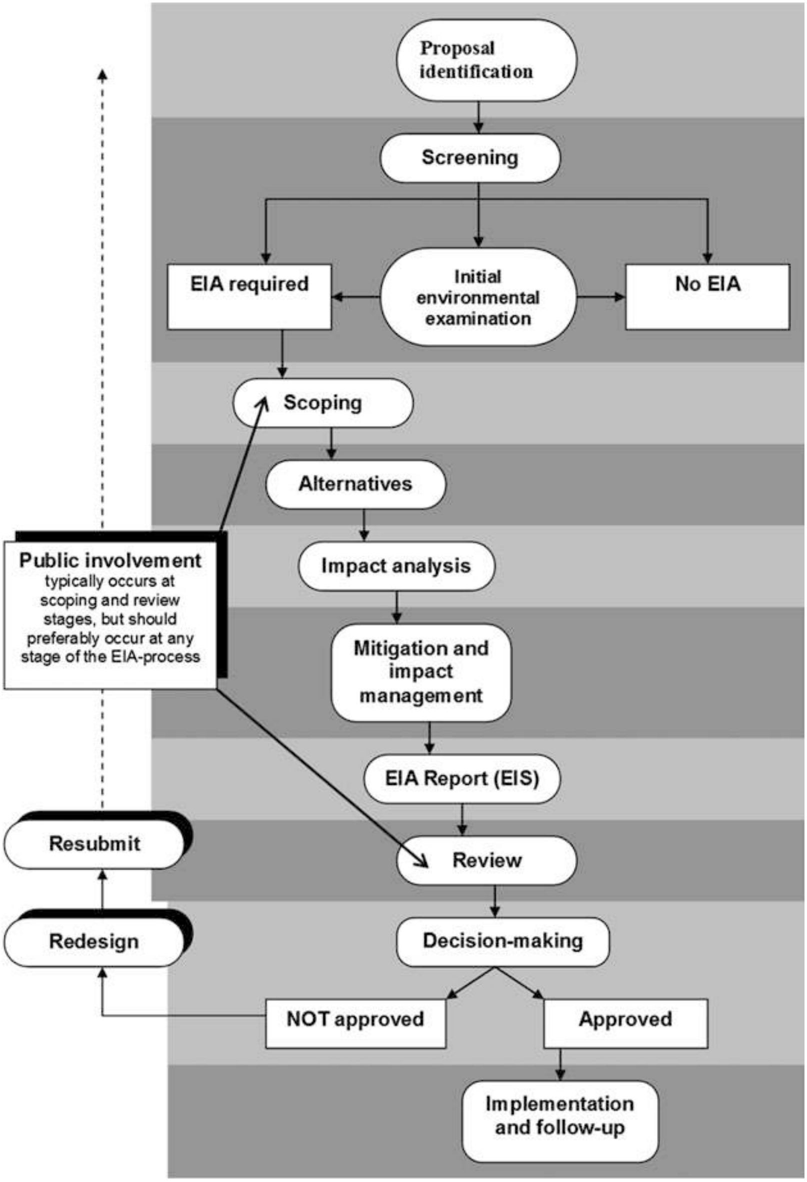

Fig. 1 Generalized EIA process flowchart (figure from Slootweg et al. 2009; used with permission from Cambridge University Press)

(i) Positive lists (or inclusion lists), identifying projects requiring EIA. A disadvantage of this approach is that the impact of projects varies substantially depending on the nature of the receiving environment, which is not taken into account.

(ii) Negative lists (or exclusion lists), identifying those projects not subject to EIA. A few countries use (or have used) this approach.
(iii) Lists of sensitive geographical areas in which projects would require EIA. The advantage of this approach is that the emphasis is on the sensitivity of the receiving environment rather than on the type of project.
(iv) Expert judgment (sometimes referred to as initial environmental examination or preliminary environmental assessment), with or without a limited study.
(v) A combination of a list and expert judgment to determine the need for an EIA. Screening criteria are usually defined by law; a screening decision whether a project needs to be subjected to an EIA is, in countries with formal EIA legislation, a government responsibility.

2. **Scoping,** to identify which potential impacts are relevant to assess based on legislative requirements, international conventions, expert knowledge, and public involvement. Scoping can also identify alternative solutions that avoid, mitigate, or compensate adverse impacts on biodiversity, including the options of not proceeding with the development, finding alternative designs or sites which avoid the impacts, incorporating safeguards in the design of the project, or providing compensation for adverse impacts. Deriving terms of reference for the impact assessment study is also done at this stage of the EIA process. Scoping also enables the competent authority (or EIA professionals in countries where scoping is voluntary) to:

(i) Guide study teams on significant issues and alternatives to be assessed, clarify how they should be examined (methods of prediction and analysis, depth of analysis) and according to which guidelines and criteria
(ii) Provide an opportunity for stakeholders to have their interests taken into account in the EIA
(iii) Ensure that the resulting Environmental Impact Statement (or environmental impact assessment report) is useful to the decision maker and is understandable to the publicScoping, including public hearing, is usually (but not always) done by the project proponent, who submits a scoping document to the competent authority for review.

3. **Assessment and evaluation of impacts,** the phase to describe and possibly quantify the likely environmental impacts of a proposed project or development, including the detailed elaboration of alternatives. EIA should be an iterative process of assessing impacts, redesigning alternatives, and making comparisons. Assessing impacts usually involves a detailed analysis of their nature, magnitude, extent and duration (e.g., on flora, fauna, soil, water, air), and a judgment of their significance, i.e., whether the impacts are acceptable to stakeholders and society

as a whole, require mitigation and/or compensation, or do not comply with formal norms or standards. The main tasks of impact analysis and assessment are:

(i) Refinement of the understanding of the nature of the potential impacts identified during screening and scoping and described in the terms of reference. This includes the identification of indirect and cumulative impacts and of the likely cause–effect chains.
(ii) Review and redesign of alternatives; consideration of mitigation and enhancement measures, as well as compensation of residual impacts; planning of impact management; evaluation of impacts; and comparison of the alternatives.
(iii) Identification of the remaining gaps in knowledge and information and an assessment of risks due to these gaps.
(iv) Reporting of study results in an environmental impact statement (EIS) or EIA report.The actual assessment study, including reporting, is a responsibility of the project proponent. Usually the proponent hires a consultant to do the work.

4. **Reporting,** the EIS or EIA report consists of (1) a technical report with annexes; (2) an environmental management plan (EMP), providing detailed information on how measures to avoid, mitigate, or compensate expected impacts are to be implemented, managed, and monitored; and (3) a nontechnical summary for the general public. The EIS is designed to support various stakeholder groups:

(i) The proponent: to plan, design, and implement the proposal in a way that eliminates or minimizes the negative effect on the biophysical and socioeconomic environments and maximizes the benefits to all parties in the most cost-effective manner.
(ii) The Government or responsible authority: to decide whether a proposal should be approved and the terms and conditions that should be applied.
(iii) The general public: to understand the proposal and its impacts on the community and environment, and provide an opportunity to share comments on the proposed action with decision makers. Some adverse impacts may be wide-ranging and have effects beyond the limits of particular habitats/ecosystems or national boundaries. Therefore, environmental management plans and strategies contained in the EIS should consider regional and transboundary impacts.

5. **Review** of the EIS. The purpose of the review is to ensure that the information for decision makers is sufficient, focused on the key issues, and scientifically and technically accurate. In addition, the review should evaluate whether:

(i) The likely impacts would be environmentally acceptable.
(ii) The design complies with relevant standards and policies or standards of good practice where official standards do not exist.
(iii) All of the relevant impacts, including indirect and cumulative impacts, of a proposed activity have been identified and adequately addressed in the EIA.
(iv) The concerns and comments of all stakeholders are adequately considered and included in the final report presented to decision makers. The process establishes local ownership of the proposal and promotes a better understanding of relevant issues and concerns.The effectiveness of the review process depends on the quality of the terms of reference defining the issues to be included in the study and approval by the authority. Scoping and review are therefore complementary stages. Review is a responsibility of the competent authority.

6. **Decision-making** on whether to formally approve the project or not and under what conditions. Decision-making takes place throughout the EIA process in an incremental way from the screening and scoping stages to decisions during data-collecting and analysis, and impact prediction, to making choices between alternatives and mitigation measures, and finally the decision to refuse or authorize the project with possible conditions.Prior to the decision on the project, in most EIA systems a decision on the acceptability of the Environmental Impact Statement is made. In most countries this is a separate decision; the quality of the EIS may be good, even if the project is creating lots of problems. A decision on the project is a political choice, sometimes going against the recommendations of the EIS. However, in others countries a project cannot be accepted if the EIA has not resulted in an environmental license. Project proponent and sector ministries in such systems may put heavy pressure on the environmental authorities to provide a license to projects, even if they have serious impacts.

7. **Monitoring, compliance, enforcement, and environmental auditing**. These activities, commonly grouped under the heading of "EIA follow-up," ensure that the recommendations from EIS or EMP are implemented. Roles and responsibilities with respect to these are variable and depend on regulatory frameworks and performance by the responsible organizations.

In theory, EIA is a tool with an enormous potential to influence decision-making. This potential is realized maximally when the good practice principles developed by IAIA (1999) are fully applied. A minimum variant as such is not described in the literature, but a number of basic conditions should be fulfilled in order to justify the term EIA as a decision support tool. The EIA directive of the EU (European Council 1985, 1997) can be considered as an example of the minimum variant. EU member countries can build on this directive to develop their own more ambitious EIA regulatory frameworks. Table 1 lists the minimum and maximum ambition level

Table 1 Minimum and maximum ambition levels in EIA, simplified

Aspects of EIA system	Minimum EIA variant	Maximum EIA variant
Objectives	Environmental protection	Environmental protection, sustainable development, well-informed and participatory decision-making
Scope of study	Environmental aspects	Environmental, social/health, and economic aspects
Environmental Protection	Mitigation measures	Alternatives, mitigating and compensatory measures
Involvement of civil society	Low	Ambitious
Transparency, accountability	Limited	Complete
Quality assurance mechanism	Limited	Advanced

Source: Table compiled from EU EIA directive (European Council 1985) for minimum variant and IAIA (1999) for maximum variant

for various aspects of the EIA system. In principle, the EIA systems of all countries and institutes in the world can be positioned in this table. A country can have different ambition levels for different aspects.

Effectiveness of EIA

An international landmark study on the effectiveness of EIA, based mainly on EIA in industrialized western countries (Sadler 1996), concluded that EIA was effective when it improved project design and site selection, when it led to more informed decision-making and more environmentally sensitive decisions, when it increased accountability and transparency during the development process, when it improved the integration of projects into their environmental and social setting and reduced environmental damage, when it helped projects meet their financial and socioeconomic objectives, and when it contributed positively towards achieving a more sustainable development.

Three effects of the EIA process influence the design of project proposals (Heuvelhof and Nauta 1996; Chistensen et al. 2003). The *dialogue and transparency* effect, arising from the combination of dialogue between proponent and government with public participation from the formal start of the procedure until final decision-making and licensing, was considered to have the most important effect on project design. The *prevention effect* occurs when the EIA forces proponents to adapt their projects plans prior to the formal start of the EIA process. Sometimes a proponent decides to abandon a project if it becomes clear that environmental standards will not be met. The prevention effect only occurs in countries with effective regulation and enforcement. The third effect is the *conditionality effect*, which occurs as a result of the EIA when a license is conditional to the implementation of alternative development options or mitigation measures.

Commonly reported reasons for the poor implementation of EIA are limited capacities of organizations responsible for the EIA process and a lack of respect among decision makers for their autonomy; poor compliance with EIA recommendations because of weak regulatory EIA frameworks and weak enforcement, exemplified, e.g., by EIAs executed when decisions have been taken already; limited consideration of alternative project options; and poor community participation with a lack of information disclosure and weak public consultation.

EIA has great potential to contribute to well-informed decision-making and protection of the environment. Currently this potential is achieved in a number of (predominantly) high-income, democratic countries. In many other countries there is a strong belief in the potential of EIA and the willingness to improve its effectiveness. Despite examples of decision makers who are not genuinely interested in accountable and transparent decision-making, EIA is considered a powerful environmental management tool all over the world. The effectiveness of EIA can be regarded as a continuum with full realization of the potential at one extreme and no realization of the potential at the other.

State of the Art: What Is Needed for Effective EIA?

The EIA system can be defined as the organizational and administrative structure to implement EIA (Espinoza and Alzina 2001). The question arises: what factors contribute to a successful EIA system, i.e., successful application and implementation of EIA. There is a growing insight that the national context such as the political system and the socioeconomic structure of a society has a major influence on EIA implementation and compliance. EIA has a legal basis in the majority of countries but seems to meet its objectives predominantly in high-income democracies. Twenty years of capacity development programs have demonstrated that copying successful EIA systems to other countries does not work or can be even counterproductive (Cherp and Antypas 2003). Three main factors seem to be of major importance.

First, the strength of the EIA system itself. The availability of scientifically sound information is a requirement for good quality EIA studies and consequently well-informed decision-making. Ideally, information should be gathered and analyzed systematically and made accessible for third parties such as civil society groups. When this requirement is not or only partly fulfilled, the execution of EIA studies will be hampered, and EIA cannot achieve its full potential as a tool to provide environmental information for decision-making. This also illustrates the linkages between the EIA system and its context, in this case the knowledge infrastructure in a country (Kolhoff et al. 2009).

The second important factor is the capacity of the environmental compliance system, i.e., the institutions making sure that actors comply with environmental standards and regulations by means of, among others, monitoring (inspections, audits), assistance, and penalties. Whether the conditions set in the environmental license are met in practice depends on the performance of the environmental compliance system, which is related to the capacities of the responsible organizations (e.g., government bodies such as pollution control boards, environmental protection agencies, etc.), the performance of the proponent of the project for which EIA has been carried out, and external factors such as influential business agents and politicians. In many cases, capacity development efforts emphasize the EIA capacity needs and largely neglect the environmental compliance system. Capacities of both systems should be developed in parallel.

Finally, the country-specific context in which an EIA system operates is important. The latter determines the enabling environment for the first two factors by influencing the opportunities and constraints of EIA and environmental compliance system performance. The checks and balances in a political system determine to what extent the formal autonomy of EIA and implementing organizations is respected. In countries where the division of powers between legislative, executive, and judiciary is weak there is significant risk of corruption undermining the role of EIA. The ability and capacity of a society to act as a countervailing power is determined to a great extent by the political system. Even when political systems become more transparent it can take a generation before society is able to participate effectively in EIA (Cherp 2001; Purnama 2003).

Wetlands and EIA

Several international treaties have incorporated directives about EIA and wetlands. Article 14 of the Convention on Biological Diversity (CBD) requires its Contracting Parties to introduce appropriate procedures for EIA in proposals that might have effects on biological diversity, and to provide mechanisms for taking the biodiversity impacts of programs and policies into account. Some other parts of the Convention may be read as implying a requirement for impact assessment, such as Article 3 which seeks to ensure that activities within one country's jurisdiction do not cause damage to another country.

Such "implied" EIA requirements can also be found in Article 3.2 of the Ramsar Convention on Wetlands, which requires its Contracting Parties to "arrange to be informed at the earliest possible time if the ecological character of any wetland in its territory has changed, is changing or is likely to change as the result of technological developments, pollution or other human interference." This implies a need for the ability to anticipate and predict the effects of actions on wetland ecosystems and, arguably, a need to go through an EIA process.

The Ramsar Convention has long recognized the importance of applying impact assessment techniques to situations where the ecological character of Ramsar sites and other wetlands may be threatened by development projects or broader policies and strategies (Ramsar Convention 2010). The Convention has adopted several Recommendations and Resolutions (notably Recommendation 6.2 in 1996 and Resolution VII.16 in 1999) which call upon Parties to incorporate impact assessment into legislative frameworks and ensure that impact assessments are undertaken where appropriate. In 2006, the CBD's Convention of Parties endorsed "Voluntary guidelines on biodiversity-inclusive impact assessment" which provide an elaboration on whether, when, and how to consider biodiversity in both project-level and strategic-level impact assessments. These guidelines were adopted by the Ramsar Convention in 2008 (Resolution X.17 at COP10), reproducing the CBD Guidelines together with additional observations on their applicability to wetlands.

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Environmental Impact Assessment for Wetlands: Screening

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Roel Slootweg

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Abstract

In Environmental Impact Assessment (EIA), screening is used to determine which projects should be subjected to EIA. For this, screening criteria are used. Biodiversity criteria are e.g. the extent to which the proposed activity increases the risk of extinction of certain species, disturbs the ecosystem or surpasses sustainable harvesting levels. For wetlands, the appropriate impact scale is larger than just the wetland itself; it should consider at least the catchment area. Use can be made of existing references for wetland species published by e.g. Wetlands International or IUCN. The outcome of the screening phase is a screening decision which specifies if the project may be pursued, and if an EIA is required. Wetlands-inclusive screening for EIA can include specific steps such as a biodiversity screening map, a list of activities for which and EIA is required, and threshold values for screening, e.g. of drivers of change.

Keywords

Environmental impact assessment · Screening · Decision-making · Screening criteria · Screening decision · Biodiversity

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Introduction

Screening is used to determine which proposals should be subject to Environmental Impact Assessment (EIA), to exclude those unlikely to have harmful environmental impacts and to indicate the level of assessment required. The outcome of the screening process is a screening decision. Guidelines for EIA, mostly related to biodiversity, were given by the Convention on Biological Diversity (CBD 2006; Slootweg et al. 2006) and additional information on wetlands was provided by Ramsar Convention Handbook 16 on Impact Assessment (Ramsar Convention 2010).

Wetlands in screening

Since legal requirements for EIA may not guarantee that biodiversity will be taken into account, consideration should be given to incorporating biodiversity criteria, including those with relevance to wetlands, into existing, or the development of new, screening criteria. Important information for developing screening criteria can be found in national biodiversity strategies and action plans (NBSAPs) or national wetlands policies. These strategies provide detailed information on conservation priorities and on types and conservation status of ecosystems. Furthermore they describe trends and threats at ecosystem as well as species level and provide an overview of planned conservation activities.

Taking into account the three objectives of the Convention on Biological Diversity, fundamental questions which need to be answered in an EIA from a biodiversity perspective include:

(a) Would the intended activity affect the biophysical environment directly or indirectly in such a manner or cause such biological changes that it will increase risks of extinction of genotypes, cultivars, varieties, populations of species, or the chance of loss of habitats or ecosystems?
(b) Would the intended activity surpass the maximum sustainable yield, the carrying capacity of a habitat/ecosystem or the maximum allowable disturbance level of a resource, population, or ecosystem, taking into account the full spectrum of values of that resource, population or ecosystem?
(c) Would the intended activity result in changes to the access to, and/or rights over biological resources?

In a wetland context, the appropriate spatial scale at which to think about impacts may be wider than the wetland ecosystem itself. In particular, the river basin (water catchment) is an important scale at which to address aspects of wetland-related impacts. Also, where impacts on particularly important species such as migratory fish or birds, are at stake, assessment at the scale of the migratory range (flyway) of the relevant populations will be very relevant. This may involve a chain of perhaps disjunct ecosystems, and therefore may need to take a broader perspective than would normally be the case under the ecosystem approach.

At the species diversity level, references to 'a population of a species' should include wetland species and migratory species. As a reference for populations, for waterbirds appropriate biogeographical populations are established in Wetlands International's periodically published Waterbird Population Estimates. For other taxa, population information regularly updated by IUCN's Specialist Groups though the IUCN Species Information Service (SIS) and published in the Ramsar Technical Report series should be used. Where a site regularly supports >1% of one or more populations of waterbirds or other wetland-dependent animal species, an additional question could be: would the intended activity threaten to cause a direct or indirect loss of the international importance of these interests at the site?

Screening Decision

A screening decision defines the appropriate level of assessment. The result of a screening decision can be that:

- The proposed project is "fatally flawed" in that it would be inconsistent with international or national conventions, policies or laws. It is advisable not to pursue the proposed project. Should the proponent wish to proceed at his/her risk, an EIA would be required;
- An EIA is required (often referred to as category A projects);
- A limited environmental study is sufficient because only limited environmental impacts are expected; the screening decision is based on a set of criteria with quantitative benchmarks or threshold values (often referred to as category B projects);
- There is still uncertainty whether an EIA is required and an initial environmental examination has to be conducted to determine whether a project requires EIA or not; or
- The project does not require an EIA.

Design of Wetlands-Inclusive Screening

Biodiversity-inclusive screening criteria set out circumstances in which EIA is justified on the basis of biodiversity considerations. They may relate tocategories of activities known to cause biodiversity impacts, including thresholds referring to size of the intervention area and/or magnitude, duration and frequency of the activity. Screening criteria can also relate to the magnitude of biophysical change that is caused by the activity, or consist of maps indicating areas important for biodiversity, often with their legal status.

A suggested approach to the development of biodiversity-inclusive screening criteria, combining the above types of criteria, starts with the design of a biodiversity screening map indicating areas in which EIA is required (Step 1). Then, the activities for which EIA is required are defined (Step 2), and threshold values set to distinguish between full, limited/undecided or no EIA (Step 3). This approach takes account of biodiversity values (including valued ecosystem services) and activities that might impact drivers of change of biodiversity. If possible, biodiversity-inclusive screening criteria should be integrated with the development (or revision) of a national biodiversity strategy and action plan or for national wetland policies. This process can generate valuable information such as a national spatial biodiversity assessment, including conservation priorities and targets, which can guide the further development of EIA screening criteria.

Step 1: Biodiversity Screening Map

According to the principles of the ecosystem approach, a biodiversity screening map is designed, indicating important ecosystem services. The map is based on expert judgement and has to be formally approved. Suggested categories of geographically defined areas, related to important ecosystem services, are:

- Areas with important regulating ecosystem services in terms of maintaining biodiversity:
  - Protected areas: depending on the legal provisions in a country these may be defined as areas in which no human intervention is allowed, or as areas where impact assessment at an appropriate level of detail is always required;
  - Areas containing threatened ecosystems outside of formally protected areas, where certain classes of activities (see Step 2) would always require an impact assessment at an appropriate level of detail;
  - Areas identified as being important for the maintenance of key ecological or evolutionary processes, where certain classes of activities (see Step 2) would always require an impact assessment at an appropriate level of detail;
  - Areas known to be habitat for threatened species, which would always require an impact assessment at an appropriate level of detail.
- Areas with important regulating ecosystem services for maintaining natural processes with regard to soil, water, or air, where impact assessment at an appropriate level of detail is always required. Examples are wetlands, highly erodable or mobile soils protected by vegetation (e.g., steep slopes, dune fields), forested areas, coastal or offshore buffer areas;
- Areas with important provisioning ecosystem services,where impact assessment at an appropriate level of detail is always required. Examples are extractive reserves, lands and waters traditionally occupied or used by indigenous and local communities, fish breeding grounds;
- Areas with important cultural services, where impact assessment at an appropriate level of detail is always required. Examples are scenic landscapes, heritage sites, sacred sites;
- Areas with other relevant ecosystem services (such as flood storage areas, groundwater recharge areas, catchment areas, areas with valued landscape quality, etc.); the need for impact assessment and/or the level of assessment is to be determined (depending on the screening system in place);

- All other areas: no impact assessment required from a biodiversity perspective (an EIA may still be required for other reasons).

Step 2: Activities for Which Impact Assessment is Required

Define activities for which impact assessment may be required from a biodiversity perspective. The activities are characterised by the following direct drivers of change:

- Change of land-use or land cover, and underground extraction: above a defined area affected, EIA always required, regardless of the location of the activity - define thresholds for level of assessment in terms of surface (or underground) area affected;
- Change in the use of marine and/or coastal ecosystems, and extraction of seabed resources: above a defined area affected, EIA always required, regardless of the location of the activity – define thresholds for level of assessment in terms of surface (or underground) area affected;
- Fragmentation, usually related to linear infrastructure. Above a defined length, EIA always required, regardless of the location of the activity – define thresholds for level of assessment in terms of the length of the proposed infrastructural works;
- Emissions, effluents or other chemical, thermal, radiation or noise emissions - relate level of assessment to the ecosystem services map;
- Introduction or removal of species, changes to ecosystem composition, ecosystem structure, or key ecosystem processes responsible for the maintenance of ecosystems and ecosystem services (see appendix 2 below for an indicative listing) - relate level of assessment to ecosystem services map.

It should be noted that these criteria only relate to biodiversity and serve as an add-on in situations where biodiversity has not been fully covered by the existing screening criteria.

Step 3: Threshold Values for Screening

Determining norms or threshold values for screening is partly a technical and partly a political process, the outcome of which may vary between countries and ecosystems. The technical process should at least provide a description of:

1. Categories of activities that create direct drivers of change (extraction, harvest or removal of species, change in land-use or cover, fragmentation and isolation, external inputs such as emissions, effluents, or other chemical, radiation, thermal or noise emissions, introduction of invasive alien species or genetically modified organisms, or change in ecosystem composition, structure or key processes), taking into account characteristics such as: type or nature of activity, magnitude,

extent/location, timing, duration, reversibility/irreversibility, irreplaceability, likelihood, and significance; possibility of interaction with other activities or impacts;

2. Where and when: the area of influence of these direct drivers of change can be modelled or predicted; the timing and duration of influence can be similarly defined;
3. Map of valued ecosystem services (including maintenance of biodiversity itself) on the basis of which decision makers can define levels of protection or conservation measures for each defined area. This map is the experts' input into the definition of categories on the biodiversity screening map referred to above under step 1.

In addressing the likelihood of effects and their relevance and significance for Ramsar-related values, reference should be made to Ramsar guidance on ecological character and risk assessment (see, e.g., Ramsar Convention 2002, 2005), and guidance on assessing the vulnerability of wetlands to change in their ecological character (Gitay et al. 2011).

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Environmental Impact Assessment for Wetlands: Scoping

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Roel Slootweg

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Abstract

In Environmental Impact Assessment (EIA), the scoping phase is used to define the focus of the EIA and to identify key issues to be studied in more detail. Scoping results in terms of reference for the EIA study and sets out the proposed approach and methodology. The steps of the scoping process can be followed in an iterative way depending on the information gathered and the project or ecosystem at hand. This includes a description of the proposed project, of the expected biophysical changes, and of possible alternative project plans; determination of the spatial and temporal extent of the changes and of effects on linked ecosystems and land use systems; assessment of effects on biodiversity and ecosystem services, and of possible measures to avoid or mitigate the impact; and identification of potential surveys needed to gather information. Many of the steps in the scoping process involve stakeholder participation. A number of practical recommendations for the scoping process is given.

Keywords

Environmental impact assessment · Scoping · Decision-making · Environmental policy · Governance

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Introduction

Scoping is used to define the focus of the impact assessment study and to identify key issues which should be studied in more detail. It is used to derive terms of reference (sometimes referred to as guidelines) for the EIA study and to set out the proposed approach and methodology. Scoping also enables the competent authority (or EIA professionals in countries where scoping is voluntary) to:

- Guide study teams on significant issues and alternatives to be assessed, clarify how they should be examined (methods of prediction and analysis, depth of analysis), and according to which guidelines and criteria
- Provide an opportunity for stakeholders to have their interests taken into account in the EIA
- Ensure that the resulting environmental impact statement is useful to the decision maker and is understandable to the public

During the scoping phase, promising alternatives can be identified for in-depth consideration during the EIA study. Guidelines for EIA, mostly related to biodiversity, were given by the Convention on Biological Diversity (CBD 2006; Slootweg et al. 2006) and additional information on wetlands was provided by Ramsar Convention Handbook 16 on Impact Assessment (Ramsar Convention 2010).

Steps of the Scoping Process and Information Needs

The following list presents the information that should be requested in the terms of reference of an impact study if the project screening suggests that the proposed activity is likely to have adverse impacts on biodiversity. This list represents a process of iterative steps. Scoping and impact study are two formal rounds of iteration; during the study, further iterative rounds may be needed, for example, when alternatives to the proposed project design have to be defined and assessed.

1. Description of the type of project, and of each project activity in terms of its nature, magnitude, location, timing, duration, and frequency.
2. Definition of possible alternatives, including "no net biodiversity loss" or "biodiversity restoration" alternatives (such alternatives may not be readily identifiable at the outset of impact study, and one would need to go through the impact study to determine such alternatives). Alternatives include location alternatives, scale alternatives, siting or layout alternatives, and/or technology alternatives.
3. Description of expected biophysical changes (in soil, water, air, flora, fauna) resulting from proposed activities or induced by any socioeconomic changes caused by the activity.
4. Determination of the spatial and temporal scale of influence of each biophysical change, identifying effects on connectivity between ecosystems, and potential

cumulative effects. In a wetland context, the appropriate spatial scale at which to think about impacts may sometimes be a particularly broad-scale interpretation of "ecosystem." In particular, the river basin (water catchment) is an important scale at which to address aspects of wetland-related impacts. Also, where impacts on particularly important species such as migratory fish or birds are at stake, assessment at the scale of the migratory range (flyway) of the relevant populations will be very important. This may involve a chain of ecosystems (perhaps disjunct ones) and therefore may need to take a broader perspective than would normally be the case under the ecosystem approach.

5. Description of ecosystems and land-use types lying within the range of influence of biophysical changes.
6. Assessment, for each of these ecosystems or land-use types, of the likelihood of biophysical changes to have adverse impacts on biodiversity in terms of composition, structure (spatial and temporal), and key processes. Indication of the certainty level of predictions, and of potential mitigation measures, irreversible impacts, and irreplaceable loss.
7. For the affected areas, information on baseline conditions and anticipated trends in biodiversity in the absence of the proposal. In the case of Ramsar sites, the "baseline" relates to the site's ecological character. Hence the baseline should be the target condition (ecological character) described in the management plan objectives. It will therefore not necessarily equate to the condition of the site described at the time of listing (or subsequent updating of the Ramsar Information Sheet) unless at such times the site happens to have achieved its optimal (target) condition or if there is no better baseline available.
8. Identification, in consultation with stakeholders, of the current and potential ecosystem services provided by the affected ecosystems or land-use types and the values of these functions for society. Indication of the main beneficiaries and those adversely affected from an ecosystem services perspective, focusing on vulnerable stakeholders.
9. Identification of the ecosystem services that will be significantly affected by the proposed project, with confidence levels in predictions, and taking into account mitigation measures, irreversible impacts, and irreplaceable loss.
10. Definition of possible measures to avoid, minimize, or compensate for significant damage to, or loss of, biodiversity and/or ecosystem services; definition of possibilities to enhance biodiversity, with reference to any legal requirements.
11. Definition of the importance of expected impacts for the alternatives considered (in consultation with stakeholders). The importance of expected impacts is related to a reference situation, which may be the existing situation, a historical situation, a probable future situation (e.g., the "without project" or "autonomous development" situation), or an external reference situation. When determining importance (weight), geographic importance of each residual impact (e.g., impact of local/regional/national/ continental/global importance) is considered and its temporal dimension indicated.
12. Identification of necessary surveys to gather information required to support decision making and of important gaps in knowledge. It may be helpful to

consult with the National Focal Point for the Ramsar Scientific and Technical Review Panel in identifying these sources and gaps.

13. Provide details on required methodology and timescale.

Some Remarks on the Practice of Scoping in EIA

Not implementing a project may in some cases also have adverse effects on biodiversity. In rare cases, the adverse effects may be more significant than the impacts of a proposed activity (e.g., projects counteracting degradation processes). An analysis of current impact assessment practice has provided a number of practical recommendations when addressing biodiversity-related issues:

- Beyond the focus on protected species and protected areas, further attention needs to be given to: (i) sustainable use of ecosystem services, (ii) ecosystem level diversity, (iii) nonprotected biodiversity, and (iv) ecological processes and their spatial scale.
- The terms of reference should be unambiguous, specific, and compatible with the ecosystem approach; too often the terms of reference are too general and impractical.
- To provide a sound basis for assessing the significance of impacts, baseline conditions must be defined and understood and quantified where possible. Baseline conditions are dynamic, implying that present and expected future developments if the proposed project is not implemented (autonomous development) need to be included.
- Field surveys; quantitative data; meaningful analyses; and a broad, long-term perspective enabling cause-effect chains to be tracked in time and space are important elements when assessing biodiversity impacts. Potential indirect and cumulative impacts should be better assessed.
- Alternatives and/or mitigation measures must be identified and described in detail, including an analysis of their likely success and realistic potential to offset adverse project impacts.
- Guidance for scoping on biodiversity issues in EIA needs to be developed at the country level, but should, where appropriate, also consider regional aspects to prevent transboundary impacts. Concerning potential transboundary impacts, Ramsar Parties should regard Article 5 of the Convention and the Guidelines for international cooperation under the Ramsar Convention on Wetlands (Resolution VII.19).
- Guidance for determining levels of acceptable change to biodiversity needs to be developed at country level to facilitate decision-making.
- Guidance on assessing and evaluating impacts on ecosystem processes, rather than on composition or structure, needs to be developed at country level. The conservation of ecosystem processes, which support composition and structure, requires a significantly larger proportion of the landscape than is required to represent biodiversity composition and structure.

- Capacity development is needed to effectively represent biodiversity issues in the scoping stage; this will result in better guidelines for the EIA study.

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Environmental Impact Assessment for Wetlands: Avoidance, Minimization, Restoration, Compensation, and Offsets

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Susie Brownlie

Contents

Abstract

Wetlands provide a range of benefits to people through ecosystem services. Levels of dependence on wetlands vary, as do options for providing substitute services. Impacts of development projects or policies on wetlands can harm biodiversity. Impacts can also harm human well-being, if local communities, businesses, and/ or society at large depend – or are likely to depend in future – on the ecosystem services provided by the affected wetland. Mitigation of impacts is thus essential to safeguard these systems and their values to people; the full hierarchy of measures to mitigate harm should be used with emphasis on avoidance and minimization.

Keywords

Ecosystem services · Mitigation hierarchy · Offsets · Wetlands

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Introduction

Wetlands provide a range of benefits to people through ecosystem services. Levels of dependence on wetlands vary, as do options for providing substitute services. Broadly speaking, impacts of development projects or policies on wetlands can harm biodiversity, at the level of the particular type of wetland and its conservation status or at the level of species and their conservation status, as well as in other ecosystems that may depend on the wetland for their viability. Impacts can also harm human well-being, if local communities, businesses, and/ or society at large depend – or are likely to depend in future – on the ecosystem services provided by the affected wetland.

The purpose of mitigation in environmental assessment (EIA and SEA) is to look for ways to achieve the project objectives while avoiding significant negative impacts or reducing them to acceptable levels. The purpose of enhancement is to look for ways of optimizing environmental benefits. Both mitigation and enhancement of impacts should strive to ensure that the public or individuals do not bear costs which are greater than the benefits that accrue to them (Slootweg et al. 2006).

The Desired Outcome of Mitigation

Mitigation goals must be specified in terms of measurable performance standards. “No loss” or “no net loss” goals for mitigating impacts on wetlands predominate. The Ramsar Convention reflects a “no loss” approach to wetlands of international importance at a landscape or watershed scale. Although wetland impacts may be permitted, compensation is necessary to counterbalance these impacts “...at the level of the totality of the wetland resource” (Ramsar 2012). While compensation may be viewed as a form of “no net loss,” the primary duty is to avoid the need for compensation in the first place. A number of countries have adopted “no net loss” policies: impacts on wetlands would be permissible provided that mitigation measures – chiefly compensation – counterbalance those impacts. Mitigation should strive to ensure that affected parties are not left worse off as a result of development and that associated risks and rewards are fairly distributed.

The Mitigation Hierarchy

Negative impacts on wetlands can be mitigated (Fig. 1) by:

1. Avoiding or preventing impacts
2. Minimizing impacts by seeking alternatives such as, e.g., different location, siting, sequencing, phasing, scale, design, process, use of technology, and rigorous impact management including monitoring and adaptive or corrective action, as appropriate
3. Repairing, restoring, or rehabilitating damage

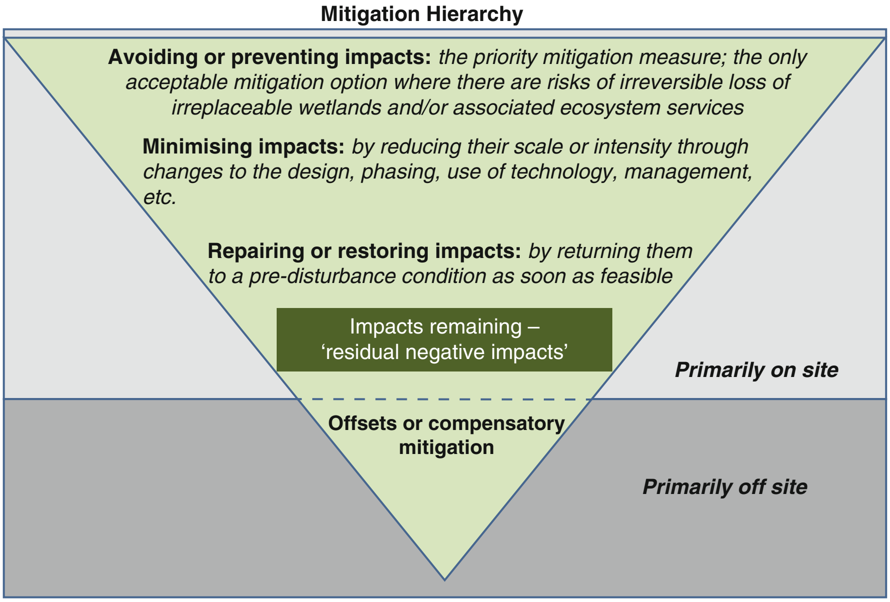

Fig. 1 Approaches for mitigation of impacts

4. Compensating or offsetting by replacing lost resources or providing substitute resources of equivalent value

This sequence of actions makes up the "mitigation hierarchy." Avoidance and minimization are preferred because prevention is better than cure. A "positive planning approach" is best suited to the early identification and avoidance of potentially significant impacts in project design. Compensation is used as a last resort measure. Successive levels of mitigation should only be chosen if an earlier level is not feasible to avoid compensation being inappropriately used as leverage for authorization. Compensation or offsets are not always possible (e.g., Pilgrim et al. 2013). There are cases where it is appropriate to reject a development proposal on grounds of irreplaceable loss of biodiversity and/or loss of ecosystem services on which people are heavily reliant and for which there are no suitable substitutes. Adherence to the mitigation hierarchy is "best practice," reflected in policies and performance standards worldwide (EC 2002; EPA 2008; BBOP 2012; IFC 2012; Ramsar 2012).

Compensation or Offsets

The "last resort" step in the mitigation hierarchy is compensation or offset (hereinafter referred to as "offset"), also known as "compensatory mitigation." Offsets are increasingly reflected in laws, policies, and practice around the world (e.g., Rajvanshi and Mathur 2010). Any offset must add value “beyond what would have happened otherwise” (Ramsar 2012). There are four ways to deliver offsets (e.g., Ramsar COP Resolution VII 24; EPA 2008), namely by:

1. Protecting or preserving an existing wetland
2. Enhancing an existing wetland by increasing one or more of its functions
3. Restoring a degraded or former wetland
4. Creating a new and similar wetland type from dry land or unvegetated water

Restoration of wetland areas is generally favored over creation given its relatively greater chance of success, while protection or preservation of an existing wetland is least favored as a form of offset (e.g., BBOP 2009). Wetland habitat can be enhanced by improving some wetland functions, e.g., repairing historical damage or reinstating diverted drainage (Rajvanshi et al. 2011). Enhancement of selected wetland functions may, however, be associated with a simultaneous decrease in other functions, so its potential as an offset would depend on the residual negative impacts being targeted. Offsets arguably assume that destroyed wetlands can be recreated (e.g., Matthews and Endress 2008). Although there are good examples of wetland offsets (e.g., Ten Kate et al. 2010), created and restored wetlands often fail (NRC 2001).

The issue of equivalence in the exchange of losses (impacts) and gains (offsets) is central to offsets. To be equitable, exchanges must be of the same type and amount and within a comparable timeframe and spatial scale (Salzman and Ruhl 2005). Similar types of wetland habitat in close proximity or in the same local watershed as the affected wetland should be targeted to ensure that gains through improved management or restoration are comparable with – or greater than – losses through development, and to effect gains preferably before, or concurrent with, the impacts (EPA 2008; Ramsar 2012).

There may be time lags between impacts and their mitigation, leading to temporal loss of wetland habitat and related ecosystem services. Restoration of wetlands may have uncertain outcomes, and there is a risk that not all ecological functions will return. Both time lags and loss of ecosystem services may have serious implications for biodiversity and for people who rely on these services for their well-being and impose high costs on society (Bendor 2009). To account for temporal losses, developers may have to compensate for a larger amount of wetland area than was impacted and/or time discounting is applied. Very high offset ratios may be needed to guarantee a fair exchange (Moilanen et al. 2009). Where delays are probable, and/or the offset site is remote from the impact site, additional compensation to affected parties may be required.

Wetland mitigation banking is a regulated system that allows the issuing of credits for wetland habitat at one or more sites (the mitigation bank) that has been restored, created, enhanced, or preserved. A wetland mitigation bank can be a privately or publicly owned land but must be protected and managed in perpetuity. Credits can be purchased by a developer as a means to offset the loss of the same type of wetland at one or more impact sites (Mead 2008), thus transferring the obligation for mitigation to the wetland bank sponsor. Wetland mitigation banking thus aims to secure measurable improvements in wetlands before impacts have occurred thereby removing the risks associated with time lags and uncertain restoration outcomes. Wetland banks often contribute to a larger conservation strategy, thus having advantages over numerous fragmented and often isolated offsets in the landscape (Mead 2008). Bank credits are the preferred offset option in the USA (EPA 2008) and increasingly favored in other countries (e.g., Australia).

Mitigation in the Environmental Assessment (EIA and SEA) Process

Environmental assessment (EIA or SEA) aims to identify and assess the potential significance of impacts, and determining measures to mitigate negative impacts and enhance benefits is part of the environmental assessment process. Potential mitigation or compensation measures have to be included in an impact study in order to assess their feasibility, so they are best identified during the scoping stage. All legal requirements and standards applicable to the particular project and its mitigation must be taken into account by specialists and the impact assessment team. Internal policies of the developer and any funders must also be addressed.

Ideally, irreplaceable wetlands should be identified and avoided in the planning process through Strategic Environmental Assessment (SEA) or at the outset of project-level impact assessment. While it has to be kept in mind that it may take time for effects to become apparent, the ability to mitigate impacts relies heavily on early identification of significant and potentially unacceptable impacts. Once detailed proposals have been developed they are relatively inflexible. Effective screening and scoping can highlight key areas of risk and potential impact, directing the consideration of better alternatives, with location, siting, or design changes. Specialist input prior to initiating the formal EIA process can improve the development proposal and streamline the EIA by identifying and mitigating impacts at the earliest possible stage of planning (so-called positive planning).

The loss of wetland habitat and its ecological functions may be unacceptable where an irreplaceable wetland type would be lost, species would become extinct, or natural or human communities would be deprived of services without substitute on which they are heavily reliant. In these cases, avoidance or prevention of impacts is essential. The higher the likelihood and magnitude of impact the greater the need to avoid – rather than manage or accept – the risks (Ramsar 2012). For “high-quality” wetlands that are difficult or impossible to restore, avoidance is recommended (Gardner 2009). To date, the emphasis on avoidance has been poor (Clare et al. 2011).

Impacts on the ecosystem services provided by wetlands must be addressed. A number of tools have been developed in this respect (e.g., Landsberg et al. 2013). If mitigation measures are to address the full range of impacts on wetlands, integration of specialist studies in social systems, health, economics, water resources, and biodiversity is required. Collaboration between the proponent, planners, engineers, and specialists is necessary in designing mitigation and enhancement to arrive at the best practicable environmental option. Too often, specialist studies are undertaken in "silos" and the recommendations of one specialist conflict with others or ignore potential synergies.

Early and ongoing engagement with key stakeholders (local communities, the public, funders, and regulators) is important to obtain local knowledge and values, identify "unacceptable" impacts up front, and evaluate the significance of potential impacts and those measures that would best mitigate or enhance impacts. The level of engagement should be commensurate with the project's risks and potential severity of impacts. Effective stakeholder engagement and independent peer review of specialist studies reduce the risk of negative publicity, public opposition, and delays, including associated costs. Involvement of interested and affected parties in implementing and monitoring mitigation enhances transparency and accountability. Engagement with local communities is invaluable in determining ways in which traditional knowledge and local cultural practices could contribute to any offset or enhancement initiative.

After efforts to avoid and minimize negative impacts have been exhausted, and residual impacts are unavoidable, a reliable measure of those impacts is needed to inform the design of offsets. A suitable currency for this measure must be used, reflecting the particular values affected. Typical currency would be the area of particular wetland habitat and the specific attributes of that habitat (BBOP 2009). For example, in Eastern Washington (Hruby 2012), the potential of a wetland's hydrologic and water quality improvement functions, its function as habitat and in maintaining food webs, the potential of the landscape to maintain each function at the site scale, and the value each function may have for society, are scored. The negative changes to functions and values as a result of the proposed development are recorded as "debits" and used to calculate how many "credits" are required from an offset site(s). The likely gains in function and values at a potential mitigation site are compared to the losses at the impact site to determine if "no net loss" would be achieved.

Offsets should target areas that are priorities for conservation of water resources and biodiversity. They should also strive to deliver local socioeconomic benefits through creation of livelihood opportunities such as wetland management, nature tourism, or Payment for Ecosystem Services (PES) schemes. Offset actions themselves may have negative impacts (e.g., removal of flood control structures) which require compensation. It is important to evaluate whether or not negative impacts could and would be satisfactorily offset. Where there is major uncertainty about the outcome of mitigation, focus should revert to avoiding or preventing the impact.

Mitigation measures must be explicit in nature and timing and in allocating responsibility for their implementation. A range of objectives and performance criteria for both structural and functional ecosystem attributes must be provided against which to check mitigation success; these criteria should be based on past performance of similar restorations, identifying consistent temporal trends in attributes of restored sites, and using natural wetlands as references (Matthews and Endress 2008). Maintenance, monitoring, and adaptive management, as well as transparency in management, are important if mitigation is to be effective, reliable, and achieve the desired ends. The duration of monitoring should be sufficiently long to gauge outcomes reliably. Independent auditing of the performance of mitigation measures and compliance with legal or policy requirements is advisable, with audit reports being made publicly available.

Robust legal and financial assurances are important ingredients of mitigation success. Measures must be in place to protect offset areas in the long term (e.g., conservation easements, protected areas; EPA 2008), at least for as long as the residual negative impacts. The calculation of financial provision must be transparent and defensible, allowing for escalation. Management plans should expand on the mitigation measures described in the impact assessment.

Adequate and Sufficient Mitigation

At the heart of discussion about mitigation is the question: “would this impact be acceptable?” If sustainable development is to be achieved, any unacceptable impacts on wetlands must be avoided or prevented. Where impacts are unavoidable but acceptable, it is crucial that they be offset and that there is a high level of assurance of successful outcomes. There are limits to what can be offset: in some situations, offsets cannot compensate for residual impacts or there may be high risks of offset failure. When development would result in unacceptable impacts, it should not be permitted (BBOP 2012; Fig. 2).In some instances, not implementing a project may have adverse effects that may be more significant than the impacts of a proposed activity (e.g., projects counteracting degradation processes).

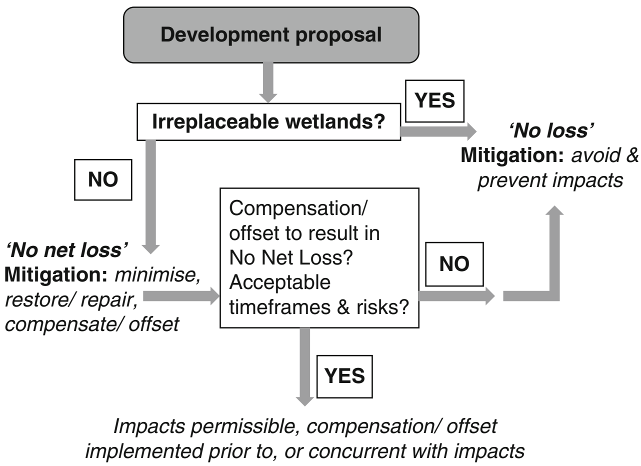

Fig. 2 Mitigation options and outcomes

Future Challenges

The main challenges with respect to mitigation of negative impacts on wetlands are: (1) setting and consistently applying explicit limits to future impacts on wetlands, i.e., determining under what conditions or contexts offsets would not be considered; (2) anticipating and taking account of the future value of wetlands in the face of climate change and growing populations, and the determination of appropriate and sufficient mitigation; and (3) effective performance, compliance monitoring and enforcement of mitigation measures.

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Environmental Impact Assessment: Wetland Mitigation Banking

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Genevieve Bennett

Contents

Abstract

In some countries, policies may be in place requiring parties responsible for causing harm to wetlands to minimize their damage and compensate for any impacts that are unavoidable. There are three common means for carrying out compensatory actions: do it yourself or hire an expert ("permittee-responsible mitigation"), pay into a fund for the compensation to be done later ("in-lieu fee fund"), or buy credits from a third party that has already developed, or "banked" compensation, in anticipation of future demand for wetland mitigation. All three of these mitigation solutions can be found in various shapes and sizes around the world. But the mitigation credit banking model is particularly widespread and mature in the United States and Germany.

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Wetlands · Mitigation · Mitigation hierarchy · Credit banking · United States of America · Germany · Compensation · Markets

Introduction

In some countries, policies may be in place requiring parties responsible for causing harm to wetlands to minimize their damage and compensate for any impacts that are unavoidable. There are three common means for carrying out compensatory actions: do it yourself or hire an expert ("permittee-responsible mitigation"), pay into a fund for the compensation to be done later ("in-lieu fee fund"), or buy credits from a third party that has already developed, or "banked" compensation, in anticipation of future demand for wetland mitigation.

All three of these mitigation solutions can be found in various shapes and sizes around the world. But the mitigation credit banking model is particularly widespread and mature in the United States and Germany. In the United States, compensatory mitigation requirements referring specifically to wetland and stream impacts drive a market for mitigation credits estimated to be worth at least \$1.3–\$2.2 billion per annum. This market has attracted a significant number of private-sector entrepreneurs (Madsen et al. 2010). In Germany, offset requirements may be triggered not only by impacts to wetlands and water bodies but also specific protected species/habitats, biodiversity, and environmental values more generally. Thus its system of land pools (*Flächenpools*) and eco-accounts (*Ökokonto*) include, but are not limited to, wetland mitigation (Morandeau and Vilaysack 2012).

Mitigation credit banking is now the preferred form of wetland mitigation in the USA (ACE and USEPA 2008), and a model of interest to many around the world, because of a number of advantages compared with the do-it-yourself or in-lieu fee fund models. Banks generally deliver higher-quality results than permittee-responsible mitigation by consolidating the ecological improvement or value on to one large site rather than many small parcels; ecological outcomes in terms of hydrologic function, habitat, and other ecosystem services can thus be greater (ACE and USEPA 2008). Additionally, the "advanced mitigation" nature of banking means that these ecological gains are achieved and verified before any impacts are even made.

Credit bankers must ensure the long-term success of mitigation through a performance bond and contingency security to cover construction and monitoring for a 5-year period after construction. Long-term management of the site must be guaranteed and endowed by the bank sponsor. Credit developers must also ensure that the wetland functions will be guaranteed to endure into perpetuity.

Permittees themselves may prefer third-party mitigation due to its transfer of liability to the mitigation banker, as well as the transfer of responsibility for designing, implementing, and long-term protection of the offset. Evidence also suggests that use of bank credits speeds time to permit in the United States compared to other mitigation methods (Birnie 2014).

Wetland Mitigation Banking in the United States

Wetland mitigation banks in the United States restore, enhance, create, or preserve an area of wetland or stream bank to generate credits for sale to parties needing to compensate for wetland impacts. Offsets must be located within the same watershed (service area) as the impact, usually designated by US Geological Survey Hydrologic Unit Codes. The US Army Corps of Engineers ("the Corps") is responsible for oversight of compensatory mitigation. Banks may be run by a government agency, nonprofit group, or a commercial developer.

Wetland protection in the United States has its roots in the 1972 Clean Water Act (CWA) which requires that parties seeking to dredge or fill navigable waters obtain a permit, typically from the Corps. "Like-for-like" replacement requirements for wetlands under federal law were first introduced in 1988 in the form of a national "no net loss" policy. Further clarification on the role of mitigation banks and guidance on their establishment followed under the Clinton administration in 1994 and 1995. In 2008, the Corps and the US Environmental Protection Agency (USEPA) jointly issued the *Final Rule on Compensatory Mitigation for Losses of Aquatic Resources*, giving official preference to mitigation banks over in-lieu fee programs or permittee-responsible mitigation, superseding the 1995 guidance (ACE and USEPA 2008).

Actors

Because the US system allows third-party development of offsets, wetland mitigation has a wealth of participants involved in creating offsets, including environmental consultants, engineers, and lawyers hired by permittees; private mitigation bankers; nonprofit organizations and government agencies running mitigation banks for commercial or their own use; and government and nonprofit organizations collecting funds and providing active programs. The most common buyers are government transportation agencies, residential and commercial developers (which account for about a third of demand), the Department of Defense, extractive industries, and utilities.

A Mitigation Banking Review Team oversees the activities of mitigation bankers, evaluates the bank product that a banker proposes to build, and certifies the creation of credits that can be used to mitigate identified impacts approved in an individual Impact Permit. The review team is made up of representatives of the Corps along with local regulatory agencies and community interests.

Rules

The number of credits generated by a restoration project is related to the area of wetland and/or the functional value of the wetland (credit determination varies by Corps district). Typically the number of credits available for sale is less than the number of hectares restored. A mitigation banker is responsible for establishing a wetland bank following financial and environmental guidelines before credits are released to the bank for subsequent sale.

Three main performance parameters are used to verify the number of credits created at the site and the site's credit value: hydrology, vegetation, and presence of exotics. Different states, and oftentimes, regulatory agencies in the same state, use different methods to certify credits. Functional assessment methods, for example, are more complex than acreage-to-credit equivalency ratios.

Market Value and Activity

Wetland and stream banking in the USA has grown in the past two decades due to official Federal Banking Guidance that had been released in 1995; agreement between federal agencies over interpretation of wetland mitigation guidance and the 2008 "Final Rule" has driven demand for mitigation bank credits. Wetland credits are estimated to account for $1.1–$1.8 billion, while streams credits account for $240–$430 million (Madsen et al. 2010).

In 2010, the active number of banks in the USA increased to 798 from 431 in 2009, according to Ecosystem Marketplace data. Median bank size in 2008 was 174 acres. The national range in credit prices in 2008 was $3,000–$653,000, with the average price at $74,535 (Madsen et al. 2010). The variability in the market value of wetland CWA section 404 credits reflects differences in the availability and price of land suitable for bank development and the cost to create an acre of wetland compensation within a given region.

Wetland Mitigation Banking in Germany

In Germany, developers and other actors seeking mitigation for their impacts have the option of contracting with land pools and associated eco-accounts, rather than undertaking permittee-responsible mitigation. Germany's 1976 Law of Nature Protection created a mitigation hierarchy, requiring compensation for unavoidable impacts. Compensatory actions must support equivalent ecological functions to those lost, within the same spatial context. Compensation requirements apply to multiple habitat types – not only wetlands. Amendments to the Federal Nature Conservation Act in March 2010 established the concept of "natural areas" to reflect spatial relationships between the sites of intervention and compensation measures; compensation measures must take place with the same "natural area." The updated Act also attempted to limit the use of high-priority agricultural land for compensation projects.

Actors

In Germany, lands with high restoration potential can be designated part of "land pools" through landscape planning processes. Compensatory activities take place within these pools. When lands are restored to high ecological values in anticipation of demand for compensation, they constitute an "eco-account." Typically the implementing actor is a municipal government.

Rules

Mitigation requirements for developers are estimated through a point system. Project developers can either compensate an agency for managing land in a land pool or purchase eco-points from already-developed projects, which are accordingly debited from the eco-account. Eco-accounts resemble wetland mitigation banks in the United States in that in both cases the offset is already implemented prior to compensation taking place.

Eco-point scores are calculated based on federal State-specific biotope lists, regional conservation priorities, and local guidance; thus there can be considerable variation in calculation methods. Eco-points are also not comparable between different habitat types. Rather, eco-points are a general term for the crediting basis for mitigation, as a way to measure ecological value of an impact and demonstrate equivalency between the impact and compensation sites (Naumann et al. 2008).

Market Value and Activity

As discussed earlier, laws governing compensatory mitigation in Germany take a wider scope than in the United States. Offset requirements may be triggered not only by impacts to wetlands and water bodies but also specific protected species/habitats, biodiversity, and environmental values more generally. In contrast, distinct laws and bodies of regulation exist in the United States for wetland/stream versus species/habitat compensatory mitigation. Thus, while land pools and eco-accounts in Germany may support wetland enhancement and conservation, they also focus on a range of other biotopes.

Twenty one eco-accounts were active in Germany as of 2012; data on which of these involve wetlands are unavailable (Morandeau and Vilaysack 2012). One known example is the Ruhr Valley's compensation pool, which undertook restoration of the Lippe River floodplain as compensation for impacts on behalf of the Rhenish-Westphalian Water Supply Company (BBOP 2009).

Outlook and Challenges

Interest in wetland mitigation, and more broadly biodiversity, banking has been growing worldwide over the past decade or two (The Biodiversity Consultancy 2012, 2013) as a possible part of the solution to avoid and compensate for negative environmental impacts of economic development. England, Colombia, China, Spain, South Africa, Sweden, and other countries are actively exploring the practice.

Businesses like Shell, Walmart, and Rio Tinto are also interested in its potential to manage biodiversity risks – physical, reputational, regulatory, and legal.

In the United States, energy development, funds for coastal restoration in the Gulf of Mexico, and renewed commitments to transportation investment and other infrastructure projects indicate continued growth in the near-term for the industry. In Germany, policy changes in 2010 attempted to strike a balance between designating land available for ecological restoration and keeping high-value agricultural lands in production. However, a number of regulatory questions have yet to be settled, including monitoring and long-term maintenance, credit generation methodologies, clarifying the mitigation hierarchy, trading and banking credits in compensation pools, and harmonizing standards across state governments in order to increase market liquidity (Heugel et al. 2010; Federal Ministry for the Environment 2010). In both countries, a lack of data on long-term ecological outcomes and clarity around methodologies and regulations remain a challenge.

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280 Environmental Impact Assessment for Wetlands: Stakeholders and Public Participation

Roel Slootweg

Contents

Abstract

Public participation is a prerequisite for Environmental Impact Assessment (EIA) and can consist of informing, consulting or full participation of stakeholders in the EIA process (shared analysis and assessment). Public participation is considered essential for the scoping and review phases of EIA, but also highly desirable for enhancing the quality of the whole assessment process. Relevant stakeholders include the project target groups, other people affected by the project, formal and informal organizations involved, and people who may rely on the wetland in the future. Benefits of participation include learning and empowerment of stakeholders, improved data collection, increased chances of success of the project, and in general good governance and sustainable development. The article concludes with a discussion of operating principles of participation (e.g. planning and facilitation). Some constraints are also presented, such as incomplete representation of stakeholders, communication barriers and legal limitations to participation.

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Introduction: EIA and Public Participation

Environmental Impact Assessment (EIA) is concerned with information, participation, and transparency of decision-making (Ramsar Convention 2010a). Public involvement consequently is a prerequisite for effective EIA and can take place at different levels: informing (one-way flow of information), consulting (two-way flow of information), or "real" participation (shared analysis and assessment). In all stages of EIA, public participation is relevant. Public participation may be defined as the involvement of individuals and groups that are positively or negatively affected by, or interested in, a proposed project that is subject to a decision-making process (André et al. 2006). The legal requirements for and the level of participation differ among countries, but it is generally accepted that public consultation at the scoping and review stages of EIA are essential. Participation during the assessment study is generally acknowledged to enhance the quality of the process.

The individuals and groups involved in or affected by a development project are called stakeholders. With respect to wetlands, relevant stakeholders in the EIA process are (Fig. 1):

- Beneficiaries of the project: target groups making use of, or putting a value to, known ecosystem services which are purposefully enhanced by the project
- Affected people: people who experience intended or unintended changes in ecosystem services, as a result of the project
- General stakeholders: formal or informal organizations and groups representing either affected people or the affected wetland
- Future generations or "absent stakeholders": stakeholders of future generations, who may in the future rely on wetlands around which decisions are made in the present

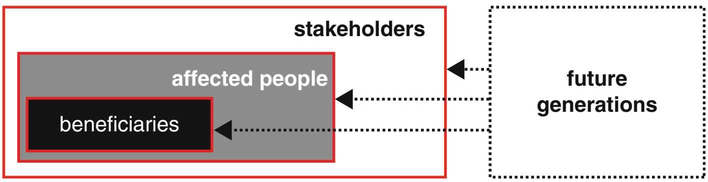

Fig. 1 Stakeholders in the EIA process (Figure re-created and modified based on original in Slootweg et al. 2006)

Objectives of Public Participation

Public participation can inform and educate the stakeholders, including the proponent, the general public, the decision-maker(s), and the regulator, on the planned intervention and its consequences, and thus help provide information to the public. In this way, participation contributes to the mutual learning of stakeholders and to improvement of EIA practice for a proposal. Participation may empower local communities to become better advocates for their own interests in the development project.

Besides informing the stakeholders, participation can also help in gathering data and information from the public about their human (including cultural, social, economic, and political dimensions) and biophysical environment, as well as about the relations they have with their environment (including those related to traditional and local knowledge). Input from the public on the planned intervention, including its scale and timing can contribute to identifying options to reduce its negative impacts, to increase its positive outcomes, or to compensate impacts which may not be mitigated. In this way, public participation can help improving the quality of the project.

Finally, participation can contribute to a better analysis of proposals leading to more creative development, more sustainable interventions, and consequently greater public acceptance and support than would otherwise be the case. Inviting the affected and interested public into the decision-making process can foster justice, equity, and collaboration. Involving the public in this way is considered essential for good governance.

Basic Principles of Participation

Public participation practice in EIA should be adapted to the context. It is important to understand and appreciate the social institutions, values, and culture of the communities in the project area, and to respect the historical, cultural, environmental, political, and social backgrounds of the communities which are affected by a proposal. The participation process needs to be informative and proactive, recognizing that the public has a right to be informed early and in a meaningful way in proposals which may affect their lives or livelihoods. Increased interest and motivation to participate occur by providing simple and understandable information to the public.

The process should be adaptive and communicative, recognizing that the public is heterogeneous in terms of demographics, knowledge, power, values, and interests. The rules of effective communication among people, respecting all individuals and parties, should be followed. It is also important that the process is inclusive and equitable, ensuring that all interests (including non- or underrepresented interests) are respected. The participation of less-represented groups, including indigenous peoples, women, children, elderly, and poor people should be encouraged, or their interests represented in another way. Equity between present and future generations in a perspective of sustainability should be promoted.

When contributing to mutual respect and understanding of the values, interests, rights, and obligations of all IA stakeholders, participation also has education and cooperation aspects. It promotes cooperation, convergence, and consensus building rather than confrontation. Engaging conflicting perspectives and values as well as trying to reach a general acceptance of the proposal toward a decision that promotes and supports sustainable development should be pursued. Ideally, the results of the participation improve the proposal under study, taking into account the results of the participatory process. This would include reporting and feedback to stakeholders about the results of the participation process, especially on how their inputs have contributed to decision-making.

Operating Principles of Public Participation

Public participation should be initiated early and sustained throughout the EIA process. The public should be involved before major decisions are made. This builds trust among participants, gives more time for participation, improves community analysis, improves screening and scoping of the EIA, increases opportunities to modify the proposal in regard to the comments and opinions gathered during the participation process, reduces the risk of rumors, and improves the public image of the proponent. It can also give the regulator more confidence for making the approval decision.

It is important that the participation process is well-planned and focused on negotiable issues. When all EIA stakeholders know the aims, rules, organization, procedure, and expected outcomes, this will improve the credibility of the process for all involved. Because consensus is not always feasible, participation should emphasize understanding and respect for the values and interests of the participants, and focus on negotiable issues relevant to decision-making. Many communities have their own formal and informal rules for public access to resources, conflict resolution, and governance. The participatory process needs to be adapted to the social organization of the impacted communities, including the cultural, social, economic, and political dimensions. This shows respect for the affected community and may improve public confidence of the process and its outcomes.

In this way, the process can become supportive to participants. Adequate diffusion of information on the proposal and on the participatory process and just and equitable access to funding or financial assistance will make people feel supported in their will to participate. Capacity building, facilitation, and assistance should be provided particularly for groups who don't have the capacity to participate and in regions where there is no culture of public participation, or where local culture may discourage participation. Participation then becomes open and transparent, involving all groups who are affected by a proposal and interested in participating, regardless of ethnic origin, gender, and income. All information required for the evaluation of a proposal (e.g., terms of reference, report, and summary) should be accessible to laypersons. Laypersons should be able to participate in relevant workshops, meetings, and hearings related to the EIA process. Information and facilitation for such participation should be provided.

Finally, it is important to safeguard the credibility and rigor of the participation process. The process should adhere strictly to established ethics, professional behavior, and moral obligations. Facilitation of the process by a neutral facilitator improves impartiality of the process as well as justice and equity in the right to information. It also increases the confidence of the public to express their opinions and reduces tensions, the risk of conflicts among participants, and opportunities for corruption. In a formal context, the adoption of a code of ethics is encouraged (see also Ramsar Convention 2010b).

Constraints to Public Participation

Potential constraints to effective public participation include:

- Incomplete representation of stakeholders in the process. This can be caused by deficient identification of relevant stakeholders, or by large distances which make communication and participation difficult and expensive. Sometimes poverty plays a role because involvement in the process requires time spent away from income-producing tasks.
- Communication barriers. These can be caused by illiteracy or a lack of command of nonlocal languages. This can inhibit representative involvement if print media are used. Local values and culture can determine behavioral norms or practices which inhibit involvement. Some groups may not feel free to disagree publicly with dominant groups. In some areas, different languages or dialects may be spoken, making communication difficult. Some interest groups may have conflicting or divergent views, and vested interests.
- Formal administrative or legal barriers. Legal systems may be in conflict with traditional systems and cause confusion about rights and responsibilities for resources. Confidentiality can be important for the proponent, who may be against early involvement and consideration of alternatives.

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Health Impact Assessment for Wetlands 281

Roel Slootweg

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Abstract

Wetlands traditionally were considered a threat to human health through water-borne diseases. More recently, the positive contributions of wetland ecosystem services to human well-being and health are recognized broadly but need more clear articulation in planning and management. Strategic Environmental Assessment (SEA) and Environmental Impact Assessment (EIA) for water and wetlands need to incorporate health issues in a comprehensive way to ensure a balanced assessment of health risks, benefits and safeguards. This can be done by recognizing that wetlands not only satisfy biophysical and material human needs for water and nutrition, but also provide important socio-cultural, aesthetic and spiritual benefits. Health Impact Assessment (HIA) consists of procedures, methods and tools that evaluate the effects of policies, plans, programs or projects on the health of a population and identify appropriate management actions. HIA thus influences decision-making to ensure effective integration of health protection and promotion into development planning.

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Keywords
Health impact assessment · Strategic environmental assessment · Environmental impact assessment · Development planning · Ecosystem services · Wetland ecosystems

Wetlands and Human Health in Environmental Assessment

Wetland ecosystems provide a set of ecosystem services that contribute to human well-being and poverty alleviation. While it is impossible to imagine human life without water, the importance of the relationship between wetlands and water is less well recognized, and this relationship has changed over time (Horwitz et al. 2012).

A widespread view of wetlands as 'the problem' for human health is that wetlands are seen primarily as the source of vector- or water-borne diseases. This misinterpretation requires careful treatment and attention. More emphasis on the benefits that humans derive from wetlands, including a richer sense of the roles of biodiversity in parasite regulation, is required. Understanding these benefits provides the basis for better land and water management, for fostering human health and well-being while managing wetlands. Wetland managers must have information that will allow them to articulate, and respond professionally to, these claims.

Some groups of people, particularly those living near wetlands, are highly dependent on wetland ecosystem services and are directly harmed by their degradation. For others, wetlands are the basis of economic structures and are embedded in cultural expressions. These benefits can also determine human health, directly and indirectly, by contributing to other forms of well-being, like providing security and basic materials for a good life and fostering good social relations. If wetlands are more than a source of disease, if they play an important role in sustaining human health and well-being, and if they continue to be lost and degraded more rapidly than other ecosystems, then more effective treatment of the trade-offs between different forms of benefits will be required.

Strategic Environmental Assessment (SEA) at the strategy, policy and programme level and Environmental Impact Assessment (EIA) at the project level have traditionally addressed health issues. Human health is, however, often a single bullet point on an EIA or SEA check list. The assessment of health effects is likely to be biased towards bio-physical health determinants rather than being a holistic view that also includes important wider determinants of health. The scope of health issues covered may reflect the industrial-country roots of EIA, and therefore lack the level of comprehensiveness necessary to make the assessment fully relevant to local health conditions. Most importantly, EIA procedures frequently do not recognize the fact that the ultimate authority for health pertains to Ministries of Health (central or peripheral levels), which should have the regulatory responsibilities for the planning, quality control and final approval of health impact assessment and its follow-up. Because of this, care needs to be taken:

- To ensure that health is covered comprehensively.
- To strike an acceptable balance between strengthening of health services and design and operational measures by other sectors to safeguard health and well-being.
- To adequately address the wider determinants of health.
- To anchor the final authority for the health component with the Ministry of Health.

Wetlands and Human Health Benefits

Wetland ecosystems determine human health and well-being through a number of characteristic influences. Wetlands are a source of hydration and safe water; a source of nutrition; sites of exposure to pollution or toxicants; sites of exposure to infectious diseases; sites of physical hazards; settings for mental health and psycho-social well-being; places from which people derive their livelihood; places that enrich people's lives, enable them to cope and to help others; and sites from which medicinal products can be derived. The benefits of wetland ecosystems for human health can be approached in at least three inter-related ways:

- by recognizing the human needs that are met by water in its setting;
- by recognizing the health products that come from wetland ecosystems;
- and by recognizing the economic value of wetlands in a full sense, in a way that allows individuals within wetland ecosystems to sustainably improve their socio-economic conditions.

With respect to human needs, health benefits will accrue when human social and cultural needs are satisfied by access to wetlands. Health relates most easily to the direct survival requirements, which include water for food, water for drinking, cooking and eating, washing, cleaning, health and health care, and for waste removal and assimilation. Water is needed to generate income and material well-being, and access to water generates prestige and social identity, contributes to social cohesion, allows for recreation while providing an aesthetic opportunity, all embedded within moral, cultural and spiritual needs.

As for health products, health benefits will accrue to societies in general and individuals in particular when products of wetlands can be used for pharmaceutical or other medicinal purposes. Wetland-associated animals, fungi, bacteria and lower plants (algae), some of them living in extreme conditions, provide the most productive sources of new natural products. The medicinal qualities of these are a good example of the value of traditional knowledge to health care today. Links between wetland biodiversity and human health should focus less exclusively on the obvious (such as birds, large mammals, or plants) and more on the “hidden biodiversity” (such as fungi and bacteria).

In terms of economic value, wetland ecosystem services contribute to the material well-being (and socio-economic status) of individuals and populations, and they can be valued in economic terms. As the socio-economic status of individuals improves, their health outcomes improve as well. Valuation studies highlight the significant contribution of wetlands to local, national, regional and global economies. Several of these studies also indicate that when both the marketed and non-marketed economic benefits are included, the total economic value of an unconverted wetland is frequently greater than that of a converted wetland.

Health Impact Assessment (HIA)

Loss of wetland components and disruptions of wetland functions and ecosystem services will have consequences for human health. Furthermore, adverse health outcomes are likely to be distributed in an unequal way, i.e., along socio-economic lines. As a consequence, health problems which are influenced by the environment cannot be solved by medical approaches to health alone. Rather, broader approaches are needed, drawing on a wider scientific base, including ecological and social sciences. This presumes that humans are not separable from the natural environment, and that socio-economic factors mediate human health.

Over the last decade the impact assessment family of tools has seen a rapid development of Health Impact Assessment (HIA). HIA may be defined as a combination of procedures, methods and tools that systematically judges the potential, and sometimes unintended, effects of a policy, plan, program or project on the health of a population and the distribution of those effects within the population. HIA identifies appropriate actions to manage those effects (Quigley et al. 2006).

The concept of HIA is based on the notion that human health and the physical and social environment are intricately linked. Individual and population health status is largely the result of the social, cultural and physical environment in which we live. Factors such as the state of our environment, access to resources to meet our basic needs, our exposure to risks and capacity to cope with these, our income and education level, and our social network of relationships with friends, family and neighbors all have considerable impacts on health and well-being.

Human health has a number of determinants that go beyond individual lifestyle choices. These include (1) Determinants related to the individual: genetic, biological, lifestyle/behavioral and/or circumstantial; (2) Social and environmental determinants: physical, community conditions and/or economic/financial; and (3) Institutional determinants: the capacity, capabilities and jurisdiction of public sector institutions and the wider public policy framework supporting the services they provide. Health Impact Assessment (HIA) aims to identify how development induces unintended changes in health determinants and resulting changes in health outcomes. HIA provides a basis for proactively addressing any risks associated with health hazards. HIA also addresses health improvement opportunities in development. Health hazards, risks and opportunities also may be addressed explicitly in environmental assessment.

Development planning is concerned with social and economic development, such as energy, agriculture, industry and transport, and is typically conducted outside the health sector. These other sectors outstrip the health sector in the potential to affect, protect and promote population health because of the considerably larger proportion of resources at their disposal, and a responsibility for action that may change environmental and social health determinants significantly. Development planning without adequate consideration of human health may pass hidden "costs" on to affected communities, in the form of an increased burden of disease and reduced well-being. From an equity point of view, marginalized and disadvantaged groups often experience most of these adverse health effects. From an institutional point of view, the health sector must cope with these development-induced health problems and incurs the costs of dealing with an increased disease burden.

HIA provides a systematic process through which health hazards, risks and opportunities can be identified and addressed early in the development planning process, to avoid the transfer of these hidden costs and to promote multisectoral responsibility for health and well-being. The production of public health management plans with safeguards, mitigating measures and health promotional activities is an integral part of HIA. Key steps in the HIA process are presented in Box 1.

Purpose and Functions of HIA

The purpose of all HIA is to inform and influence decision making on proposals and plans so that health protection and promotion are effectively integrated into them. Linked to this central purpose, HIA has an important function contributing to healthy projects and healthy public policy. For example, HIA involves and engages health experts, project proponents, other key players and the community affected by the proposal, and facilitates public participation in decision making. HIA also attempts to identify health inequalities that may arise from a proposal and addresses cross-cutting health issues with repercussions for sustainability. HIA can help place public health on the agenda of many different agencies and individuals and increases awareness of what determines health status, thereby providing a basis for improved collaboration within and between agencies. In this way, HIA provides a "license to operate," not only for public bodies, but also for private sector companies who incorporate social and health responsibility into their activities. HIA can thus be a tool for intersectoral action for health. By focusing on the health status of vulnerable groups, HIA may reduce the burden on health sector services.

HIA has no formal basis in law or regulations. It can be implemented voluntarily as a stand-alone assessment of proposed projects or it can be integrated into an EIA procedure if the project is requiring such an assessment. Government bodies in some countries (public health authorities) as well as some large corporations (predominantly in energy & mining) have adopted HIA procedures.

Box 1: the Health Impact Assessment (HIA) process (based on key steps and responsibilities presented in Quigley et al. 2006)

1. SCREENING
Deciding the scale of HIA that is required. This is a desk exercise by the responsible ministry or authority.

2. SCOPING
Defining time and space boundaries for assessment and setting Terms of Reference (TOR) for a full-scale HIA. This is usually done by the Ministry of Health (MOH, at central, province and/or district level) and key stakeholders.

3. FULL SCALE HIA
This is done by the HIA team and follows the specifications in the TOR.

4. PUBLIC ENGAGEMENT AND DIALOGUE
This is initiated by the relevant authority (e.g. MOH).

5. APPRAISAL OF THE HIA REPORT
To assess if the report complies with the TOR, control the quality of independent criteria, and appraise the recommendations in terms of feasibility, soundness and acceptability. This is done by the MOH or a consultant appointed by the MOH.

6. ESTABLISHMENT OF FRAMEWORK FOR INTERSECTORAL ACTION
This is done by the MOH and other ministries involved.

7. NEGOTIATION OF RESOURCE ALLOCATIONS
This is for the resources needed for health safeguard measures, done by the Ministry of Finance together with other relevant ministries.

8. MONITORING, EVALUATION AND FOLLOW-UP
Monitoring of compliance and health indicators, and evaluation and follow-up by MOH and line ministries.

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Environmental Impact Assessment for Wetlands: Assessment and Evaluation

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Roel Slootweg

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Abstract

The main goal of the "Impact assessment and Evaluation" phase of Environmental Impact Assessment (EIA) is to improve the understanding of impacts identified during the screening and scoping and to evaluate various aspects including e.g. assessment criteria, mitigation and management of impacts. In this article some practical lessons of wetland assessment studies are presented, such as the importance of seasonal variability, the involvement of local communities and other stakeholders, and the importance of considering the full range of drivers of change. The paper concludes with a discussion of the assessment of genetic, species and ecosystem diversity in wetlands.

Keywords

Environmental impact assessment · Impact assessment and evaluation · Decision-making · Environmental policy · Governance · Biodiversity

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Introduction

EIA should be an iterative process of assessing impacts, redesigning alternatives, and comparison (Ramsar Convention 2010). The main tasks of impact analysis and assessment are:

(a) Refinement of the understanding of the nature of the potential impacts identified during screening and scoping and described in the terms of reference. This includes the identification of indirect and cumulative impacts, and of the likely cause–effect chains.
(b) Identification and description of relevant criteria for decision-making can be an essential element of this stage.
(c) Review and redesign of alternatives; consideration of mitigation and enhancement measures, as well as compensation of residual impacts; planning of impact management; evaluation of impacts; and comparison of the alternatives.
(d) Reporting of study results in an environmental impact statement (EIS) or EIA report.

Lessons from Practice

Assessing impacts usually involves a detailed analysis of their nature, magnitude, extent and duration, and a judgment of their significance, i.e., whether the impacts are acceptable to stakeholders and society as a whole, require mitigation and/or compensation, or are unacceptable.

Available wetland information is often limited and descriptive and cannot be used as a basis for numerical predictions. The priorities and targets set in the National Wetland Policy or National Biodiversity Strategy and Action Plan process can provide guidance for developing assessment criteria. In order to deal with uncertainty, risk assessment techniques, precautionary approach, and adaptive management may be needed.

In evaluating the significance of residual impacts for wetland-related values, reference can be made to Ramsar guidelines on ecological character and on risk assessment (see e.g., Resolutions VIII.8, IX.1 Annex E, [X.16] and Ramsar Technical Report [A Framework for Assessing the Vulnerability of Wetlands to Climate Change]).

A number of practical lessons with respect to the study process have emerged including that the assessment should:

(a) Allow for enough survey time to take seasonal features into account, where confidence levels in predicting the significance of impacts are low without such survey; For seasonally fluctuating wetlands, inundation mapping and hydroperiod data may be crucial. Remote sensing/earth observation sources are increasingly available to assist with this – see e.g., Ramsar Technical

Report No 2 (2006): Low-cost GIS software and data for wetland inventory, assessment and monitoring.

(b) Focus on processes and services, which are critical to human wellbeing and the integrity of wetlands. Explain the main risks and opportunities for wetlands.

(c) Apply the ecosystem approach and actively seek information from relevant stakeholders and indigenous and local communities. Address any request from stakeholders for further information and/or investigation adequately. This does not necessarily imply that all requests need to be honored; however, clear reasons should be provided where requests are not honored.

(d) Consider the full range of factors affecting wetlands. These include direct drivers of change associated with a proposal (e.g., land conversion, vegetation removal, emissions, disturbance, introduction of invasive alien species or genetically modified organisms, etc.) and, to the extent possible, indirect drivers of change, including demographic, economic, sociopolitical, cultural, and technological processes or interventions.

(e) Evaluate impacts of alternatives with reference to the baseline situation. Compare against legal standards, thresholds, targets, and/or objectives for biodiversity. Use national biodiversity strategies and action plans or wetland policies and other relevant documents for information and objectives. The vision, objectives, and targets for the conservation and sustainable use of wetlands contained in local plans, policies, and strategies, as well as levels of public concern about, dependence on, or interest in, wetlands provide useful indicators of acceptable change. In the case of Ramsar sites, the “baseline” should relate to the site’s ecological character, as distinct from the attributes which cause it to qualify as internationally important. Hence the baseline should be the target condition (ecological character) described in the objectives of the management plan for the relevant Ramsar site. It will therefore not necessarily equate to the condition of the site described at the time of listing (or subsequent updating of the Ramsar Information Sheet) unless at such times the site happens to have achieved its optimal (target) condition or if there is no better baseline available.

(f) Take account of cumulative threats and impacts resulting either from repeated impacts of projects of the same or different nature over space and time, and/or from proposed plans, programs, or policies.

(g) Recognize that wetlands are influenced by cultural, social, economic, and biophysical factors. Cooperation between different specialists in the team is thus essential, as is the integration of findings, which have bearing on biodiversity.

(h) Provide insight into cause–effect chains. Also explain why certain chains do not need to be studied.

(i) If possible, quantify the changes in wetland composition, structure, and key processes, as well as ecosystem services. Explain the expected consequences of the loss of wetland associated with the proposal, including the costs of replacing ecosystem services if they will be adversely affected by a proposal.

(j) Indicate the legal provisions that guide decision-making. List all types of potential impacts identified during screening and scoping and described in the terms of reference and identify applicable legal provisions. Ensure that potential impacts to which no legal provision applies are taken into account during decision-making. When the consequences of impacts on a Ramsar site include reduction or deletion of the site, the provision of compensation is governed by Article 4.2 of the Convention and the guidelines adopted under Resolution VIII.20 will apply.

Genetic, Species, and Ecosystem Diversity

In practice it is not common that all three levels of diversity are treated in impact assessment; most attention goes to legally protected species and their habitats (CBD 2006; Slootweg et al. 2006). Yet, there are some biological and/or societal reasons why certain issues merit special attention without necessarily being covered by regulations. Conducting **genetic studies** per se for determining impacts of projects at the genetic level is both extremely difficult and not usually feasible within the time frame in which EIA studies are generally conducted. One aspect which may merit special attention at genetic level is the risk of genetic erosion, especially for: (i) highly threatened or legally protected species in the wild; (ii) varieties/cultivars/breeds of cultivated plants and domesticated animals and their relatives; (iii) species which are limited in numbers and/or have highly separated populations (rhinoceros, tigers, etc.); or (iv) ecosystems that may become isolated and thus obstruct gene flow (this applies to many species that depend on construction of so-called eco-ducts across major line infrastructure). Another issue at genetic level is the introduction of living modified organisms that can possibly transfer transgenes to endemic plant or animal species.

Species receiving attention in EIA have restricted distribution ranges; occupy specialized habitats; are endemic to an area; are locally distinct subspecies; are already vulnerable on account of existing threats to its habitats; have small isolated populations; or are rare or uncommon, either internationally, nationally, or locally. The species selection can be guided by a list of nationally protected species under country law as these would represent species that command highest conservation priority at the local level. Additionally, the policy documents like Biodiversity Action Plans should be useful in prioritizing species recommended for conservation action. For a globally and regionally important species, the IUCN Red List serves as a good guide to species selection.

Ecosystems include habitats that are critical for survival of rare and endangered species; or perform critical functions such as routes for migration, dispersal, and genetic exchange of wild species; or serve as buffer areas of designated conservation units such as national parks and habitats suitable for reintroduction of species in alternative habitats. Assessment of magnitude and nature of impacts on habitats provide adequate guidance for determining the significance of impacts to suggest a "no go" option or, alternatively, suggest appropriate mitigation strategies for timely action for conservation.

Ecosystems and the plants and animals within them provide humans with ecosystem services that would be very difficult to duplicate. For example, coral reef and associated mangrove forests play a critical but often undervalued role for coastal residents throughout the world. Coastal protection from waves and storm surge, food security, recreation, and tourism are but a few of the ecosystem services provided by coral reef ecosystems. While many of these services are performed seemingly for "free," they are worthy for assessment and monetary valuation. Environmental impact assessment requires consideration of how biodiversity can be sustained as the basis for provision of ecosystem services and the support of livelihoods. The fundamental concept is to understand the relationship between the biophysical status of ecosystems (and biodiversity) in terms of provisioning of ecosystem services and the links to livelihood of people. Valuation of the changes to the provision of ecosystem services under alternative project scenarios to different uses becomes relevant to asses changes in costs and benefits from the alternative project scenarios for different groups, and highlights the incremental cost or benefit of changing the biophysical status of a particular ecosystem.

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Social Impact Assessment for Wetlands

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Frank Vanclay

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Abstract

Social impact assessment (SIA) comprises the processes of analyzing, monitoring, and managing the social consequences of planned interventions, such as projects, plans, programs, or policies. SIA arose alongside environmental impact assessment (EIA) in the early 1970s to focus on the social (rather than biophysical environmental) impacts. Social impacts are changes that occur as a result of the planned intervention to how people live, work, play, and interact with one another, their culture, community and political systems, their environment, health and well-being, personal and property rights, and their fears and aspirations. SIA consists of an analysis of the communities likely to be affected by the planned intervention (stakeholder analysis), and collection of baseline data to enable measurement of change over time. Alternative options for the intervention and for mitigating potential impacts are identified. A monitoring plan to monitor change over time is developed and an adaptive management process to address unanticipated changes is implemented. An agreement making process between the communities and the developer should be facilitated, including an Impacts and Benefits Agreement (IBA) and a Social Impact Management Plan (SIMP). Finally, processes should be put in place to enable proponents, government authorities, and civil

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society stakeholders to implement arrangements implied in the SIMP and IBA and to incorporate management action plans into their own organizations. A grievance mechanism should be established to ensure that people with complaints against the proponent have a mechanism by which their concerns can be heard and resolved.

Keywords
Social impact assessment · Social licence to operate · Social performance · Free, prior and informed consent · Human rights

Introduction

Social Impact Assessment (SIA) comprises the processes of analyzing, monitoring, and managing the social consequences of planned interventions (Vanclay 2003; Esteves et al. 2012). Planned interventions can be projects, plans, programs, or policies. SIA arose alongside Environmental Impact Assessment (EIA) in the early 1970s to be a form of impact assessment focusing on the social (rather than biophysical environmental) impacts (Vanclay 2014). However, whereas EIA is required by law in most countries, relatively few countries specifically require SIA although some include the assessment of social impacts within EIA. In the commercial world, social impacts are often assessed within an integrated environmental, social, and health impact assessment (ESHIA).

Defining and Describing Social Impact Assessment

The International Principles for Social Impact Assessment (Vanclay 2003) state that: “Social Impact Assessment includes the processes of analysing, monitoring and managing the intended and unintended social consequences, both positive and negative, of planned interventions (policies, programs, plans, projects) and any social change processes invoked by those interventions. Its primary purpose is to bring about a more sustainable and equitable biophysical and human environment.” Although SIA has a common origin with EIA, over time many differences have emerged. In particular, SIA has demonstrated its business case to the commercial world and is often undertaken for project development in many contexts whether required by law or not, especially in the extractive industries (Esteves and Vanclay 2009; Franks and Vanclay 2013). SIA is also advocated by many industry organizations, the International Finance Corporation, and other development banks, and by those project financing institutions that subscribe to the Equator Principles, the global financial industry benchmark for determining, assessing, and managing environmental and social risk in projects. SIA will be a necessity if proponents are to meet the requirements of the principle of free, prior, and informed consent when dealing with Indigenous peoples (Hanna and Vanclay 2013). SIA is also likely to be useful for companies to comply with the due diligence expectations of the United Nations Guiding Principles for Business and Human Rights (United Nations 2011; Kemp and Vanclay 2013).

SIA therefore should not be understood merely as the task of predicting social impacts in a regulatory impact assessment process. Instead, the focus nowadays in SIA is on managing the social issues. There is a strong concern with enhancing positive outcomes and not just with mitigating negative consequences (João et al. 2011). As the International Principles (Vanclay 2003) argues, this contemporary understanding of SIA implies that the goal of impact assessment is to bring about a more ecologically, socioculturally, and economically sustainable and equitable outcome. Therefore, SIA promotes community development and empowerment, builds capacity, and develops social capital (social networks and trust). SIA takes a proactive stance to development and better development outcomes, not just the identification or amelioration of the negative or unintended outcomes. Assisting communities and other stakeholders to identify their development goals, and ensuring that positive outcomes are maximized, can be more important than minimizing harm from negative impacts.

SIA can be applied to a wide range of planned interventions and can be undertaken on behalf of a wide range of actors, and not just within a regulatory framework. SIA contributes to the adaptive management of policies, programs, plans, and projects, and informs the design and operation of the planned intervention. SIA builds on local knowledge and utilizes participatory processes to analyze the concerns of interested and affected parties. It involves stakeholders in the assessment of social impacts, in the analysis of alternatives, in the monitoring of the planned intervention, and in the planning of enhancement measures.

Good practice in SIA accepts that social, economic, and biophysical impacts are inherently and inextricably interconnected. Change in any of these domains will lead to changes in the other domains. SIA must, therefore, develop an understanding of the impact pathways that are created when a change in one domain triggers impacts across other domains, as well as the iterative or flow-on consequences within each domain. In other words, there must be consideration of the second and higher order impacts and of cumulative impacts.

In order for the discipline of SIA to learn and grow, there must be analysis of the impacts that occurred as a result of past activities. Thus, ex-post SIA is important, as well as the typical ex-ante assessments. SIA must also be reflexively critical of its theoretical bases and practice. While SIA is typically applied to planned interventions, the techniques of SIA can also be used to consider the social impacts that derive from other types of events, such as natural disasters and epidemics, as well as demographic and other changes in a community.

What Are Social Impacts?

Social impacts are all the impacts on humans, including on all the ways people and communities interact with their sociocultural, economic, and biophysical surroundings. SIA thus includes a wide range of specialist subfields addressing topics such as: aesthetic impacts, archaeological and cultural heritage impacts (both tangible and nontangible), community impacts, cultural impacts, demographic impacts, development impacts, economic and fiscal impacts, gender impacts, health and mental health impacts, impacts on Indigenous rights, infrastructural impacts, institutional impacts, leisure and tourism impacts, political impacts (human rights, governance, democratization, etc.), poverty, psychological impacts, resource issues (access and ownership of resources), impacts on social and human capital, and other impacts on societies. As such, comprehensive SIA requires a team approach.

A convenient way of conceptualizing social impacts is as changes that occur as a result of the planned intervention to one or more of the following (Vanclay 2002, pp.185–186):

- People's way of life – that is, how they live, work, play, and interact with one another on a day-to-day basis
- Their culture – that is, their shared beliefs, customs, values, and language or dialect
- Their community – its cohesion, stability, character, services, and facilities
- Their political systems – the extent to which people are able to participate in decisions that affect their lives, the level of democratization that is taking place, and the resources provided for this purpose
- Their environment – the quality of the air and water people use; the availability and quality of the food they eat; the level of hazard or risk, dust, and noise they are exposed to; the adequacy of sanitation, their physical safety, and their access to and control over resources
- Their health and well-being – where health is understood in a manner similar to the World Health Organisation definition: "a state of complete physical, mental, and social well-being, not merely the absence of disease or infirmity"
- Their personal and property rights – particularly whether people are economically affected, or experience personal disadvantage, which may include a violation of their civil liberties
- Their fears and aspirations – their perceptions about their safety, their fears about the future of their community, and their aspirations for their future and the future of their children.

Vanclay (2002), using a framework developed by Slootweg et al. (2001), develops this understanding of social impacts into a thorough analysis of the social change processes and the personal and collective experience of perceptual and corporeal impacts created by the activities that comprise a planned intervention. What is particularly important to appreciate is that people react to the impacts they experience, and their changed behavior can then lead to more social and biophysical changes and impacts. Thus, the second and higher order (indirect) impacts can often be more consequential than the first order impacts (Vanclay 2012).

Activities Comprising Social Impact Assessment

Good practice SIA includes the following activities (Vanclay and Esteves 2011; Esteves et al. 2012; Vanclay 2012). First, a thorough understanding is needed of the communities likely to be affected by the planned intervention (i.e. profiling), including a thorough stakeholder analysis to understand the differing needs and interests of the various sections of those communities. Participatory processes and deliberative spaces should be facilitated to enable community discussions about desired futures, the acceptability of likely impacts and proposed benefits, and community input into the SIA process, so that there can be a negotiated agreement with a developer based on free, prior, and informed consent. All this should lead to the identification of community needs and aspirations.

The key social issues associated with the planned intervention (the significant negative impacts as well as the opportunities for creating benefits) should be scoped. Baseline data to provide a benchmark to measure change over time is collected, and the social changes that may result from the policy, program, plan, or project are predicted. The significance of the predicted changes is established, and the likely response of various affected groups and communities is determined. Other options are examined, especially in terms of social issues, and ways of mitigating potential impacts and maximizing positive opportunities are identified.

In addition, a monitoring plan to monitor change over time is developed and an adaptive management process to address unanticipated changes is implemented. An agreement making process between the communities and the developer should be facilitated, ensuring that the principle of Free, Prior, and Informed Consent (FPIC) is observed and that human rights are respected, leading to the drafting of an Impacts and Benefits Agreement (IBA). The proponent should be assisted in the drafting of a Social Impact Management Plan (SIMP) that operationalizes all benefits, mitigation measures, monitors arrangements and governance arrangements that were agreed to in the IBA, as well as plans for dealing with any ongoing unanticipated issues as they arise.

Finally, processes should be put in place to enable proponents, government authorities, and civil society stakeholders to implement arrangements implied in the SIMP and IBA and to develop their own respective management action plans and embed them in their own organizations, establish respective roles and responsibilities throughout the implementation of those action plans, and maintain an ongoing role in monitoring. Care should be taken that the proponent has fully considered all impacts on human rights by either ensuring that human rights impacts are considered in the SIA, or that a separate human rights impact assessment is conducted. A grievance mechanism should be established, consistent with Principle 30 in the United Nations Guiding Principles (United Nations 2011) to ensure that affected people with complaints against the proponent have a mechanism by which their concerns can heard and resolved.

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Section XXI

Strategic Environmental Assessment for Wetlands

Roel Slootweg

Strategic Environmental Assessment (SEA) for Wetlands: Overview

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Roel Slootweg

Contents

Abstract

Whereas Environmental Impact Assessment is applied at the project level, the objective of Strategic Environmental Assessment (SEA) is to address the environmental consequences and stakeholder concerns in the development of policies and other high-level initiatives. Because SEA is integrated into the policy processes of individual countries, there is not one generally agreed SEA procedure. Nevertheless, SEA generally consists of the following phases: (1) visioning and objective setting; (2) technical assessment; (3) analysis and decision making; (4) monitoring and evaluation. SEA can be a standalone process alongside the regular planning process, but to influence policies effectively a stronger integration is necessary. Key factors promoting the effectiveness of SEA are a flexible

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approach that follows the decision-making process, participation of stakeholders, the political will to use SEA results, and good regulatory capacity and public relations for SEA. SEA can range from a mainly environmental focus to an integrated approach which also incorporates social and economic concerns.

Keywords
Strategic environmental assessment · Wetlands · Environmental policy and planning · Stakeholder participation · Decision-making

Introduction

SEA is "a family of tools that identifies and addresses the environmental consequences and stakeholder concerns in the development of policies, plans, programmes and other high level initiatives" (OECD 2006). Since its early beginnings, the field of environmental assessment has expanded, both in scope and application. Practitioners now recognize two levels of environmental assessment: Environmental Impact Assessment (EIA) that is applied at the level of individual projects and Strategic Environmental Assessment (SEA) which is applied to plans, programs, and policies (see Box 1).

Box 1: Defining Policies, Plans, and Programs

- **Policy**: A general course of action or proposed overall direction that a government is or will be pursuing and that guides ongoing decision-making.
- **Plan**: A purposeful forward-looking strategy or design, often with coordinated priorities, options, and measures that elaborate and implement policy.
- **Program**: A coherent, organized agenda or schedule of commitments, proposals, instruments, and/or activities that elaborate and implement policy.

Source: Compiled from Sadler and Verheem (1996) and OECD (2006)

In comparison with EIA, SEA is practiced less widely at present, but its application is rapidly catching up. Several years of practice with EIA showed that cumulative and large-scale effects are not addressed adequately at the project level by EIA. A new instrument was needed to assess such effects at the strategic level of policies, plans, and programs. Initially, SEA practice did not take off at the same rate as EIA. This is probably attributable to the more complex nature of strategic assessment, the long-standing preoccupation with economic priorities in strategic decision-making, and the perception that sufficient assessment tools already existed. By the 1980s, a distinct SEA practice was gaining momentum. Canada, New Zealand, and the Netherlands were among the first countries to develop a regulatory basis for SEA.

In the 1990s, many more developed countries embedded SEA into regulation. A recent expansion of the application of SEA is the European Union SEA Directive, which came into effect in 2006 (European Council 2001). All 25 EU member states are now faced with the legal obligation to apply SEA to plans and programs. If their own regulatory framework for SEA is not fully developed, then the EU SEA Directive applies directly. The adoption of SEA regulations is spreading rapidly.

One of the driving forces behind this growth in application is the United Nations Economic Commission for Europe (2003) Protocol on Strategic Environmental Assessment to the Convention on Environmental Impact Assessment in a Transboundary Context (the “Kiev Protocol”). The protocol is open to all UN members and was signed by 38 countries in July 2008. Interest in SEA is also sparked by the call for more holistic, integrated, and balanced strategic decision-making made in influential initiatives such as the 2002 World Summit on Sustainable Development and the Millennium Development Goals. The Millennium Development Goal 7, environmental sustainability, is supported by a target that reads like a SEA mission, namely, “integration of the principles of sustainable development into country policies and programs to help reverse the loss of environmental resources.”

Current Thinking

The practice of SEA is less easily demarcated than that of EIA. Some countries, especially those with a strong planning tradition, had SEA-like assessment even before the term itself came into use. There are now a large number of assessment tools in planning that do not necessarily carry the label SEA but have strong similarities. At least fifteen different terms for SEA-type approaches exist (examples include integrated assessment, country environmental analysis, integrated trade assessment, poverty and social analysis, sustainability appraisal, strategic environmental analysis). However, the fundamental differences between approaches are fewer than might be assumed from existing publications. In 2006, the SEA Task Force of the OECD Development Assistance Committee brought together representatives of a wide range of countries and international organizations with SEA experience. In the process of drafting SEA guidance, this diverse group also adopted a shared definition of SEA, appearing at the top of this text.

There is no generally agreed SEA procedure as such, no “one-size-fits-all” approach. The principal reason is that good practice SEA should ideally be fully integrated into a planning (or policy development) process. Since planning processes differ widely, there is no typical sequence of procedural steps in SEA. Even when SEA is captured in a formal procedure in legislation (e.g., the SEA Directive of the European Union), there will still be great differences in how, when, and with whom the SEA activities are undertaken. However, there is general agreement about the activities that make up a SEA process. There is a logical sequence to these activities, but logic is not the only, nor necessarily the dominant, principle governing a given planning process. Realistically, the activities outlined here may take more or less effort and may follow each other sequentially or not, and some may be repeated or combined.

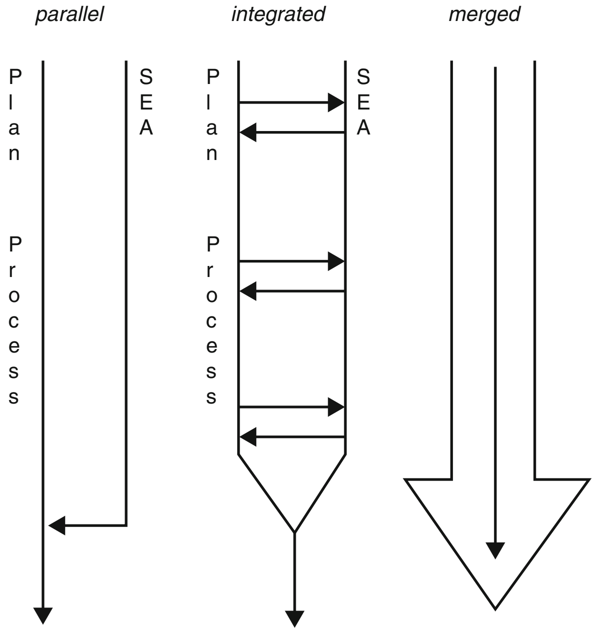
- First phase – creating transparency and joint objective setting
  – Announce the start of the SEA and assure that relevant stakeholders are aware that the process is starting.
  – Bring stakeholders to develop a shared vision on (environmental) problems, objectives, and alternative actions to achieve these.
  – Check in cooperation with all agencies whether objectives of the new policy or plan are in line with those in existing policies, including environmental objectives (consistency analysis).
- Second phase – technical assessment
  – Make clear terms of reference for the technical assessment, based on the results of stakeholder consultation and consistency analysis.
  – Carry out a proper assessment, document its results, and make these accessible for all.
  – Organize effective quality assurance of both SEA information and process.
- Third phase – use information in decision-making
  – Bring stakeholders together to discuss results and make recommendation to decision-makers.
  – Make sure any final decision is motivated in writing in light of the assessment results.
- Fourth phase – post-decision monitoring and evaluation
  – Monitor the implementation of the adopted policy or plan, and discuss the need for follow-up action (OECD 2006a).

Parallel to or Integrated Within a Planning Process?

A key factor that determines what a SEA process will actually look like for a given application is the degree of integration of the SEA activities into the planning process itself. Traditionally, SEA was often applied as a stand-alone series of activities, in parallel to planning (left hand diagram in Fig. 1). This might be a good way to build experience with SEA, but it is less effective in influencing the plan, program, or policy. By the time results of the assessment are provided to the planning team and the public, a plan strategy has already been developed. Time and resources have been committed to this strategy, and major deviations from it are not likely to be welcome. This does not mean that a stand-alone SEA cannot improve a plan at all. Even when decisions have already been made, SEA can play a meaningful role, for example, to decide on necessary monitoring measures and mitigating actions, as a reference for evaluating plan implementation outcomes, or to set the agenda for future policies and plans.

However, when SEA is more integrated, it has better scope to influence planning (middle diagram in Fig. 1). In this form of integration, the planning and SEA activities are distinct but feed each other at different stages in the process. Taking

Fig. 1 Combinations of SEA and planning process (Based on original figure in Slootweg et al. 2009; used with permission from Cambridge University Press)



integration further, the SEA and planning process could also be merged (right most diagram in Fig. 1). The key SEA activities are then no longer separate from the planning process but an integral part of it. This means that critical environmental issues are still identified, and alternative strategies will be developed and assessed, but there will not be a SEA team or SEA report as such. Instead, the assessment work continually informs the planning process, and communication and participation are organized at key moments around assessment and planning issues. A merged SEA may well be the ideal, but this approach does require willing planners that consider environmental concerns a priority. It is also more difficult to regulate, to apply checks and balances that ensure a minimum assessment level.

One of the key questions for the integration of SEA and planning is when the assessment process should start. The answer to this depends on whether SEA is seen as a plan development tool or more as an impact assessment tool. If it is seen as a plan development tool, the SEA process needs to start early on, before the policy proposals exist. SEA then assists in the analysis of the problems that the plan needs to solve and contributes to the development of proposals. If the SEA starts after the policy proposals have been developed, there is a stronger focus on impact assessment and the reactive identification of alternatives to avoid the negative impact identified. Examples from both approaches can be found in practice, but the assessment tool approach is probably more common. Again, while a SEA will be more effective with an earlier start, this is difficult to regulate. That stage of the planning process is often an in-house phase, difficult to access unless planners themselves choose to make use of SEA at that time.

SEA Is Not EIA

It is important to point out that SEA is not EIA, because it is necessarily different in nature. At project level, decision-making is about a concrete set of activities that makes up a specific development proposal. EIA concentrates on the activity-effects relationships (Herrera 2007). Strategic decision-making is less about the concrete activities that will follow from the plan, as it focuses more on identifying and assessing and comparing the different ways in which the plan can achieve its objectives. Also, the process of designing and approving a project is more amenable to linear structuring and simplification than the commonly more changeable and politically charged development of a plan or policy. As EIA aims at better projects, SEA aims at better strategies, ranging from legislation and countrywide development policies to more concrete sector and spatial plans. Ideally, SEA is applied at each planning tier, and higher level SEAs inform those at a less strategic level so that there is no overlap in the assessments. In theory, if not always in practice, the process starts with a policy broadly describing objectives and setting the context for proposed actions, usually with a sectoral or geographic scope. Policy objectives are then translated into an action plan and further operationalized in programs; and actual implementation is done through projects.

Figure 2 shows an example of such tiering for the flood management planning in the Netherlands. A national policy and strategy on water management provided the framework for decision-making in the water sector. SEA was conducted for the national plan: space for rivers. An EIA was conducted for the design of interventions

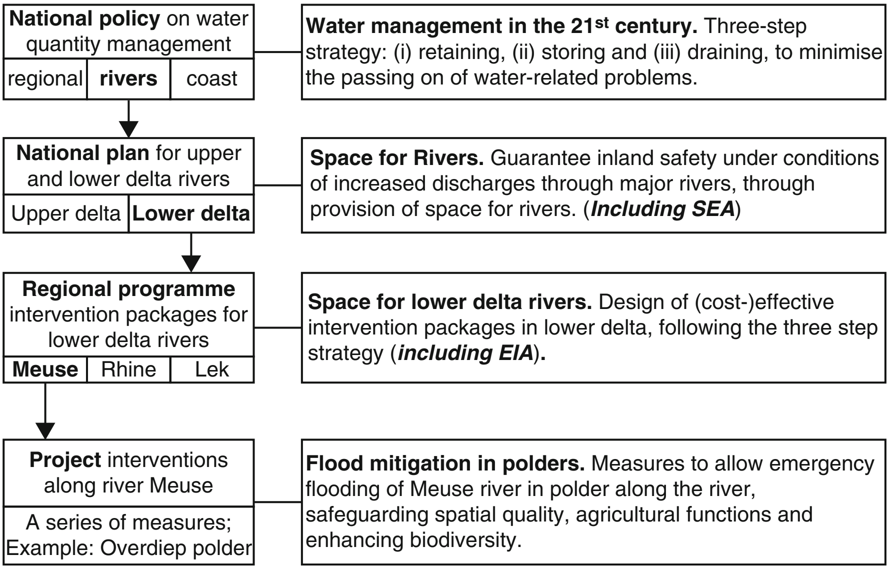


Fig. 2 Hierarchy of policies, plans, and programs: an example from the Netherlands (Based on original figure in Slootweg et al. 2009; used with permission from Cambridge University Press)

Table 1 Characteristics of SEA and EIA compared

SEA	EIA
Takes place at earlier stages of the decision-making cycle	Takes place at the end of the decision-making cycle
Considers broad range of potential alternatives	Considers limited number of feasible alternatives
Early warning of cumulative effects	Limited review of cumulative effects
Emphasis on meeting objectives and maintaining systems	Emphasis on mitigating and minimizing impacts
Broader perspective and lower level of detail to provide a vision and overall framework	Narrower perspective and higher level of detail
Multistage process, continuing and iterative, overlapping components	Well-defined process, clear beginning and end
Focuses on sustainability agenda and sources of environmental deterioration	Focuses on environmental standards and norms and symptoms of environmental deterioration

for the River Meuse. If it is undertaken in this tiered manner upstream from project considerations, SEA can help streamline EIA processes. The SEAs will consider broader environmental issues likely to be common to multiple project initiatives in a sector or in a region and are able to look at cumulative effects. This allows the subsequent EIA processes to concentrate on impacts specific to individual proposals, which improves efficiency and effectiveness of the overall process.

Another aspect in which SEA is different from EIA is the more expansive spatial and temporal horizons that are addressed. Where EIA (and earlier SEA approaches) addresses issues that could be expected in the intervention area within a relatively short time, the earlier application of environmental assessment in the decision-making hierarchy leads to more broader spatial and time horizons, i.e., looking at effects elsewhere (such as in other countries) and later (effects on future generations). Table 1 sets out the key difference between SEA and EIA.

Effectiveness of SEA

Concretely, then, SEA improves planning by:

(i) Structuring the public and government debate in the preparation of policies, plans, and programs
(ii) Feeding this debate through a robust assessment of the environmental consequences and their interrelationships with social and economic aspects
(iii) Ensuring that the results of assessment and debate are taken into account during decision-making and implementation

By doing this, SEA promises to improve planning by increasing the quality of the plans themselves, the credibility of decision-making, and thereby the support for the plan implementation (see Box 2). However, while the practice of SEA has expanded rapidly, research into this practice has been much slower to develop. Seminal studies that took a broad-based view of SEA practice were published (Sadler and Verheem 1996; Therivel and Partidario 1996; Dalal-Clayton and Sadler 2005), but apart from these, analyses of SEA practice are fragmented and often limited to a small selection of case studies. As a result, it is not very clear whether and under what conditions SEA can deliver what it promises. Many of the studies that have looked at how SEA is working on the ground have focused on the procedural and legal requirements of country or regional SEA systems. Such studies (e.g., Chaker et al. 2006) tend to show a wide variety of SEA systems that have been put in place. They might observe that SEA is generally initiated by the proponent of the planning process, that review of the SEA is delegated to an independent body in some countries, or that participation requirements are very limited in some places and more comprehensive in others.

Box 2 Advantages of SEA to Decision-Makers (Based on CBD 2006)

SEA can offer the following advantages to decision-making:

- Enhanced credibility of their decisions in the eyes of stakeholders
- More knowledge of the social feasibility of a decision, thus avoiding resistance from unhappy local groups, bad image for planners, useless mitigating measures, and simply missing the bigger picture
- Improved economic efficiency because potential environmental stumbling blocks for economic development are better understood
- Bringing promising alternatives into focus
- A better understanding of the cumulative impact of a series of smaller projects, thus preventing costly and unnecessary mistakes
- Better insight in the trade-offs between environmental, economic, and social issues, enhancing the chance of finding win-win options

What Is Needed for Effective SEA?

Despite the limited empirical evidence, there is a reasonable consensus among those working in the field about how SEA should be undertaken for it to be effective. In 2002, the International Association for Impact Assessment published the SEA performance criteria (see Box 3), which were the result of a wide debate among association members. These are still considered a benchmark for SEA practice, although it is important to note that thinking and publishing on effectiveness criteria have generally been dominated by authors from a limited range of countries and are mostly based on experiences in Europe. The studies into SEA effectiveness have brought certain performance criteria more directly into focus. For example, different effectiveness studies show the importance of flexible SEA that follows the decision-making process as well as the value of stakeholder participation. Others emphasize the importance of political will to use the results of the assessment, which can follow from regulatory requirements as well as good SEA public relations.

Box 3: Strategic Environmental Assessment Performance Criteria (IAIA 2002)

STRATEGIC ENVIRONMENTAL ASSESSMENT Performance Criteria

A good-quality Strategic Environmental Assessment (SEA) process informs planners, decision makers and affected public on the sustainability of strategic decisions, facilitates the search for the best alternative and ensures a democratic decision making process. This enhances the credibility of decisions and leads to more cost- and time-effective EA at the project level. For this purpose, a good-quality SEA process:

Is integrated	• Ensures an appropriate environmental assessment of all strategic decisions relevant for the achievement of sustainable development. • Addresses the interrelationships of biophysical, social and economic aspects. • Is tiered to policies in relevant sectors and (transboundary) regions and, where appropriate, to project EIA and decision making.
Is sustainability-led	• Facilitates identification of development options and alternative proposals that are more sustainable[1].
Is focused	• Provides sufficient, reliable and usable information for development planning and decision making. • Concentrates on key issues of sustainable development. • Is customized to the characteristics of the decision making process. • Is cost- and time-effective.
Is accountable	• Is the responsiblity of the leading agencies for the strategic decision to be taken. • Is carried out with professionalism, rigor, fairness, impartiality and balance. • Is subject to independent checks and verification • Documents and justifies how sustainability issues were take into account in decision making.
Is participative	• Informs and involves interested and affected public and government bodies throughtout the decision making process. • Explicity addresses their inputs and concerns in documentation and decision making. • Has clear, easily-understood information requirements and ensures sufficient access to all relevant information.
Is iterative	• Ensures availability of the assessment results early enough to influence the decision making process and inspire future planning. • Provides sufficient information on the actual impacts of implementing a strategic decision, to judge whether this decision should be amended and to provide a basis for future decisions.

[1] i.e., that contributes to the overall sustainable development strategy as laid down in Rio 1992 and defined in the specific policies or values of a country

Scope of Assessment

Recent reviews of practices in SEA and related approaches show there is an emerging spectrum or "continuum" of interpretation of the scope of assessment. At one end of the continuum, the focus is mainly environmental (what we might call "conventional" SEA). It is characterized by the goal of mainstreaming and upstreaming environmental considerations into strategic decision-making at the earliest stages of planning processes to ensure they are fully included and appropriately addressed. The 2001 SEA Directive of the European Union (European Council 2001) is an example of this approach.

At the other end of the continuum is a more holistic and comprehensive approach which aims to assess environmental, social, and economic concerns in a more integrated manner and involves possible trade-offs between these considerations in strategic decision-making at the earliest stages of planning processes.

The ends of the continuum are characterized by Table 2, not just in terms of the dimensions that are considered but also in terms of the time and space horizon that an assessment can cover. The upper left hand corner represents conventional EIA, which concentrates on immediate environmental effects within a short time horizon. Integrated EIA covers the entire upper row in the table (the cross-hatched area). Extending the time and spatial horizon is especially important in SEA. The strategic nature of the decisions that SEA supports necessitates a broader view in order to fully grasp the consequences and alternative options (Partidario 2007). For a national energy policy, for example, the SEA will need to explore the global implications as well as the national ones. Full sustainability assessment SEA would cover the entire range: addressing environmental, social, and economic effects, those that are immediate as well as those that manifest themselves over an extended time horizon, and both at the directly impacted area as well as beyond (cross-hatched and diagonally hatched areas).

Of course, full incorporation of biodiversity in environmental assessment will in many cases require longer geographical and time horizons. Biodiversity very often depends on geographically different areas (e.g., migratory birds or fish), while biodiversity is important for the maintenance of life support systems for future generations to have at least the same quality of living as present generations have, while it also holds unknown potential for the future. The emergence of ecosystem services as a means to link biodiversity to stakeholders requires a more integrated approach including biophysical as well as social and economic aspects.

Table 2 Simplified characterization of the continuum of impact assessment scope

Scope >> Time/space horizon	Environmental (ecological) aspects	Social aspects	Economic aspects
Here and now	Conventional EIA	ESIA	Integrated EIA
Elsewhere and later	Conventional SEA	Broad SEA	Sustainability assessment

There are clear advantages to a comprehensive and integrated assessment. For the decision-makers and all stakeholders, the assessment information is conveniently brought together. This can concentrate the debate there where the real trade-offs are to be made between beneficial and detrimental effects. However, such assessment is methodologically more complex and requires intense cooperation between assessment professionals, across disciplinary and organizational boundaries. This is not easy to achieve, especially in settings where such cooperation is not commonplace. It could also be argued that in an aggregate assessment the environmental effects may get snowed under the economic and social ones, which tend to be of more interest to decision-makers. Integrated assessment can furthermore obscure trade-offs that are being made against the environment. On the other hand, when biodiversity is translated into ecosystem services and is assigned its real value in social or economic terms, the case for biodiversity can become stronger, and decision-making can be significantly influenced.

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Wetland Triggers for Strategic Environmental Assessment

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Roel Slootweg

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Abstract

Wetland triggers in Strategic Environmental Assessment - factors that signal the need for special attention to wetlands - are (1) ecosystem services provided by the affected area; (2) activities that may be direct drivers of change in ecosystem services; (3) indirect drivers of change. For Trigger (1), the ecosystem services (including biodiversity) need to be assessed and valued. For Triggers (2) and (3), the drivers of change leading to biophysical change need to be identified and the potential impact estimated. In case of a combination of (1) and (2), a combined assessment is done which allows for greater detail in defining the expected impacts. The article presents an overview of the procedures for incorporating wetlands and biodiversity in the SEA process.

Keywords

Strategic environmental assessment · Wetlands · Triggers for SEA · Ecosystem services · Biodiversity · Decision-making

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Introduction

To be able to make a judgment if a policy, plan, or program has potential impacts on wetlands, two elements are of overriding importance (Slootweg et al. 2006, 2010; Ramsar Convention 2010):

(i) Affected area and ecosystem services linked to this area
(ii) Types of planned activities that can act as driver of change in ecosystem services

When any one or a combination of the conditions below applies to a policy, plan, or program, special attention to wetlands is required in the SEA of this policy, plan, or program:

- **Important ecosystem services**. When an area affected by a policy, plan, or program is known to provide one or more important ecosystem services, these services and their stakeholders should be taken into account in an SEA. Geographical delineation of an area provides important information as it is possible to identify the ecosystems and land use practices in the area and identify ecosystem services provided by these ecosystems or land use types. For each ecosystem service, stakeholder(s) can be identified who preferably are invited to participate in the SEA process. Wetland-related policies and legislation can be taken into account.
- **Interventions acting as direct drivers of change**. If a proposed intervention is known to produce or contribute to one or more drivers of change with known impact on wetland-related ecosystem services (see Box 1), special attention needs to be given to wetlands. If the intervention area of the policy, plan, or program has not yet been geographically defined (e.g., in the case of a sector policy), the SEA can only define wetland impacts in conditional terms: impacts are expected to occur in case the policy, plan, or program and will affect certain types of wetlands. If the intervention area is known, it is possible to link drivers of change to ecosystem services and their stakeholders.
- **Interventions acting as indirect drivers of change**. When a policy, plan, or program leads to activities acting as indirect driver of change (e.g., for a trade policy, a poverty reduction strategy, or a tax measure), it becomes more complex to identify potential impacts on wetlands. In broad terms, wetland attention is needed in SEA when the policy, plan, or program is expected to significantly affect the way in which a society:
  - Consumes products derived from wetlands or products that depend on wetland services for their production
  - Occupies areas of wetland
  - Exploits wetland resources and services

Box 1: Direct Drivers of Change

Direct drivers of change are human interventions (activities) resulting in biophysical and social/economic effects with known impacts on biodiversity and associated ecosystem services.

Biophysical changes known to act as a potential driver of change comprise:

- *Land conversion*
- *Fragmentation* or isolation
- *Extraction of living organisms*
- *Extraction of minerals, ores, and water*
- *Wastes* (*emissions, effluents, solid waste*) *or other chemical, thermal, radiation, or noise inputs*
- *Disturbance of ecosystem composition, structure, or key processes*

The Assessment Framework

Figure 1 depicts a conceptual framework for wetland triggers in SEA, describing pathways of activities to impacts. It positions the wetland triggers, i.e., (1) affected ecosystem services and activities producing direct (2) or indirect (3) drivers of change in ecosystem services.

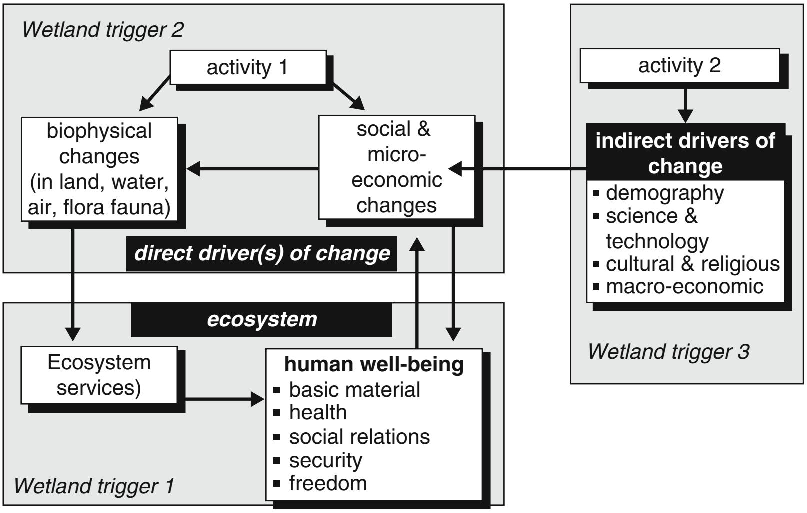

Fig. 1 Assessment framework. For explanation, see text (Figure based on author's original figure in Slootweg et al. 2009)

Activities resulting from a policy, plan, or program lead to biophysical changes and/or social/economic changes (activity 1 in Fig. 1). Social/economic changes influence human well-being directly, but some of these changes may in turn also lead to biophysical changes (e.g., in-migration of people leads to occupation of land). Within their spatial and temporal range of influence, biophysical changes may influence the composition or structure of wetlands or influence key processes maintaining these wetlands. Activities resulting in this type of biophysical changes are referred to as direct drivers of change. The ecosystem services provided by impacted wetlands may be affected, thus affecting groups in society who depend on these services for their well-being. People may respond to changes in the value of ecosystem services and act accordingly, thus leading to new social/economic changes. Good participatory scoping and application of the best available scientific and local knowledge result in the identification of most relevant impacts and associated cause-effect chains that need further study in the SEA.

Identifying impacts on ecosystem services resulting from indirect drivers of change (activity 2 in Fig. 1) is a more challenging task. The links between indirect and direct drivers of change are not yet fully understood.

Identifying Potential Wetland Impacts Through Wetland Triggers

Trigger 1: The area influenced by the policy, plan, or program provides important wetland-related ecosystem services.

Focus: Area-oriented policies, plans or programs without precisely defined activities. Wetlands within the area can be described in terms of ecosystem goods and services for the development and/or well-being of people and society. The maintenance of biodiversity (for future generations or because biodiversity is considered to have an intrinsic value) is often emphasized as a special ecosystem service, described in terms of conservation status of ecosystem, habitats, and species, possibly supported by legal protection mechanisms.

This trigger is often associated with land use planning/spatial planning where interventions are potentially wide ranging and the objective is to develop suitable land uses in line with the natural conditions.

Summary of Procedure

- Identify wetland types in the area to which the policy, plan, or program applies. Identify and map ecosystem services provided by these wetlands.
- Identify which groups in society have a stake in each ecosystem service; invite such stakeholders to participate in the SEA process. Identification and valuation of ecosystem services is an iterative process initiated by experts (ecologists, natural resources specialists) but with stakeholders playing an equally important role. The frequency of reliance on ecosystem goods or services should not necessarily be used as an indication or measure of their value because ecosystem services on which local communities rely even on an occasional basis can be

critical to the resilience and survival of these communities during surprise or extreme natural conditions.
- For absent stakeholders (future generations), identify important protected and non-protected biodiversity which is a representative of species, habitats, and/or key ecological and evolutionary processes (e.g., by applying systematic conservation planning or similar approaches).
- Ecosystem services identified by experts but without actual stakeholders may represent an unexploited opportunity for social, economic, or ecological development. Similarly, ecosystem services with conflicting stakeholders may indicate overexploitation of this service representing a problem that needs to be addressed.

Trigger 2: The policy, plan, or program is concerned with interventions producing direct drivers of change.

Focus: As explained above, interventions resulting from a policy, plan, or program can directly, or through socioeconomic changes, lead to biophysical changes that affect wetlands and their services. Impacts on ecosystem services can only be defined as potential impacts, since the location of the intervention or the area where its influence is noticed may not be known.

This trigger is often associated with policies, plans, or programs without defined geographical area of intervention, such as sectoral policies or policies, plans, or programs producing social/economic drivers of change which cannot be geographically demarcated.

Summary of Procedure
- Identify drivers of change, i.e., activities leading to biophysical changes known to affect wetlands (see Box 1).
- Within the administrative boundaries (province, state, country) to which the policy, plan, or program applies, identify wetlands sensitive to the expected biophysical changes. The SEA needs to develop a mechanism to avoid, mitigate, or compensate potential negative impacts to these wetlands including the identification of less damaging alternatives.

Triggers 1 and 2 Combined: The policy, plan, or program concerns activities producing direct drivers of change in an area with important wetland-related ecosystem services.

Focus: Knowledge of the nature of interventions and the area of influence allows relatively detailed assessment of potential impacts by defining changes in composition or structure of wetlands or changes in key processes maintaining wetlands and associated ecosystem services.

This combination of triggers is often associated with SEAs carried out for programs (resembling complex, large-scale EIAs). Examples are detailed spatial plans, program level location, and routing alternatives or technology alternatives.

Summary of procedure: The procedure is a combination of the procedures for Triggers 1 and 2, but the combination allows for greater detail in defining expected impacts:

- Identify direct drivers of change and define their spatial and temporal range of influence.
- Identify wetlands lying within this range of influence (in some cases, species or genetic level information may be needed).
- Describe effects of identified drivers of change on identified wetlands in terms of changes in composition or structure of biodiversity or changes in key processes responsible for the creation or maintenance of wetlands.
- If a driver of change significantly affects either composition, structure, or a key process, there is a very high probability that ecosystem services provided by the wetland will be significantly affected.
- Identify stakeholders of these ecosystem services and invite them to participate in the process. Take into account the absent (future) stakeholders.

Trigger 3: The policy, plan, or program is concerned with interventions affecting indirect drivers of change.

An example of such a trigger would be trade liberalization in the agricultural sector and the effects this might have on wetlands. Baseline conditions, trends, and characteristics of the production and socioeconomic systems determine whether indirect consequences will affect wetlands. This SEA works with a combination of economic modeling studies, empirical evidence from literature, case study analysis, and causal chain analysis. Impact is described in very broad terms, mainly as changes in surface area and species richness. More research and case material are needed to elaborate this biodiversity trigger.

Figure 2 provides a summary of the way in which potential wetland impacts of a policy, plan, or program can be identified. It is derived from the CBD Guidelines and

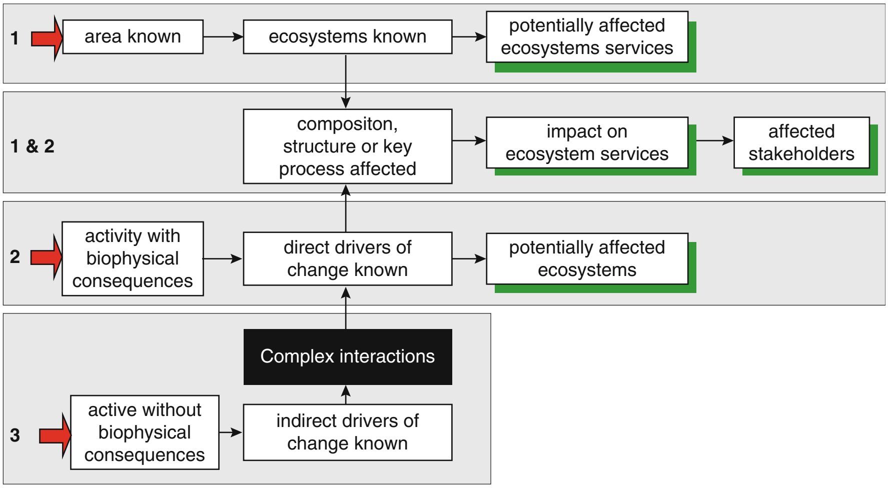

Fig. 2 Summary overview of procedure to define biodiversity impacts starting with one or a combination of biodiversity triggers (Figure based on author's original figure in Slootweg et al. 2009)

considers biodiversity in its broadest meaning, thus including wetlands. It starts with the identification of potential biodiversity triggers in the policy, plan, or program to be analyzed, including (i) an area with valued ecosystem services, (ii) activities affecting direct drivers of change, (iii) activities affecting indirect drivers of change, or a combination of (i) and (ii) where activities with known drivers of change influence a known area with valued ecosystem services. If one of these triggers is present in the policy, plan, or program, the flow chart shows the type of information that can and should be obtained in the SEA process. The link between indirect and direct drivers of change is characterized by complex interactions, many of which are presently subject to intense research efforts worldwide.

Table 1 presents an overview of the conditions under which a strategic environmental assessment should place particular attention to biodiversity issues and how they should be addressed.

Table 1 When and how to address biodiversity in SEA

Biodiversity triggers in policy, plan, or program	When is biodiversity attention needed	How to address biodiversity issues
Trigger 1	*Does the policy, plan, or program influence:*	*Area focus*
Area known to provide important ecosystem services	Important ecosystem services, both protected (formal) or non-protected (stakeholder values)	Systematic conservation planning for non-protected biodiversity
	Areas with legal and/or international status	Ecosystem services mapping
	Important biodiversity to be maintained for future generations	Link ecosystem services to stakeholders
		Invite stakeholders for consultation
Trigger 2	*Does the policy, plan, or program lead to*:	*Focus on direct drivers of change and potentially affected ecosystem*
Policy, plan, or program affecting direct drivers of change	Biophysical changes known to significantly affect ecosystem services (e.g., land conversion, fragmentation, emissions, introductions, extraction, etc.)	Identify drivers of change, i.e., biophysical changes known to affect biodiversity
(i.e., biophysical and non-biophysical interventions with biophysical consequences known to affect ecosystem services)	Non-biophysical changes with known biophysical consequences (e.g., relocation/migration of people, migrant labor, change in land use practices, enhanced accessibility, marginalization)	Within administrative boundaries to which the policy, plan, or program applies, identify ecosystems sensitive to expected biophysical changes

(*continued*)

Table 1 (continued)

Biodiversity triggers in policy, plan, or program	When is biodiversity attention needed	How to address biodiversity issues
Combined triggers 1 and 2	Combination of triggers 1 and 2 above	*Knowledge of intervention and area of influence allows prediction of impacts on composition or structure of biodiversity or on key processes maintaining biodiversity*
Interventions with known direct drivers of change affecting area with known ecosystem services		Focus on direct drivers of change, i.e., biophysical changes known to affect biodiversity. Define spatial and temporal influence
		Identify ecosystems within range of influence
		Define impacts of drivers of change on composition, structure, or key processes
		Describe affected ecosystems services and link services to stakeholders
		Invite stakeholders into SEA process
		Take into account the absent (future) stakeholders
Trigger 3	*Are indirect drivers of change affecting the way in which a society*:	*More research and case material needed*
Policy, plan, or program affecting indirect drivers of change but without direct biophysical consequences	Produces or consumes goods	MA methodology potentially valuable to identify linkages between indirect and direct drivers of change
	Occupies land and water or	
	Exploits ecosystem services	

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Strategic Environmental Assessment for Wetlands: Resilience Thinking

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Mike Jones

Contents

Abstract

Conventional strategic environmental assessment (SEA) is based on the assumption that ecosystems and people behave predictably and that cause and effect have a direct and linear relationship. Recognition of wetlands as complex adaptive systems has profound implications for wetland management and requires a paradigm shift from command-and-control management to management for complexity and uncertainty. Resilience assessment as a potential alternative assessment method takes the uncertainty and complexity of ecosystems into account and improves the manner in which SEA is undertaken. This chapter provides a brief description of resilience theory, the resilience assessment process, and some general resilience perspectives on the use of wetlands. The main steps in resilience assessment are (1) System description; (2) General resilience assessment; (3) Specified resilience assessment; (4) Governance assessment; and (5) Strategy and management planning. The assessment considers the biophysical, social, and economic domains of the system, its subsystems, and the environment

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within which it is located. An assessment of the governance structures that regulates natural resource use in the system may be indicated by the number, nature, and severity of issues identified in the initial description of the system. The assessment process is iterative and allows for a social learning process to develop a deeper understanding of the system's dynamics and an adaptive management strategy. Resilience assessment is a strategy development tool, providing a big picture understanding of a system and the possible outcome of the selected direction for change or no change. It helps managers decide whether to enhance the resilience of the existing system or to transform the system and develop its resilience.

Keywords
Resilience · Complex adaptive systems · Adaptive cycle · Panarchy · Wetlands SEA

Introduction

Conventional strategic environmental assessment (SEA) is based on the assumption that ecosystems and the people who use them behave predictably and that cause has a direct and linear relationship with effect. Environmental managers know this assumption to be false, but until recently, there has not been an alternative assessment method that takes the uncertainty and complexity of ecosystems into account. Slootweg and Jones (2011) introduced resilience assessment as a potential tool for improving the manner in which SEA is undertaken. This text provides a brief description of resilience theory, the resilience assessment process, and some general resilience perspectives on the use of wetlands.

Resilience Theory

Resilience theory in relation to ecological systems was first proposed by Holling (1973) and subsequently extended to include social, ecological, and economic systems (Holling 2001, 2004). Resilience is one property of complex adaptive systems (Meadows 2008), others being the capacity to self-organize into systems and the tendency to become increasingly complex and hierarchical. It is the self-organizing or emergent properties of living systems that make them complex and unpredictable. Resilience theory is one aspect of complexity science, providing three metaphorical models that enable understanding of change processes within systems and across nested systems at multiple levels of scale. Understanding change processes in complex adaptive systems enables environmental managers to assess the resilience of a system to specified shocks, the capacity of a system for transformational change from one state to another, and to decide whether to enhance the capacity of the system to bounce back from a disturbance event or use a disturbance event as an opportunity for transformational change. These two aspects of resilience

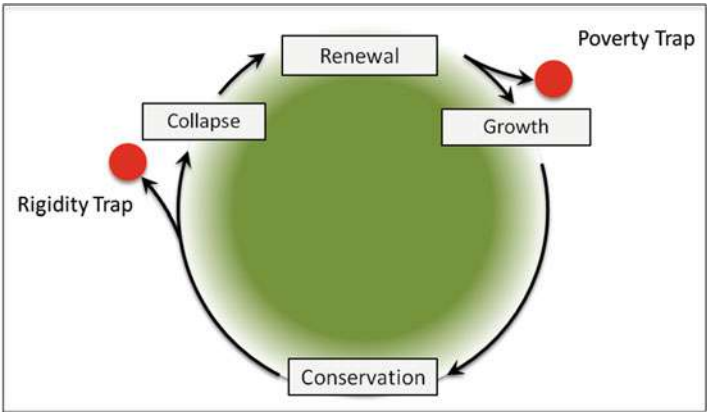


Fig. 1 The adaptive cycle illustrates how change occurs over time. It has relatively long growth and conservation phases followed by rapid phases of collapse and renewal. A rigidity trap is in essence an over-resilient system that cannot or will not change when change is necessary. A poverty trap develops when the potential of a system for change has been severely eroded and the connections necessary for internal regulation have been broken

are commonly known as the ability to bounce back and the ability to bounce forward (Davoudi et al. 2012).

Adaptive Cycle

The adaptive cycle (Holling 2001, 2004) is defined by three axes:

1. Potential: represents the ecological, economic, and social wealth of the system – the components and materials necessary for the system to either bounce back from a disturbance event or bounce forward into a new system state.
2. Connectedness: represents the feedback relationships between the different components of the system and determines the system's ability to self-regulate in response to events in its environment. Connectedness reflects the flexibility or rigidity of the system in response to disturbance. Over-connected systems are vulnerable to disturbance as the shock may be rapidly transmitted throughout the system, precipitating a collapse.
3. Resilience: the interaction between potential and connectedness determines the resilience of the system.

The adaptive cycle (Fig. 1) operates in two phases: a long and generally slow growth phase from exploitation to conservation and a rapid collapse and reorganization phase that creatively destroys the mature system, reorganizing its components either for a new phase of growth or for transformational change.

Panarchy

The Panarchy (Holling 2001, 2004) represents the hierarchical nature of complex adaptive systems, comprising nested and interconnected adaptive cycles operating at different levels of spatial and temporal scale (Fig. 2). Relatively small, fast-changing systems (fast variables) within the hierarchy exert an influence for change over the larger-scale systems within which they are embedded. Relatively large, slow-changing systems (slow variables) exert a stabilizing influence that resists the pressure for change from small, fast-changing systems. Evolutionary change occurs over time in response to fast variable influences and the opportunities for introducing novelty presented by systemic collapse. The collapse, renewal, and change processes of complex adaptive systems are an integral part of life, development, and evolution.

Thresholds

The thresholds model (Scheffer et al. 2001) describes how systems can shift from one stable state to another when they are subject to a disturbance event after they have lost resilience. The model (Fig. 3) uses a ball to represent the system and a basin (the "basin of attraction") to represent the conditions under which the system remains in a particular state. A system located at the bottom of a deep basin of attraction is resilient to disturbance, while one located near the rim of the basin may

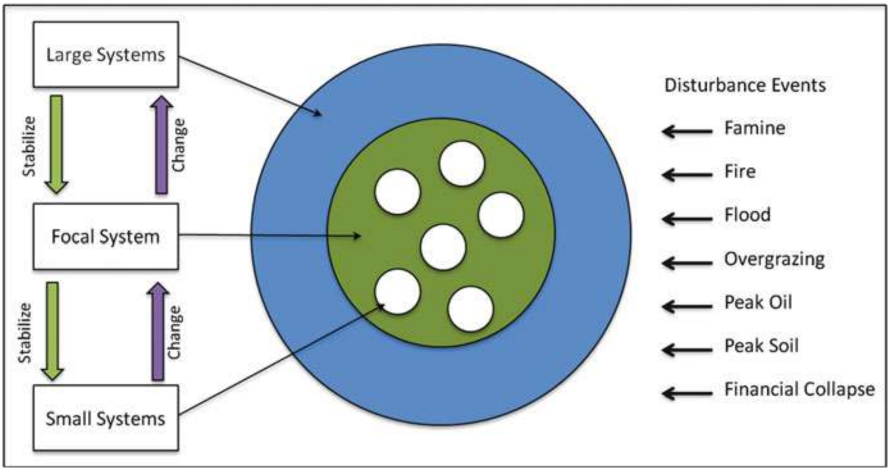


Fig. 2 Panarchy explains the interactions that occur between nested systems of different size. The focal system is the system of interest to management: small systems embedded within the focal system change relatively quickly and large systems within which the focal system is embedded are part of the larger environment. Small systems create change; large systems provide stability. When large systems collapse, the focal system and subsystems will change in a "cascade of collapse" creating opportunities for transformational change or "bounce forward"

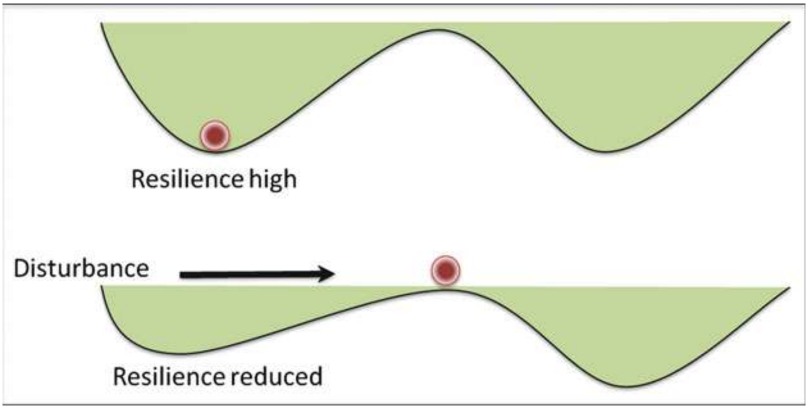


Fig. 3 The ball and basin metaphor is a simple way of understanding the manner in which systems (the *red ball*) can switch from one stable state to another. Management that reduces the resilience of a system increases the likelihood that it will cross the threshold or tipping point from one state to another as a consequence of a disturbance

enter an alternate state on being disturbed. The essence of resilience-based stewardship is to identify thresholds, which, when crossed, would place the system in an undesirable state from a human perspective.

Resilience Assessment

Resilience assessment (Walker and Salt 2012) is a process that enables natural resource stewards to evaluate the resilience of particular parts of a system to particular disturbance events (specified resilience) and to evaluate the general resilience of a system to unspecified disturbance events (general resilience). The resilience assessment process has parts common to other assessment processes but there are some significant differences (Therivel et al. 2016):

1. The assessment of changes likely to occur as a consequence of disturbance is based on the history of past events, current status, and expected future trends of the system.
2. The assessment is carried out as part of a strategic, multi-stakeholder process that integrates different kinds of formal and informal knowledge and develops a common understanding of dynamics of key parts of the system within which a plan will be implemented.
3. The assessment includes identification of evolving long-term changes and thresholds of potential undesirable change. The existing conditions may be evaluated as undesirable in which case a plan for transformative change is required.

4. Strategic alternatives are considered as an integral part of the assessment process using scenario development based on a model of identified key system dynamics and discussion of acceptable change. The implementation plan is selected from strategic alternatives that were assessed for positive and negative consequences.
5. Comprehensive participation in the assessment process ensures that the strategy is tested against thresholds and negative consequences on social groups that have traditionally been disadvantaged.
6. The selected implementation plan is based on a long-term model of change. Only strategies that deliver net sustainability benefits, taking into account the precautionary principle, would proceed. The plan is implemented adaptively to test the assumptions contained in the change models and measure positive and negative outcomes.
7. Governance is related to the social and ecological scales at which the plan operates with delegation of authority and responsibility to public and private entities with supportive institutions that create a nested hierarchy of learning organizations.

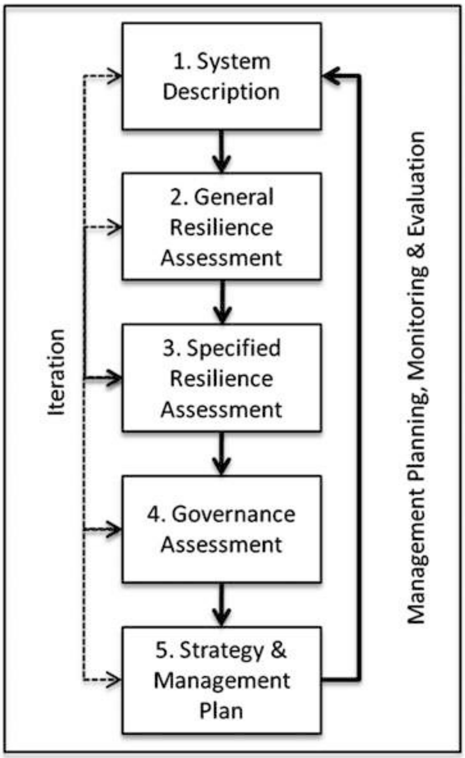
The primary components of a resilience assessment process are shown in Fig. 4, illustrating its iterative nature and the mechanism for social learning (Hoverman et al. 2011; McCarthy et al. 2011) that must occur during plan implementation to develop deeper understanding of a system's dynamics. Plan implementation without adaptive management is the equivalent of management by belief, rather than scientific management to test plan assumptions in a learning environment where success and failure count equally as valid information about system processes.

The collection and organization of information in the background part of the assessment lead to a preliminary description of the system to be managed including its major components, the history of disturbance events, and the links between the focal system, its subsystems, and the environment within which it operates. It provides a preliminary mental model of system process and change based on the panarchy (Fig. 2).

The assessment of general resilience provides an indication of the system's overall resilience to any kinds of disturbance, including novel disturbances such as climate change. It enables managers to identify the current status and trends in a set of attributes that determine the adaptability and transformability of a system. This is a relatively simple exercise that does not require the technical skills needed for an assessment of specified resilience. For the kinds of large-scale, unpredictable, and far-reaching changes that will be associated with climate change, a general resilience assessment may be enough to assess the resilience of a wetland system in relation to a SEA. The necessity of a specified resilience assessment would depend on the kind of development intervention being planned.

An assessment of a specified resilience provides an operational definition of resilience by describing the resilience of a specified system to a particular disturbance. It lays the foundation for quantitative and qualitative models of the risk of systemic collapse in the event of specified disturbance events. This part of the

Fig. 4 Flow diagram of the major components of a resilience assessment



assessment develops a model of "thresholds of potential concern" (TPC, i.e., those known or suspected thresholds of undesirable system change that managers would like to avoid) to management in the "slow variables" that provide stability to the system. The assessment considers the biophysical, social, and economic domains of the focal systems, its subsystems, and the environment within which it is located. *This is a strategic assessment to help managers decide whether to enhance the resilience of the existing system or to transform the system and develop its resilience.* In the context of SEA, proposed developments would be assessed in terms of their shock severity to the system and the system's resilience as defined by the thresholds of potential concern.

An assessment of the governance structures that regulate natural resource use in the focal system may be indicated by the number, nature, and severity of issues identified in the initial description of the focal system. In cases of conflict within complex mixtures of jurisdictions and rights, cultural differences in worldview, or inability to govern resources across multiple scales, deeper assessment may be required. In a SEA of wetlands where inflows and outflows may cross multiple jurisdictional boundaries where different laws and policies pertain to water catchments, a governance assessment would be necessary.

Once complete, the information and analyses generated by the resilience assessment are used to develop alternative strategies from which one can be selected for management planning. Resilience assessment is a strategy development tool in the true sense of the word "strategy," providing a big picture understanding of a system and the possible outcome of the selected direction for change (or no change). The basic choice for decision-making is "Do we enhance the resilience of the existing system, or do we transform the system?" "Resilience for whom?" can be a critical matter in this regard as people compete for access to the resources that sustain their livelihood. The more powerful will tend to use their power to increase their resilience at the expense of the poor. While this may be a successful short-term strategy for the powerful, it will lead to rigidity and poverty traps in the longer term that will leave all sectors of society more vulnerable to highly disruptive events like climate change.

In the event of a decision to transform, there may be several alternative structures for a new system. Scenario analysis can be used as a participatory tool to help stakeholders determine whether they want to enhance the resilience of their existing system or transform the system according to the selected vision of a desirable future. To be consistent with resilience-based stewardship, a management plan based on a resilience assessment will require monitoring of drivers and variables associated with thresholds of potential concern. The management plan may also specify experiments on small parts of the system to determine how it behaves when pushed beyond what are believed to be its limits for change before tipping into an alternate state.

Relevance of Resilience Assessment to Wetlands

Wetlands comprise physical, biological, and human components and can be readily conceptualized as complex adaptive systems that change in response to a wide range of disturbance events. Recognition of wetlands as complex adaptive systems has profound implications for wetland management (Méndez et al. 2012) and requires a paradigm shift from command-and-control management to management for complexity and uncertainty. Command-and-control management, based on the false assumption that living systems are relatively simple and predictable, is the normal approach for SEA and will tend to drive the human–wetland–water system into pathological states, characterized by more vulnerable ecosystems and rigid institutions for governance (Méndez et al. 2012).

Wetlands are inherently dynamic, acting as water storage areas during times of heavy rain, and flooding, or in the case of coastal wetlands such as mangroves and tidal marshes, on a diurnal basis with the ebb and flow of the tides. Wetland species are flexible to the alternate wet and dry within the ranges defined by daily or seasonal patterns of change. Unless materially altered in a major way by engineering structures that, for example, change river flow regimes or tidal rhythms, wetlands tend to have high resilience. Wetlands are frequently a major source of human livelihood providing grazing land, agricultural land, and areas for hunting, fishing, and gathering a range of plant material. Excessive use of either the wetland itself or the catchment in ways that alter water flow, either by drainage due to gully erosion or intensification of water inflow through catchment degradation, may cause permanent reduction in wetland structure (change in species composition and soil characteristics) and water storage function.

In resilience terminology, wetlands are "slow variables" that is large-scale slow-changing systems in the panarchy. They tend to be extensive and, apart from their seasonal or daily patterns of flooding, they change little over long periods of time, unless substantially altered by human activity. Terrestrial wetlands play critical roles in the hydrological cycle and the cycles of weathering, erosion, and deposition that provide the foundation for landscape form and function. These are both slow variables that provide the stability necessary for the resilience of a high diversity of plants, animals, and human livelihood strategies. Coastal wetlands such as marshes and mangroves provide stability against coastal erosion, tidal surges, and tsunamis, are habitat for a high diversity of animal life, and provide spawning grounds for many species of fish, including species valuable for commercial fisheries. Development activities that alter or destroy wetlands lead to cascades of collapse in subsystems that will reduce biodiversity and ecosystem function, ultimately threatening human livelihood. The stabilizing functions of wetlands have been extensively reduced through activities such as drainage for agricultural development, impoundments for hydropower and irrigation, flood control embankments, and in coastal areas through things like tourism and residential development and, in the case of mangroves, shrimp farming.

Wetlands also pose governance problems because of their connections to other parts of the system that operate at larger scales such as migratory bird species and rivers. These higher levels of scale can cross jurisdictional boundaries at levels ranging from local authority to multinational, making it difficult to match the scale of governance to the ecological scale at which some wetland components such as rivers and migratory birds operate. Scale mismatches of this kind are frequently the cause of environmental pathology (Anderies et al. 2006).

Future Challenges

There are numerous future challenges to wetlands and in creating new and more flexible approaches for their management. Global-scale changes such as those anticipated as part of the Anthropocene (Rockström et al. 2009) all have the potential to create significant disturbance events that will test the resilience of wetlands. The increased frequency of high-intensity weather events, the commoditization of nature, the finite supplies of essential resources such as fossil fuels and phosphorus against a backdrop of rising human population density, and the demand for food, fuel, fiber, and livestock feed can also create disturbance events. The continued development and use of wetlands and the hydrological systems of which they are part make wetlands increasingly vulnerable to collapse.

The appropriate management response to wetland vulnerability is to stop developing them in ways that undermine their ecological integrity and start restoring their capacity to act as a stabilizing variable for other components of the biosphere and a source of multiple ecosystem services for human livelihood. This is unlikely to happen because of the ubiquitous rigidity in the current paradigm of unlimited economic growth based on global industrialization. Current SEA practice, based on command-and-control management, is inadequate for dealing with the problem. SEA is caught in a rigidity trap and seems unable change even though change is necessary to enable society to adapt to climate and other global-scale change. This rigidity is a function of the competitive selection for fitness that affects living entities from the cell through all levels of social organization (Scheffer and Westley 2007). In simple terms, the implications of the genetic predisposition to rigidity mean that cycles of growth and collapse are inevitable and unavoidable. Once this is recognized, then collapse can be deliberately brought about through a process of sequenced transformational change. This implies that managers can detect an impending collapse before it occurs and are willing to take appropriate action. The difficulties of detecting the point at which a system will collapse and taking appropriate action to prevent the collapse were explored by Biggs et al. (2009). Detection and successful avoidance of collapse are only possible with small-scale, fast-changing systems such as fish populations in an enclosed water where local management has the authority and capacity to take prompt corrective action. Detection of tipping points in large-scale systems is difficult and, because of their size and complexity, collapse is unavoidable. The implications of this are that government laws and policies for environmental management must change from the current command-and-control paradigm to one based on the principles of subsidiarity. SEA practice needs to be transformed from a process based on the false assumption of predictability to one based on as assumption of complexity and unpredictability. Resilience assessment, adaptive management, and social learning are appropriate tools for incorporation into an improved SEA process.

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Section XXII

Economic Valuation of Wetlands

Ritesh Kumar

Economic Valuation of Wetlands: Overview

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Contents

Abstract

The continued loss and degradation of wetlands and the role of economic drivers therein, urgently call for communicating the diverse values of wetlands, and the consequences of loss of vital ecosystem services, in the language of the world's dominant economic and ecological paradigms. Economic valuation helps bridge this communication gap by enabling expression of the impact of public and private decisions on ecosystem service values in comparable metrics. A fuller and meaningful application of economic valuation assessments merits can be enabled by understanding of why valuation is needed, whose and what values are important, how to derive values and ultimately ensure integration in decision making processes. As akin to various assessment tools, economic valuation is also associated with uncertainty of various forms and levels, which need to be understood for a meaningful application. Valuation in general, and economic valuation in particular, is an evolving field, and needs to be continually enriched with better understanding of ecosystem functioning and plurality of values.

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Values · Preferences · Uncertainty · Benefit transfer · Scaling

Wetland Wise Use and Economic Values

Wise use of wetlands entails stakeholder engagement and transparency in negotiating ecosystem services trade-offs associated with various forms of wetland use in order to determine equitable conservation outcomes (Finlayson et al. 2011). As public goods, a large category of these services are not internalized in sectoral policy decisions. Economic valuation helps better decision making related to use and management of natural resources, including wetlands, by making explicit how decision making would affect ecosystem service values, and expressing these value changes in units that allow for their incorporation in decision making (Mooney et al. 2005). It is a means of communicating the value of wetland ecosystem services to different groups of people using a language that speaks to dominant economic and political viewpoints across the world (TEEB 2012).

Economic valuation forms a part of a wider set of wetland assessment tools which help describe the site drivers, pressures, and management needs. Through use of this tool, it is possible to determine the contribution that wetlands make to economies, evaluate outcomes of alternate development options related to wetlands, or analyze impacts of developmental projects on wetlands (De Groot et al. 2006). Information derived from such assessments can help improve management of wetlands by raising awareness on wetland values, create a business case for investing into wetland restoration and sustainable management, identify better resource management options for wetlands, and, most importantly, promote mainstreaming of wetland ecosystem services and biodiversity in developmental planning and decision making. By alerting on the consequences of consumption choices and behavior, economic valuation serves as a societal feedback mechanism related to natural resources including wetlands (Zavetoski 2004).

Value is “the contribution of an action or object to user-specified goals, objectives, or conditions” (MEA 2005; after Farber et al. 2002). Valuation is “the process of expressing a value for a particular good or service . . . in terms of something that can be counted, often money, but also through methods and measures from other disciplines (sociology, ecology and so on)” (ibid.). Valuation involves assigning relative weights to the various aspects of individual and social decision problems, with the weights given being reflections of the goals and worldwide views of the community, society, and cultures of which individuals are parts (Costanza 1991; North 1994). Economic valuation of wetland ecosystem services and biodiversity is an expression of these weights in monetary terms, making them comparable with alternate uses which often have benefits and cost flows defined in similar units. It is an anthropocentric way of looking at wetlands, wherein values are assigned to the extent that these fulfill and directly or indirectly contribute to well-being (a positive change in well-being, hereinafter termed as "benefit" after TEEB 2012).

Economic Valuation Approaches and Methods

In economics, value is associated with trade-offs. Economic value exists when we are willing to give up something for the enjoyment of the value. The willingness to pay for the benefit or willingness to accept a compensation for being denied the benefit is an economic measure of this value. This economic measure is reflective of the choice pattern of all human-made, financial and natural resources given a multitude of socioeconomical conditions as preferences, distribution of income and wealth, the state of the natural environment, production technologies, and expectations of the future (Barbier et al. 2009).

Economic values of wetland ecosystem services can be derived based on biophysical or preference-based approaches. Biophysical approaches involve estimation of intrinsic properties of wetland ecosystems (e.g., material flows, primary productivity) for producing an ecosystem service. These intrinsic properties are treated as a "cost of production" of these ecosystem services. Energy analysis (Costanza 1980) and emergy analysis (Odum 1996) are some examples of application of biophysical approaches to valuation of wetlands. Preference-based approaches are based on models of human behavior and rest on the assumption that values arise from subjective preferences held by individuals (Appendix 2 of TEEB 2012 refers to a number of wetland valuation studies).

Ecosystems bear at least two major value components, one in terms of benefit arising from ecosystem service provision within a given state (termed as output value) and the second in terms of capacity of the system to maintain these values (termed as insurance values; Gren et al. 1994). The various components of output value are described by the total economic value (TEV) concept. The insurance value is related to the system's resilience and reorganizing capacity (Holling 1973; Walker et al. 2004). Ensuring resilience involves maintaining minimum amounts of ecosystem infrastructure and processing capability to remain at a given state or prevent regime shifts (ibid). Valuation of ecosystem resilience is an evolving field with challenges associated with nonlinear behavior, identifying and predicting thresholds, and the likely regime shifts.

Within the framework of neoclassical economics, techniques for value estimation derive information on individual behavior from market transactions related to ecosystem services. In cases where such markets do not exist, information from parallel markets is derived. If both direct and indirect markets are absent, hypothetical markets may be constructed for value elicitation. Accordingly, valuation techniques are classified into direct market valuation techniques, revealed preference techniques, and stated preference techniques. Revealed preference techniques are based on observation of choices in existing markets related to ecosystem services. Stated preference methods simulate a market and demand for ecosystem services by means of surveys or hypothetical changes in levels of provision of ecosystem services. Each of these techniques has its own assumptions, merits, and shortcomings.

Ecosystem services are multidimensional, contested, and context-specific, and thereby no single method is capable of generating a representative value. Each of the valuation methods implies certain models of human-nature relationship and defines whether values are revealed, discovered, constructed, or evolved during the process of valuation and in that sense are "value articulating institutions" (Jacobs 1997). While neoclassical methods assume existence of preferences which are discovered, deliberative methods are being increasingly applied to support emergence of values from a communicative social process (Zografos and Paavola 2008). A mix of methodologies is required to express the multiple values stakeholders hold for wetland ecosystem services and biodiversity.

Framework for Integrated Assessment and Valuation of Wetlands

A fuller and meaningful application of wetland valuation assessment merits careful consideration within ecological, sociopolitical, and institutional management contexts. For a wetland manager, understanding why valuation is needed, whose and what values are important, how to derive economic values, and integration in a decision-making process are critical to setting up an objective-led assessment. A framework for integrated assessment and valuation of wetland services provides process steps for such an exercise (De Groot et al. 2006). The five-steps of the framework are as follows:

Step 1 – Policy analysis to sets the stage for discussing why valuation is necessary and what kind of valuation is required.
Step 2 – Stakeholder analysis to determin the relevant ecosystem services and associated trade-offs.
Step 3 – Function analysis for assessing the capability of wetlands to deliver ecosystem services on a sustainable basis.
Step 4 – Valuation of ecosystem services. The framework recommends expression of a range of values, ecological, social, economic, and cultural using appropriate indicators.
Step 5 – Communicating wetland values to stakeholders.

Scaling Up Values

Given the differences in site characteristics, ideally a detailed value assessment for each site of interest should be commissioned. However, there are practical limitations of various sorts, key being cost and time implications. The benefit transfer method addresses the lack of information on values for a particular site by transferring an existing valuation estimate from a similar ecosystem. If care is taken to adjust for important differences between the two, benefit transfer provides a cost- and time-saving approach for estimation of economic value of ecosystem services (Smith et al. 2002).

Application of benefit transfer methods has its challenges. Transfer errors can be introduced due to errors in primary valuation studies or due to application in sites without accounting for differences in characteristics. Differences in spatial scales at which ecosystem services are supplied and demanded and nonconstant marginal values also bring in added complexity. Capacity to predict future demand for critical ecosystem services essential to human life and for which no adequate substitute exists is likely to remain limited. The problem of dealing with nonconstant marginal values over large changes in ecosystem state and functioning becomes more difficult in the face of nonlinear dynamics and calls for alternate approaches as multiple criteria or deliberative approaches (Spash and Vatn 2006).

Accounting for Uncertainty

As akin to various assessment tools, economic valuation is also associated with uncertainty of various forms and levels, which need to be understood for a meaningful application in policy. Uncertainty refers to either a situation in which the decision maker cannot enumerate the possible consequences of a decision (also termed radical uncertainty) or to a situation in which possible states of outcome can be enumerated but cannot be objectively assigned probabilities. Ecosystem services supply side uncertainty can be attributed to limited knowledge of ecosystem functioning and delivery of ecosystem services. Several empirical studies on stated preference indicate that individuals often do not act as utility maximizers, but are uncertain about their willingness to pay (e.g., Ready et al. 1995; Akter et al. 2008). Each of the valuation methods has its own conceptual, technical, and methodological shortcomings, giving rise to technical uncertainty (Kontoleon et al. 2002). One way to handle technical and preference uncertainty is to combine revealed and stated preference methods to increase reliability of valuation estimates. Preference calibration approaches also allow calibration of preference functions by using values from multiple methods. Uncertainty associated with ecosystem services supplies can be addressed with increasing understanding of complexities of ecosystem functioning, within ecological as well as socioeconomic systems.

Choosing How to Value

Valuing wetlands is a complex, spatial, and institutional cross-scale problem (Turner et al. 2003). Economic valuation is one of the several diagnostic and assessment tools and political-institutional mechanisms that facilitate understanding of wetlands as complex socioecological systems (Ostrom 2009). While it has an intrinsic appeal and utility in terms of supporting informed decision making in relation with wetlands and in particular exposing the impacts of conventional economic thinking on health and functioning of wetlands, it has several critiques as well.

Monism and utilitarianism implicit in economic valuation are seen as inducing an instrumental conceptualization of the relationship between humans and nature, based on a very limited rationale of comparing costs and benefits (McCauley 2006). The scientific objectivity associated with valuation has been questioned as economic values are negotiated based on the economic activities that surround it and thus sensitive to several factors that prevent its reduction to a single "representative" value (Sagoff 2011). The legitimacy of individual rationality and choice and preference relationships which form the basis of neoclassical economics based valuation techniques have also been extensively critiqued (Bromley and Paavola 2002; Sagoff 1994).

These limitations notwithstanding, the continued loss and degradation of wetlands and the role of economic drivers therein, urgently call for communicating the diverse values of wetlands, and the consequences of loss of vital ecosystem services, in the language of the world's dominant economic and ecological paradigms. Economic valuation is an evolving field and needs to be continually enriched with better understanding of ecosystem functioning and plurality of values, so that wetland ecosystems continue to deliver their wide ranging ecosystem services in the longer term. Shying away from valuation is not an option; rather the emphasis needs to be on a credible valuation process built on robust understanding of ecosystem dynamics and complementing societal decision-making structures.

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Economic Valuation of Wetlands: Total Economic Value

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Lucy Emerton

Contents

Abstract

Total economic value (TEV) is one of the most widely used and commonly accepted frameworks for classifying wetland economic benefits and for attempting to integrate them into decision-making. Its major innovation is that, rather than just considering commercial or extractive values, TEV also takes into account subsistence and nonmarket values, ecological functions, and nonuse benefits. TEV is used to overcome the problems associated with the under-valuation of wetland benefits that has plagued conventional economic analysis and decision-making.

Keywords

Economic value · Wetland services · Decision-making

Introduction

Total economic value (TEV) is an all-encompassing framework that is used by economists to identify and categorize environmental benefits. The concept of TEV first came into general use in the late 1980s and early 1990s (Pearce et al. 1989). It

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has now become one of the most widely used and commonly accepted systems for classifying wetland economic benefits and for attempting to integrate them into decision-making (Barbier et al. 1997).

TEV emerged largely in response to the perception that conventional economic approaches tended to see the value of the natural environment only in terms of the raw materials and physical products generated for human production and consumption, especially focusing on market activities and commercial profits. It was argued that this persistent under-valuation of environmental goods and services had in many cases led to decisions being made which resulted in economically suboptimal outcomes and, in the worst case, had incurred substantial costs and losses to the economy (Emerton 2005).

Rather than just considering commercial or extractive values, TEV also takes into account subsistence and nonmarket values, ecological functions, and nonuse benefits. Looking at the TEV of a wetland essentially involves considering its full range of characteristics as an integrated system – its resource stocks or assets, flows of environmental services, and the attributes of the ecosystem as a whole (Barbier 1994). As well as presenting a more complete picture of the economic importance of wetlands, TEV clearly demonstrates the high- and wide-ranging economic costs associated with their degradation, which extends beyond the loss of direct use values.

Total Economic Value

Total economic value distinguishes between use values and nonuse (or passive use) values. Whereas use values refer to the value of actual, planned, or possible uses of a wetland and its resources, nonuse values are the values that people ascribe to keeping the wetland in existence, even when there is no actual, planned, or possible use (OECD 2006).

The TEV categories of use and nonuse values are usually disaggregated further into four components: direct use value, indirect use value, option value, and existence value (Pearce 1993; see Fig. 1).

The TEV of a wetland would thus typically include (from Emerton and Bos 2004):

- **Direct values**: the raw materials and physical products that are obtained from wetlands, which are used directly for production, consumption, and sale – such as those providing energy, shelter, foods, agricultural production, water supply, transport, and recreational facilities
- **Indirect values**: wetland ecological functions which maintain and protect natural and human systems – such as maintenance of water quality and flow, flood control and storm protection, nutrient retention and microclimate stabilization, and the production and consumption activities they support
- **Option values**: the premium placed on maintaining a pool of wetland sites, species, and genetic resources for future possible uses, some of which may not

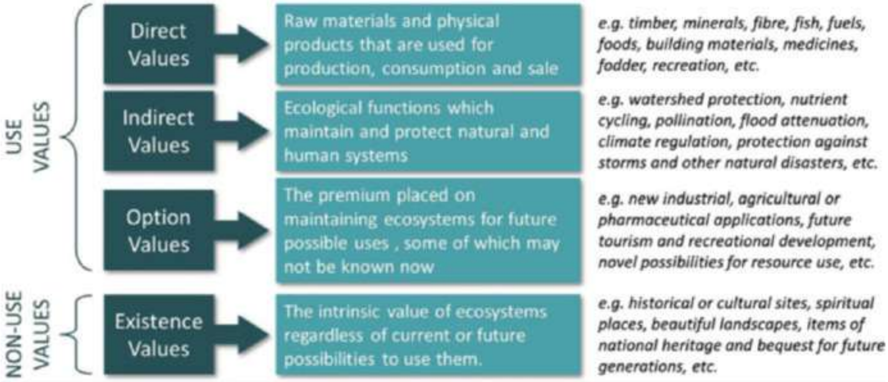

Fig. 1 The total economic value of wetland ecosystems

be known now – such as leisure, commercial, industrial, agricultural, and pharmaceutical applications and developments

- **Existence values**: the intrinsic value of wetland ecosystems and their component parts, regardless of their current or future use possibilities – such as cultural, spiritual, esthetic, and heritage significance

Some authors have proposed a further breakdown of these four, generally agreed, categories of value. For example, some interpretations of TEV separate out "bequest value": the value of preserving resources for the use of future generations (Bateman et al. 2003). This has elements of both option and existence value. "Quasi-option value" may also be included as an element of option value. This reflects the value of information gained by delaying an irreversible decision to develop a natural environment or of waiting for the resolution of uncertainty (Pascual and Muradian 2010).

The Monetary Valuation of Wetlands

One of the main applications of the TEV framework has been as a basis for estimating and communicating the monetary value of wetland goods and services. The basic aim of economic or monetary valuation is to determine people's preferences: how much they are willing to pay for wetland goods and services and how much better or worse off they would consider themselves to be as a result of changes in their supply. By expressing these preferences, valuation aims to make wetland goods and services directly comparable with other sectors of the economy when investments are appraised, activities are planned, policies are formulated, or land and resource use decisions are made (Emerton 2005).

The question of how to place a monetary value on wetlands has long posed something of a challenge to economists. The easiest and most straightforward way to value a good or service, and the method used conventionally, is to look at its market

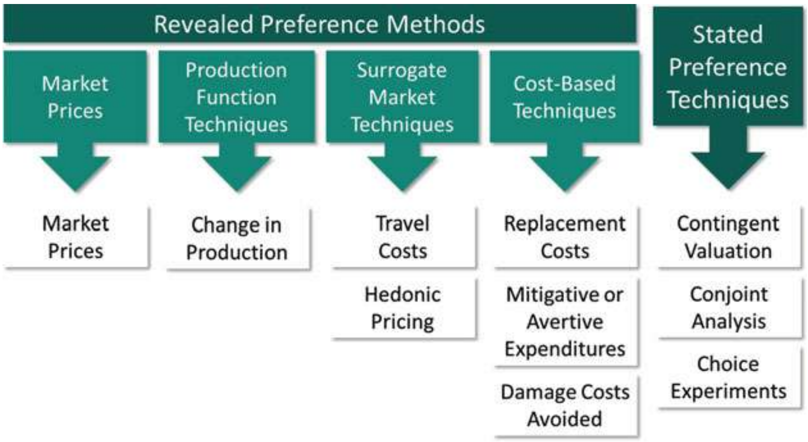

Fig. 2 Commonly used wetland valuation techniques

price: what it costs to buy or is worth to sell. Wetland goods and services, however very often have no price, are subject to prices that are highly distorted or have characteristics of public goods which mean that they are not adequately allocated or priced by the free market. Whereas it is relatively easy, for example, to estimate the revenues from a hydropower scheme or the returns to a fishing enterprise just by looking at the market prices involved, it is virtually impossible to carry out a comparable calculation for wetland water flow and water quality regulation services or for the provision of natural habitat for fish breeding and nursery. For these reasons, many wetland values cannot be calculated accurately via market prices.

Parallel to the advances made in the definition and conceptualization of TEV, techniques for quantifying environmental values and expressing them in monetary terms have also evolved over the last decades. Today a wide range of methods are available, and used, for valuing wetland benefits which move beyond the use of direct market prices (Barbier 1994; see Fig. 2). These include approaches which relate changes in the quality or quantity of ecosystem goods and services to changes in the output of a marketed good or service ("production function" techniques), look at the ways in which the value of ecosystem goods and services are reflected indirectly in people's expenditures or in the prices of other market goods and services ("surrogate market" techniques), assess the market trade-offs or costs avoided of maintaining ecosystems for their goods and services ("cost-based" techniques), or ask consumers directly how they value ecosystems ("stated preference" techniques).

Reflecting these methodological innovations, there is today a growing body of literature on the valuation of wetland ecosystem services (see, e.g., Barbier et al. 1997; Emerton and Bos 2004; Ledoux 2004; de Groot et al. 2006). There are also a large number of published studies on the value of particular wetland sites, services, and benefits across the world (see, e.g., Brouwer et al. 1999; Woodward and Wui 2001; Schuyt and Brander 2004; Emerton 2005; Brander et al. 2006; Pascual and Muradian 2010).

Future Challenges

New adaptations of economic concepts, methods, and models have enabled wetland values to be much more easily and accurately expressed and have yielded important information and insights. Yet, despite the steps forward that have been made in calculating and articulating the value of wetland goods and services, a major challenge remains – to ensure that the results of these studies, and the figures they generate, are actually fed into decision-making processes and used to influence development agendas.

However high the value of wetlands is demonstrated to be in theory, this has little meaning unless it actually translates into changes in real-world policy and practice. The better understanding and more accurate quantification of the economic value of wetlands is still reflected weakly in the policies, markets, and prices which determine the decisions faced and trade-offs reached by policy-makers, investors, producers, and consumers (Finlayson et al. 2005). While much more is now known and understood about the economic benefit of wetland services and the economic costs of their degradation and loss, these values are only just starting to be captured and internalized in the private and public decisions that are made in the real world (TEEB 2010).

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Ritesh Kumar

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Abstract

Economic valuation methods are diagnostic tools which help assess individual or collective preferences for wetland ecosystem services in monetary terms. Such values can be estimated based on information directly derived from market transaction or through transactions in related markets. In cases where markets do not exist, individuals can still express value in terms of their 'willingness to pay' for a perceived increase in benefits or 'willingness to accept' compensation for a decrease in benefits. The choice of a valuation method needs to be made based on consideration of ecosystem services and the value component that is to be elicited. While direct market valuation techniques are suited for provisioning services, regulating services have been mostly valued using production function techniques or avoided costs and cultural services through a range of stated and revealed preference methods. The potential advantages and limitations, including the range of uncertainties associated with value estimates, need to be carefully

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considered while applying the values in policy contexts. For a robust policy interpretation, economic values need to be complemented by a good understanding of biophysical complexities as well as socio-political contexts in which ecosystem services are delivered and managed.

Keywords
Economic valuation · Revealed preference · Stated preference · Benefit transfer · Uncertainty

Introduction

Economic valuation of wetland ecosystem services involves assigning money values in order to aid informed decision making for wise use of these ecosystems within the wide range of economic production and consumption decisions influencing their ecological character. Value is attributed by economic agents based on the preferences held for ecosystem services, which in turn stand influenced by socioeconomic contexts in which valuation takes place (Pearce and Pretty 1993). Economic valuation methods are diagnostic tools which help assess individual or collective preferences for wetland ecosystem services in monetary terms.

Economic values for wetland ecosystem services can be estimated based on information directly derived from market transaction or through transactions in related markets. In cases where markets do not exist, values can still be expressed in terms of "willingness to pay" for a perceived increase in benefits or "willingness to accept" compensation for a decrease in benefits. Correspondingly, valuation methodologies can be classified as being based on (a) direct market valuation, (b) revealed preference, and (c) stated preference.

Direct Market Valuation Methods

Direct market valuation methods use data from actual markets to derive economic values. These can be broadly classified into three: (a) market price-based methods which derive values based on quantity and prices traded in a perfect market, (b) cost-based methods which are based on estimation of costs incurred if the ecosystem services were to be recreated using alternate means, and (c) production function-based methods wherein values are derived from the knowledge of ecosystem services' contribution to an economic activity.

Market price method is often used to value the provisioning services of wetlands, as several wetland products (fish, timber, fiber, etc.) can be traded in markets. In perfect market conditions and in absence of distortions as taxes and subsidies, market prices provide the exchange value for these products. In several instances, market prices need to be corrected to derive shadow prices to address distortions. Based on this approach, a study on Barotse floodplains (Zambia) concluded that wetland products contributed US$ 8.64 million toward sustenance of wetland communities (Emerton 2005).

Cost-based methods are based on estimation of avoided costs (cost that would have been incurred in absence of ecosystem services), mitigation costs (the costs of mitigating the effect of ecosystem services loss), or replacement costs (replacement of ecosystem services by artificial methods). By relating mangrove width and number of deaths in the event of a supercyclone that hit Odisha state of eastern India, Das and Vincent (2009) concluded that the opportunity cost of saving a life by retaining mangroves was Rs 11.7 million (equivalent to about US$ 290,000) per life saved. Costanza et al. (2008) estimated the economic value of coastal wetlands for storm protection using a relationship between storm damage and wetland area. The study assigned an annual value of US$ 8,240 per ha for the storm protection function alone.

Production function methods are used to value ecosystem services in circumstances in which they support an economic activity (Barbier 2007). The method enables eliciting economic value based on the knowledge of the physical effect of changes in ecosystem services on an economic activity and the corresponding changes in economic output and are particularly useful in capturing regulating services of wetlands. Santhirathai and Barbier (2001) used production function modeled on mangrove-fisheries interlinkages to estimate the welfare impact associated with the loss of mangroves. With 1,200 ha annual loss of mangroves in Surat Thani province, as was the case in the early 1990s, welfare impact ranged annually between US$ 40 and US$ 132,000. A study on the Hadejia-Nguru wetlands of Northern Nigeria used groundwater recharge function of wetlands as an input to dry season agriculture to derive welfare change. As per the study, a 1 m reduction in natural groundwater recharge led to a welfare loss of US$ 82,832 over a single season alone (Acharya and Barbier 2000).

Market price-based approaches have their inherent limitations as well. A large category of wetland ecosystem services is not traded directly or indirectly in markets and hence cannot be valued using these approaches. Application of the production function approach often includes understanding of the causal relationship between wetland functioning and delivery of ecosystem services, so that inference on changes in ecosystem services resulting from changes in wetland components and processes can be derived in a meaningful manner (Daily 1997). Recent research has indicated complexity of such relationships (Koch et al. 2007).

Methods Based on Revealed Preference

As per revealed preference theory, consumers' preference can be revealed by what they purchase under different income and price circumstances, entailing that the bundle of goods and services purchased is "revealed preferred" to any other bundle of goods and services available at that particular level of income and prices. Travel cost method and hedonic pricing method are the two main methods based on revealed preference.

Travel cost method is usually applied to measure the recreational value attached to a wetland. The value is derived from the relationship between the number of visits an individual is expected to make considering the travel and time cost for the visit, which represents her willingness to pay for the recreational benefits. Using the travel cost method, the sacredness value associated with Khecheopalri Lake in the north-eastern Indian state Sikkim was assessed to be US$ 2,378 per annum (Maharana et al. 2000).

Hedonic pricing method uses information on intrinsic environmental attributes of a marketed commodity to derive the value of ecosystem services. It has been used mostly to value environmental amenities that affect residential property prices. In a study on urban wetlands in Perth, Australia, it was found that the distance to the nearest wetland and number of wetlands within 1.5 km of a property significantly influenced house sale price (Tapsuwan et al. 2009). The total sales premium for a randomly selected wetland in the area was estimated to be around AU$ 140 million (equivalent to about US$ 112 million).

Revealed preference-based methods have an intrinsic appeal in application as they are based on actual observed behavior. However, the derived values can be distorted by market imperfections and policy failures. The use of the travel cost method can be complicated in the case the trip is taken for multiple sites or when substitute sites are available. Methods used to assign opportunity cost to the time spent for recreation can also be a major source of differences in value estimates. Application of hedonic pricing is based on the assumption that housing markets are perfect and the users have the possibility to select combinations of housing attributes, including those related to environment, for a given income. Additionally, if environmental amenities are not considered to be impacting housing value, the housing prices will not include the amenity value.

Methods Based on Stated Preference

In contrast with the revealed preference-based methods which deduce willingness to pay from observed evidence of behavior when faced with real choices, stated preference-based methods rely on deriving willingness to pay from choices made in hypothetical or constructed situations. Stated preference-based methods are more suited for assessing nonuse values of wetland ecosystems and broadly include two methods: (a) contingent valuation and (b) choice modeling.

Contingent valuation method is based on use of questionnaire to estimate how much a respondent would be "willing to pay" to ensure improvement or maintenance of ecosystem services or "willing to accept" for a degradation. By means of an appropriately designed questionnaire, a hypothetical market is described wherein the ecosystem service(s) can be traded. The contingent market defines the ecosystem service, institutional context in which it would be provided, and the way the provision would be financed. The respondents are then asked to state their willingness to pay for a hypothetical improvement of ecosystem service or a willingness to accept a hypothetical decline in level of provision. In a study on stakeholder preferences for coastal wetland conservation, the total economic value of conserving

Muthurajawela Marsh and Negombo Lagoon was assessed using the contingent valuation method (Wattage and Mardle 2008). The study estimated respondents' willingness to contribute toward costs for setting up an institution to conserve and manage a good fish stock, clean water, and healthy mangroves in the wetland complex. The willingness to pay per respondent was assessed to be Sri Lankan Rs 264.26 per month for 2 years, of which Rs 118.92 was attributed to nonuse values (equivalent to about US$ 2.40 and US$ 1.00, respectively).

Choice modeling involves modeling an individual's preferences for the ecosystem services, which are described in terms of their attributes and varied levels that these attributes can take. Respondents are presented with alternative descriptions of ecosystem services and are asked to elicit their preference. By including price or cost as one of the attributes, willingness to pay can be derived from the elicitation. In a study to assess nonuse values of Cheimatida Wetlands, Greece, a choice experiment was designed to assess willingness to pay for a wetland conservation program likely to yield improved scenarios across biodiversity, open water surface area, and research and education opportunities (Birol et al. 2006). The willingness to pay per person was estimated to range between 0.12 and 15.59 euros (equivalent to about US$ 0.14 and US$ 18.70, respectively), and when aggregated across the sampled population was higher than the program cost. Do and Bennett (2009), based on choice modeling to estimate the biodiversity protection values of Tram Chim National Park in southern Vietnam, a wetland ecosystem of the Mekong Delta, estimated the net social benefit of a proposed biodiversity conservation program to range between US$ 0.52 million and US$1.84 million.

The use of hypothetical markets for estimating values has led to numerous questions on the validity of economic estimation. For example, it is not clear whether respondents' behavior in real life when faced with real choices, costs, and benefits will be the same as expressed in hypothetical situations (Kontoleon and Pascual 2007). While theoretically under perfectly competitive market conditions, willingness to pay and willingness to accept for a change related to ecosystem services need to be equal, the observed divergence between the two for similar ecosystem services has been questioned (Vatn and Bromley 1994). Studies have also indicated that respondents are unable to factor in scale and part-whole biases (discrepancies between values estimated for a whole and those derived from valuation of individual component). Commensurability of use and nonuse values in monetary terms has also been questioned as they represent different value frameworks (Martinez-Alier et al. 1998; Carson et al. 2001).

Most of the valuation techniques described above are based on individual choices for ecosystem services. Such approaches are often criticized as failing to reflect the social and cultural circumstances that influence value creation and articulation and assume that values are held in advance and can be generated through use of an appropriate valuation technique. Deliberative valuation methods which are based on participatory, deliberative, and social learning approaches have been developed as a response. The method uses reasoned discourse to explore values that guide group decisions regarding management of a resource (Howarth and Wilson 2006).

Conclusion

A range of economic valuation methods exist for valuing wetland ecosystem services. The choice of a valuation method needs to be based on consideration of ecosystem services and the value component that is to be elicited. While direct market valuation techniques are suited for provisioning services, regulating services have been mostly valued using production function techniques or avoided costs and cultural services through a range of stated and revealed preference methods. The potential advantages and limitations, including the range of uncertainties associated with value estimates, need to be carefully considered while applying the values in policy contexts. Finally, it is underlined that the economic valuation methods mostly emanate from a utilitarian framing of wetland ecosystem services and human well-being relationships. For a robust policy interpretation, economic values need to be complemented by a good understanding of biophysical complexities as well as sociopolitical contexts in which ecosystem services are delivered and managed.

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Economic Instruments to Respond to the Multiple Values of Wetlands

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Patrick ten Brink and Daniela Russi

Contents

Abstract

Wetlands are some of the most important biodiverse areas in the world, and provide key ecosystem services. A wide range of policy-based instruments need to be increasingly employed to protect them, including regulatory instruments like the designation and management of terrestrial and marine-protected areas; environmental regulation (i.e. regulation of water discharges, regulation of products and spatial planning); property rights, as well as a range of integrated management approaches (Integrated Water Resource Management, Integrated Coastal Zone Management and Maritime Spatial Planning); permit or licence decisions on land use changes, water abstraction and discharges; and restoration targets.

Keywords

Ecosystem services · Policy-based instruments · Protected areas · Spatial planning · Property rights · Integrated management · Permit or license decisions · Restoration targets

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Introduction: Wetlands Have Multiple Values to Society and the Economy

Wetlands are a fundamental part of local and global water cycles and play a key role in carbon and nutrient cycles (Russi et al. 2013). They are essential providers of ecosystem services such as clean water for drinking, water for agriculture, fishing, cooling water for the energy sector, and regulating water quantity (e.g., flood regulation). Wetlands can also play an important role in erosion control and sediment transport, contribute to nutrient retention and land formation, and enhance resilience to storms. In addition, some types of wetlands, like coral reefs and mangroves, are among the most biodiversity-rich areas in the planet and key tourism and recreational areas.

Despite the range of important ecosystem services that wetlands provide to humankind, wetlands continue to be degraded or lost (ten Brink et al. 2013). UNWWAP (2003) estimated about 50% loss of wetlands since 1900, but the true figure could be much higher (Costanza et al. 2014). For example, it was calculated that between 1980 and 2005, 20% of global mangroves were lost (FAO 2007) and that between the 1980s and early 2000s, 52% of the wetland areas of the 14 most important deltas were degraded (Coleman et al. 2008).

Policies and decisions do not adequately take into account the importance of wetlands and their contribution to human well-being and economy. The evidence base on the multiple values of water and wetlands is growing (Ghermandi et al. 2010; Barbier 2011; Brander et al. 2012; de Groot et al. 2012). Recognizing, communicating, and integrating these values into decision-making will improve governance and help meet our future social, economic, and environmental needs (Russi et al. 2013).

Achieving wise use of wetlands (as one of the pillars of implementation of the Ramsar Convention on Wetlands; Finlayson 2012) can be progressed through a range of regulatory tools and planning instruments, management and investment, and market-based instruments (MBIs) (TEEB 2010; Russi et al. 2013). Which instrument is best will be issue, country, and context-specific. This chapter focuses on market-based instruments that can be used to contribute to the conservation and improvement of wetlands.

Market-Based Instruments Can Be Used to Progress Toward the "Wise Use" of Wetlands

The behavior of governments, companies, and citizens is strongly influenced by the prices they pay for goods and services. However, the prices of goods and services often do not take account the value of the goods and services or indeed the costs of their provision. In particular, market prices usually do not take into account the environmental negative externalities, i.e., the impacts caused by one agent to another one without compensating it, like, for example, the impact on health due to a decrease in water quality. A range of market-based instruments can and have been used to protect or improve the state of wetlands by improving how prices reflect real value or costs. Prices alone will generally not address the loss of wetlands or their wise use, and they will also rarely be able to fully reflect values and costs (for legal, political, practical, and methodological issues). However, they remain an important set of instruments within policy toolkit.

Market-based instruments include taxes and charges, payments for ecosystem services (PES), subsidies and their reform, quantity-based instruments such as trading of water rights and funds (e.g., water funds), banking schemes (e.g., habitat or wetland banking), liability rules, contracts for access and benefit sharing (ABS), as well as green public procurement and associated product labels that can be important niche markets (ten Brink et al. 2011). Examples include:

- **Resource pricing**. Prices can be modified in order to take into account the contribution of key resources (e.g., water, fisheries, trees, minerals) to the economy. In general, the prices of key resources are based on the costs of extraction and trade, but they do not reflect the value of that resource for society (sometimes labeled as "shadow price"), the present and future opportunity costs of their use, and the externalities that their exploitation causes. Many countries are reforming prices to recover the costs of water provision (at least for operation and maintenance costs). Examples include the recent move toward water pricing in Ireland to help fiscal consolidation and the progress across the European Union toward fuller cost recovery, as required by the EU Water Framework Directive (Directive 2000/60/EC, article 9). However, policies aiming to increase resource pricing in order to reflect their economic values and internalize externalities are still rare, but they are very important to encourage resource efficiency and avoid wastes and misuse of important natural resources (Withana et al. 2014).
- **Reforming environmentally harmful subsidies** (**EHS**). Prices that do not reflect the cost of the provision can be considered subsidies, and moving toward cost recovery can be seen as part of reforming subsidies harmful to the environment (ten Brink et al. 2014). Examples include the lack of full cost recovery for water abstraction or to cover road infrastructure costs (Withana et al. 2014; Oosterhuis and ten Brink 2014). Similarly, EHS can exist where favorable treatment is given in the market for certain practices that lead to damage to the environment. For example, a lack of fuel excise taxes for fuel use by fishing boats and direct subsidies for new fishing vehicles can lead to overcapacity and overextraction of fishery resources (Oosterhuis and ten Brink 2014).
- **Protected area entrance pricing** can be a useful economic instrument to raise revenues for conservation and management of protected areas. One example is the entrance fee to Lake Nakuru National Park in Kenya (US$80 for international and US$11 for domestic visitors). The park has around 149,500 international visitors and 95,500 domestic visitors per year. The fee allows roughly US$13 million to be raised every year (Ramsar and UNWTO 2012). User fees for divers, introduced in 2000 in Tubbataha Reef National Marine Park in the Philippines, facilitated the acceptance of a no-take zone because they compensated local fishermen for the loss of access to the park (Kettunen and ten Brink 2013).
- **Pollution charges**, **liability, and compensation requirements** for pollution incidents or environmental damage can reduce the pressures on wetlands, because

they encourage polluting companies to put in place mitigation or risk-prevention measures to reduce the risk of incurring liability payments. These kinds of instruments help implement the polluter pays principle by financing restoration projects. One example of liability payment is the US$20 billion escrow compensation fund created by British Petroleum (BP) after the 2010 Deepwater Horizon oil spill in the Gulf of Mexico (the ceiling increased in July 2010, when BP set aside a pretax charge of US$32.2 billion to cover liabilities) (BP 2010).

- **Payments for ecosystem services** (**PES**) to remunerate land uses that deliver ecosystem services, through programs funded by government agencies, private ecosystem services users (e.g., water utilities, beverage companies, citizens), and foundations or NGOs. A famous example of PES is the Payment for Hydrological Environmental Services program in Mexico. The program was established to finance the hydrological ecosystem services provided by forests and, in particular, the protection of watersheds and aquifer recharge. The program is financed through part of the federal taxes on water and remunerates forest owners for maintaining the forest cover in areas where forests have a high impact on the water ecosystem services and are subject to high risk of deforestation (Muñoz-Piña et al. 2008). REDD+ (reducing emissions from deforestation and forest degradation) schemes and arguably the ABS scheme (see below) can be considered global PES.
- **Water funds** are also proving a useful tool to improve water management while creating employment and ecosystem benefits. They consist of long-term financial mechanisms that involve a public-private partnership of water users who determine how to invest financial resources in activities for maintaining or enhancing water services in priority areas. For example, the Quito Water Conservation Fund (FONAG) in Ecuador receives payments from water users, which are then used to invest in watershed protection projects. This helps conserve the watersheds (~500,000 ha), support reforestation (2,033 ha and 2,000,000 trees), and generate employment (Arias et al. 2010).
- **Access and benefit sharing** (**ABS**). Genetic resources support human activities in a range of sectors: botanic gardens, academic research, biocontrol, industrial biotechnology, plant breeding or seed industry, horticulture, cosmetics, pharmaceuticals, farm animal breeding, and food and beverages. However, people helping to preserve and provide genetic resources rarely get a fair share of the income these resources provide. The Nagoya Protocol on access and benefit sharing (ABS) is leading to discussions on suitable measures to support access to genetic resources. The sharing of benefits deriving from these should lead to improved financial flows to countries and communities providing genetic materials or associated traditional knowledge (ten Brink et al. 2012).
- **Offsets and habitat banking** are tools that are being used in a few countries (e.g., wetland banks in the USA, the biobanking system in Australia, and various offsetting schemes in Germany) but are still controversial. They work by allowing environmental impacts to be offset with conservation projects in a different area. The conservation projects are assigned offsets, which those responsible for the environmental impacts (in general building companies) are required to buy to

offset the biodiversity loss caused by their project. The equivalence between restoration projects and biodiversity losses can involve either the same kind of habitat or species (like-for-like) or habitats or species considered of equal or higher importance or value. Offsets arise from actions to protect habitats at risk (i.e., risk aversion offsets) or to restore degraded or destroyed habitats. Biodiversity banks create a market-based instrument by turning offsets into tradable assets (credits). Design and a good regulatory baseline are important for these instruments to offer added value and ensure that they are used only for non-avoidable residual impacts, once all regulatory instruments to protect the environment have already been used.

- **Environmental/reverse auctions for land use and water protection** have also been used in a few countries, notably Australia (e.g., in the BushTender and EcoTender in Victoria, as well as in the Murray-Darling basin for water entitlements) and in the USA (in the US Conservation Reserve Program, as well in the state of Georgia for irrigation permits, where farmers offer to forego irrigation in return for a payment). In auctions, sellers make bids (or tenders) on the cost to manage their land for biodiversity conservation or water protection, and the buyer selects the lowest-cost bid that meet their criteria. These auctions are called "reverse auctions" as it is the buyer that chooses the lowest price from multiple sellers, rather than multiple buyers competing by offering the highest price as in traditional auctions. Reverse auctions have been criticized for having important weaknesses, notably being sensitive to strategic bidding behavior, ease of political capture, and for lack of sensitivity for environmental criteria; however, they are evolving tools, being refined to help address weaknesses (i.e., by ensuring that the auction attracts a lot of bidders).

Future Challenges: How Are Market-Based Instruments Evolving and How Can Their Suitable Use Be Encouraged?

Interest in the use of market-based instruments (MBIs) has increased in recent years, partly due to global financial restrictions which triggered the search for environmental policy instruments that do not require high amounts of public funding (e.g., private PES, offsetting) or reduce government expenditures (e.g., reduction of environmentally harmful subsidies). However, MBIs are not without risks and complications. First of all, a concern has been increasingly expressed that assigning a price to environmental resources leads to commoditization of nature and makes conservation dependent on market prices (which are volatile in nature) and not on moral motivations (McCauley 2006).

MBIs should be seen as a complement and not a substitute to environmental regulation. MBIs are appropriate instruments only in specific contexts and for specific environmental challenges, and their effectiveness depends very much on instrument design. In many cases, regulation and spatial planning may be more practical than market creation and pricing (see ▶ Chap. 291, "Securing Multiple Values of Wetlands: Policy-Based Instruments" in this Volume). In addition, careful design is critical to avoid risks of impacts – for example, rare ecosystems should remain outside of habitat banking and equivalence issues (habitat, biodiversity, ecosystem service flows) looked at carefully and integrated into design.

It is also important to take into account that MBIs can be expensive. The design and implementation of a PES scheme can require considerable investment in understanding the state and functions of the ecosystems, the flow of the ecosystem services, the economic relationships, and the institutional and political context. Also, resources need to be invested to ensure additionality (i.e., remuneration of only the activities that would not be carried out without the payment) and conditionality (i.e., the link between payment and provision of ecosystem services). This all costs time and money.

Another concern is that in some places management of ecosystems is part of a social or cultural norm and establishing payment schemes may weaken moral or cultural motivation to protect nature and therefore be counterproductive (see the debate on the "crowding-out" phenomenon, e.g., in Kosoy and Corbera 2010). There is a lot to learn from the successes and failures of the application of economic instruments around the globe. The challenge is for decision-makers to learn from these, select and design new instruments, and reform existing instruments in light of the evidence of impacts on biodiversity, ecosystem services, and the stakeholders involved.

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Securing Multiple Values of Wetlands: Policy-Based Instruments

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Patrick ten Brink, Daniela Russi, and Andrew Farmer

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Abstract

Interest in the use of market-based instruments (MBIs) has increased in recent years, partly to improve economic incentives to address environmental problems, and partly due to insufficient public funding available for environmental policies. In this context, MBIs can represent a useful complement to more classical public policies as they do not require high amounts of public funding (e.g., private payments for ecosystem services (PES), offsetting) or reduce government expenditures (e.g. reform of environmentally harmful subsidies). MBIs should be seen as a complement and not a substitute to environmental regulation. They are appropriate instruments only in specific contexts and for specific environmental challenges. Their effectiveness depends very much on instrument design, institutional context and complementarity with environmental regulation. A range of MBIs can be employed to protect or improve the state of wetlands, including changes in resource prices; removal of environmentally harmful subsidies; entrance fees to protected areas; pollution charges, liability, and compensation

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requirements for pollution incidents; PES programmes; water funds; access and benefit sharing programmes to ensure that countries and communities providing genetic materials or associated traditional knowledge receive related economic benefits; offsets and habitat banking; environmental/reverse auctions for land use and water protection. Which instrument mix is best, depends on the specific problem and the country context, including regulatory and institutional environment.

Keywords

Resource pricing · Environmentally harmful subsidies · Protected area entrance pricing · Pollution charges, liability, and compensation requirements · Payments for ecosystem services (PES) · Water funds · Access and benefit sharing · Offsets and habitat banking · Environmental/reverse auctions for land use and water protection

Introduction: Wetlands Have Intrinsic and Economic Values

Wetlands are some of the most important biodiverse areas in the world. They support biodiversity in ecosystems (e.g., coral reefs, peatlands, freshwater lakes, marshes, and mangroves), species (e.g., waterbirds, amphibians, and wetland-dependant mammals such as hippopotamus, manatees, and river dolphins), and genetic diversity.

In addition to being critically important for rare and endangered species and the diversity of ecosystem, species, and genes, wetlands provide fundamentally important services to local communities, wider society, and the economy (ten Brink et al., 2012). For example, coral reefs provide habitat to a wide range of fish and invertebrate species, sustaining the livelihood of millions of people. Salt marshes provide an important contribution to water quality by removing pollutants and absorbing carbon dioxide. They also protect boat moorings and marinas and reduce the need for costly artificial sea flood defenses. Peatlands and sea grasses both store vast quantities of carbon, helping to avoid climate change. Wetlands have important esthetic, educational, and recreational ecosystem services that contribute to human well-being, cultural identity, and economy. They also hold important spiritual values for some cultures. The importance of these ecosystem services to society and the economy can be represented by a mix of qualitative, quantitative, and monetary terms depending on the issue and the audience (Russi et al. 2013).

Wetlands continue to be under pressure, very often because there is a lack of appreciation of their multiple roles and values. There is, however, an increasing recognition of the multiple values of wetlands, and a range of instruments can help protect wetlands and the services they provide (Russi et al. 2013). While economic, market-based instruments (that try to influence the prices of ecosystem goods and services to reflect their value and cost better) are addressed in another entry in this volume, this chapter focuses on policy-based instruments.

The Use of Policy-Based Instruments to Protect Wetlands and Support Ecosystem Service Provision

A wide range of policies, instruments, and tools is used for biodiversity protection, to support the wise use of wetlands and at the same time safeguard or improve the delivery of multiple ecosystem services. At the highest level, this includes global multilateral environmental agreements, protocols and targets under these agreements (e.g., the 20 Aichi targets of the Strategic Plan for Biodiversity 2011–2020, agreed at CBD-COP10 in 2010, CBD 2010), National Biodiversity Strategies and Action Plans (NBSAPs)), and other strategies and plans (UNEP-WCMC and IEEP 2013).

As regards policy instruments, these include regulatory instruments such as the designation and management of terrestrial- and marine-protected areas (e.g., Ramsar sites), wider spatial planning (e.g., for rivers and catchments), property rights, as well as a range of integrated management approaches.

Designation of protected areas (**PA**) and improving PA site management. There are currently 2,186 registered Ramsar sites in the world, covering 208,674,247 ha (Ramsar 2014). There are many more wetland sites that are under national or other designations (e.g., EU's Natura 2000 network). Designation itself and the associated wise use of the wetland, due to site management and associated investment, can improve the ecological status of the site and increase the provision of ecosystem services. It is increasingly appreciated that PAs can help not only to provide biodiversity conservation benefits but also to offer multiple local, national, and global benefits from the provision of multiple ecosystem services (Kettunen and ten Brink 2013).

Regulation and land use planning. The regulation of activities that impact water and wetlands is, where applied and enforced effectively, an essential tool to halt losses, stimulate restoration, and maintain the integrity of ecosystems and the ecosystem services they provide to people. This requires legal and institutional frameworks for regulatory action, control, inspection, and noncompliance enforcement activities. There are three main types of environmental regulatory approaches (TEEB 2011):

1. Regulation of water discharges that sets standards for emissions, ambient quality and technical practice (e.g., best available techniques), performance (e.g., water quality objectives) or management practices (e.g., agricultural activities), and water quantity regulation (e.g., limits on abstraction).
2. Regulation of products, which sets restrictions on product use (e.g., activities damaging endangered species) or production standards (e.g., certification, best practice codes).
3. Spatial planning, which regulates land uses (e.g., spatial planning frameworks such as IWRM, ICZM and MSP and designated protected areas, as noted above). Examples of regulation and spatial planning to improve water and wetland management include the control of pollution from wastewater treatment plants

to protect the quality of surface water, the designation of areas protecting drinking water sources from nitrate contamination, and the design of non-conversion zones in order to safeguard mangroves that provide important benefits.

Property rights. Institutional arrangements, such as property rights, set by law and tradition or simply defined by practice, are tools that influence the linkages between wetlands and human societies. These rights set up the rules that delimit the range of activities granted to individuals or groups over specific (or range of) ecosystem services. They include ownership rights, defining access and exclusion rights, use or withdrawal rights, as well as management rights. The complexity of property rights has an influence on the way costs and benefits of ecosystem services are distributed and shared across societies.

Wetlands and integrated water resources management. Water and wetland management have historically focused on individual management objectives, mainly aimed at maximizing provisioning ecosystem services such as agricultural production and fish catch. This has led to an impoverishment in the capability of such ecosystems to deliver regulating, supporting, and cultural ecosystem services. It is being increasingly recognized, however, that wetlands should be managed to meet a multiple interacting objectives (Russi et al. 2013). Such "multi-objective" management results in provision of a wider range of ecosystem services, including fishery preservation, improved water quality, flood control, carbon sequestration, and recreation, in parallel with improved biodiversity.

In order to facilitate this task, Integrated Water Resource Management (IWRM), Integrated Coastal Zone Management (ICZM), and Maritime Spatial Planning (MSP) have been developed in recent years as innovative approaches to water and coastal management. They are focused on the landscape scale (e.g., river basin, coastal zone, marine region), are multidisciplinary in nature, and pursue the involvement of different types of stakeholders (GWP and NBO 2009). Other spatial planning approaches, such as urban planning, can also provide a key role in "multi-objective" management.

These approaches allow decision makers to simultaneously discuss and formulate multiple objectives (e.g., ensuring water, food, and energy security, mitigating and adapting to climate change, alleviating poverty) and to identify synergies not only between the services and objectives but also between the potential management options. In addition, these approaches facilitate the process of dealing with the tradeoffs between policies aimed at improving different ecosystem services (e.g., provisioning ecosystem services versus regulating/supporting ecosystem services). Finally, integrated approaches are also important to mainstream protection/restoration solutions into water, food, energy, climate, and development policies.

Permit or license decisions on land use changes (e.g., draining a wetland), water abstraction, and discharges should take into account the multiple benefits of wetlands and thereby avoid permit decisions that would lead to an unacceptable level of wetland degradation. Improved integration in the permit decision of likely changes in ecosystem service flows to different beneficiaries (were a permit granted) can help clarify how public and private benefits might change and hence help clarify where the permit is (or is not) in the public interest. This can help encourage the wise use of wetlands, support their conservation and improvement, and secure multiple benefits from ecosystem service flows.

Restoration targets are being set at global level (Aichi targets 14, 15; see CBD 2010) and at the European level (the EU's biodiversity strategy sets a target of restoring 15% of degraded ecosystems by 2020; see CEC 2011). There is restoration potential not only for degraded protected areas but also for non-protected ones. Restoration of protected areas tends to focus on increasing ecological quality of the site and achieve conservation benefits. This often leads to co-benefits in the form of increased flows of ecosystem services. Restoration of non-protected areas on the other hand can focus on meeting a range of policy objectives through the provision of ecosystem services. Examples include water security through clean water availability, personal security through natural hazards mitigation, climate mitigation and adaptation, health and recreation for citizens, and food or materials provision, attracting investment through improved locational quality, regional development, and territorial cohesion (IEEP and Milieu 2013). In these cases, there will often be biodiversity co-benefits. For example, in South Africa the Working for Water and Working for Wetlands programs focus on restoration with a view of addressing water objectives and, at the same time, development objectives (Pollard et al. 2008). While restoration in itself can lead to multiple benefits, preventing degradation in the first place is generally a still more cost-effective option.

Two important information-based tools that complement the above core policy instruments are the **strategic environmental assessment** (**SEA**) and the **environmental impact assessment** (**EIA**). Integration of evidence on wetland ecosystem services in SEAs of policies and programs and project-level EIAs can help avoid wetlands degradation or conversion and thereby help avoid loss of ecosystem services.

A further information-based tool that supports policy is that of **environmental-economic accounts**. Where sufficient data is available, they could help provide evidence on the state of wetlands, the ecosystem services they provide, and on the relationship between land use, carbon storage, and water. Under existing commitments (Aichi target 2), accounts are to be developed by 2020 and have the potential to create a long-term evidence base to support future policy actions that could contribute to securing the multiple benefits of wetlands (CBD 2010).

Future Challenges: How Are Instruments Evolving?

The designation of terrestrial- and marine-protected areas has been increasing rapidly over the past two decades. However, many protected areas are still far from reaching the necessary conservation or ecological status due to lack of sufficient management which, in turn, reflects a lack of sufficient funding. The need to obtain funding needed for due management remains a key challenge. Clear identification and communication as to the importance and benefits of wetland conservation is being increasingly used as a tool to help leverage funding and hence realize the multiple benefits from protected areas (Kettunen and ten Brink 2013).

The use of SEA and EIA requires their timely development, due content, and integration in decision-making processes, whether policy and planning in the former or projects in the latter. This requires skills (e.g., ecologists and environmental economists in planning departments), timing, and commitment (e.g., to actually make use of the results in the decisions).

For restoration, the challenge is to identify the areas where restoration offers the greatest added value, both for nature conservation and ecosystem service provision. This requires ecological and economic expertise, as well as mapping skills. Integrating the concepts into existing processes is also a challenge, e.g., using ecosystem-based mitigation and adaptation to climate change.

There is an increased uptake of ecosystem-based management, i.e., management that takes into account the whole ecosystem (including people and the environment), their functions, and interactions. This systems approach contrasts with management approaches that focus on a single issue or objective in isolation. While there is progress, more needs to be done for this ecosystem-based management to become the norm.

Finally, for a number of challenges, transition management measures will be critically important to get support from, and/or reduce opposition by, stakeholders. This can take the form of spatial planning decisions – e.g., designating certain areas for protection and others for service provision – or financial mechanisms such as compensation of those whose benefits are eroded as a consequence of wetland protection.

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Economic Valuation of Wetlands: Case Studies

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Mishka Stuip and Anne A. van Dam

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Abstract

Economic valuation is recognized as an important tool to incorporate the value of nature into decision making on wetlands, but its application in the policy process is still a challenge. Valuation can be applied at different scale levels (from local to global) and a range of different methods is available. In this paper, four examples of valuation studies at different scales are reviewed: a global study estimating the value of the world's wetlands using the benefit transfer method; a study of the Kala Oya River Basin in Sri Lanka using a cost-benefit analysis of four management scenarios; a study of Randers Fjord in Denmark using a contingent valuation study to estimate the value of water quality improvement; a study of Nakivubo wetland in Uganda using market value and the replacement cost

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method; and a study of shrimp farming in Thailand which uses the production function and expected damage function approaches. These cases demonstrate that estimation of indirect use and of non-use values is more challenging than estimating direct use values, both in terms of the availabe valuation methods and in terms of uncertainty around the outcomes. However, the indirect use values are usually much higher than the direct use values, which emphasizes the need for governments to incorporate estimates of these values in their decision making processes to avoid the loss of the ecosystem services represented by these high values.

Keywords
Economic valuation of wetlands · Decision-making · Ecosystem services · Trade-off analysis

Introduction

Economic valuation is now recognized as an important tool to incorporate the value of nature into decision making and to prevent the further loss of biodiversity and natural resources in the world (Costanza et al. 1997, 2014; de Groot et al. 2006; Russi et al. 2013). However, valuation of ecosystem services has limitations, and the application of economic valuation in the policy process at different governance levels (river basins, countries, wetland sites) is still a challenge. Values by definition are instrumental, anthropocentric, individual-based, subjective, context dependent, marginal, and state dependent (TEEB 2010). There is an ongoing debate about the concept of valuation and the utilitarian approach to valuation of ecosystem services as opposed to the ethics around the intrinsic value of ecosystems (Tallis and Lubchenco 2014). Also, governments and nongovernment organizations involved in decision making about wetlands face challenges when applying valuation methods for different ecosystem services. Because of gaps in the literature on the values of wetlands, the existing data should be seen as indicative, and further research and capacity building are needed to support the application of economic valuation in wetland policy and management (TEEB 2010).

The selection of the valuation method depends on a range of factors. Each wetland site has its own biophysical characteristics and its own group of stakeholders, making each valuation case unique. Valuation methods differ in the degree of necessary resources, require different levels of stakeholder involvement, and provide different kinds of information. Methods thus have advantages and disadvantages, and adopting a valuation method implicitly means adopting a certain model of how humans and nature interact (TEEB 2010). To give an idea of the application of different methods, in this overview we present some examples of economic valuation case studies at different scale levels (global, at wetland site level) and with different objectives (e.g., for management decision making).

Economic Valuation of the World's Wetlands

A number of valuation studies attempted to estimate the value of the world's ecosystems on a global scale. These studies use the "value transfer" approach (Schuyt and Brander 2004; Brander et al. 2006; Costanza et al. 2014), in which statistical relationships (using regression analysis) between wetland characteristics (such as wetland type, income per capita, population density, and wetland size) and the economic value of the wetland are estimated for known wetlands. The data for these relationships are obtained from individual wetland valuation studies. Then, these relationships are applied to a much larger number of wetlands (for which the characteristics are known), assuming that the same relationship with economic value exists. This form of value transfer is called "function transfer" and uses the characteristics of the wetlands to obtain a more accurate estimate of value. "Direct value transfer" simply applies the value per hectare of a study wetland to another wetland, without considering characteristics such as wetland type, population, etc. A study to estimate the value of the world's wetlands used a regression analysis of 89 wetlands with the resulting value function being applied to 3,800 wetlands around the world (with a total surface area of 63 million hectares) to obtain a global economic value of 3.4 billion US$/year. The highest benefits were found in Asia with an economic value of 1.8 billion US$/year (Brander and Schuyt 2010).

A first estimate of the value of the world's ecosystems was published by Costanza et al. (1997). They estimated the total value of the world's ecosystems at 45,900 million US$/year (converted to 2007 dollars), of which wetlands, coastal ecosystems, and lakes and rivers (all within the Ramsar definition of wetlands) represented 26,300 billion US$/year. In a recent paper, they updated their earlier estimates and arrived at 56,600 million US$/year for the same wetland categories (Costanza et al. 2014). This increase was mostly caused by better recent estimates of the value of ecosystem functions compared to 15–20 years ago. Applying the current value estimates to both the 1997 and 2011 ecosystem areas, the loss of ecosystem services from all biomes in the world due to land use change was estimated between 4.3 and 20.2 trillion US$ per year.

A common critique in connection with benefit transfer at this large scale is that the wetlands to which the regression equations are applied are often different in terms of size, the functions and services provided, and the socioeconomic characteristics. Transferring estimated values from one study to another and aggregating them at larger scales creates inaccuracies. As an illustration, the per ha value from the two studies cited above were US$ 54 and 7,536 (assuming 1997 dollars), showing the large differences that can occur when aggregating numbers from widely varying wetland types. Some of this can be avoided by using more sophisticated value transfer methods that take into account wetland characteristics and expert opinion. The most recent models use spatially explicit information and even include dynamic function modeling to increase the precision of the estimates. The main objective of these large-scale estimates of economic value is a reframing of nature as natural capital, one of the important assets contributing to human well-being (next to built capital, human capital, and social capital) and to raise awareness of the importance of incorporating the value of nature in decision making (Costanza et al. 2014).

Wetland Site Valuation Case Studies

Kala Oya River Basin, Sri Lanka

The Kala Oya River Basin in Sri Lanka covers an area of about 2,870 km^2 and expands over three provinces and four districts. The basin includes different types of ecosystems and different forms of land use. The river basin is characterized by serious ecosystem degradation and has known several conflicts between resource users (Vidanage et al. 2005; Emerton et al. 2005). Part of this system consists of human-made wetlands for water storage that are locally known as "water tanks." These tanks store water for rice production but also provide fish, lotus flowers, and roots that contribute to household incomes. The tanks also contribute to maintaining water quality in wells and are a source of drinking water and fodder for livestock. Unsustainable land use and an increased demand for water in the catchment have led to a reduced inflow of water into these water tanks and an increase in sedimentation. Until now, tank management consisted of mechanically raising the spillway of the sedimented tanks in order to rapidly restore their capacity for water storage. Although in the short term this was the least expensive and required the least effort, in the long term this method is not the most cost effective. Raising spillways does not reduce tank sedimentation and wetland degradation – it merely postpones it and does nothing to address its cause. A valuation study was done to come up with a better, cost-effective management strategy for the future.

The valuation study consisted of two parts. Direct use values of the wetland (provisioning ecosystem services: resources that could be bought or sold such as water, crops, and fish) were estimated according to their market prices based on a participatory rural appraisal (PRA) and focus group discussions (FGD) to understand what the tanks were used for. Indirect use values (regulating ecosystem services: ground water and subsurface water recharge, sediment and nutrient retention, and biodiversity) were valued qualitatively using a plus/minus system that was applied to four future management scenarios. The natural capital of the system (the ability of the tanks to provide ecosystem services in the long term) was also assessed qualitatively. Results showed that the tanks yielded an average value of US$ 425 per household per year in terms of water and aquatic resource use or almost US$ 3,000 per hectare of inundated area. A cost-benefit analysis of the four scenarios showed that net benefit of the most expensive scenario was the highest. This scenario also scored highest on the ecosystem services and natural capital indicators (see Table 1).

Scenario 4 was selected because in the long run this scenario would prove the most beneficial to the community. The findings of the Kala Oya study underline the importance of looking at livelihoods and environmental values when land use and water allocation decisions are made. Before the valuation took place, there was a

Table 1 Comparison of four future management scenarios for wetlands (water tanks) in the Kala Oya River Basin, Sri Lanka. Net present value (NPV) was estimated using participatory methods. Indirect use values were estimated using qualitative methods (Adapted by permission from Emerton 2005, © 2005 IUCN)

Future management scenario	Direct use values (NPVin US$ per tank)	Indirect use values	Natural capital
1. Do nothing. Here, sedimentation loads remain the same or increase, and tank wetlands continue to deteriorate	0	−7	↓↓
2. Raise the spill. Here, the water body will grow, and additional land will be flooded, but sedimentation loads remain the same or increase	23,800	−4	↓
3. Raise the spillway and rehabilitate the area around the tank. Here, the water body will grow, additional land will be flooded, and future sedimentation loads will be reduced, thus prolonging the lifespan of the wetlands	28,800	6	↑
4. Remove the silt and rehabilitate the tank reservation. Here, original tank capacity and seasonality is restored, and future sedimentation loads will be reduced, thus prolonging the lifespan of the wetlands and restoring their ecosystem goods and services	57,900	7	↑↑

lack of ownership feeling by the community, who placed the responsibility for upkeep of the tanks with the (local) authorities. The valuation made the community realize that their tanks not only support their agricultural activities but also fishing, collecting lotus flowers, weaving mats, water for livestock, etc. There is now a general acknowledgement of the value of the tanks. Further improvement would be a mechanism to ensure community participation in regular maintenance of the tanks as well as more thorough maintenance and repair by the (local) authorities.

An Initial Economic Evaluation of Water Quality Improvements in the Randers Fjord, Denmark

Randers Fjord is a 27 km long shallow estuary in the County of Arhus on the east of Jutland, Denmark. The Fjord is located in the Gudenå catchment which has an area of 3,260 km^2. Land use in the catchment is dominated by agriculture (64% of land use), but there are also industrial activities and urban areas with associated wastewater discharge. Randers Fjord is used for tourism and recreational activities like sports fishing, water sports, swimming, and camping. Wastewater discharges and agricultural runoff have led to a tenfold increase in nitrogen and phosphorus loading compared with natural levels in the Randers Fjord. Increased microalgal growth and sedimentation of organic matter caused decreased light penetration and a marked decrease of submerged aquatic vegetation (eelgrass beds).

An economic analysis of the implementation of European water quality legislation (the EU Water Framework Directive) was done to weigh the costs and benefits of reduced eutrophication of Randers Fjord (Atkins and Burdon 2006). A contingent valuation study collected data on the public's preferences for water quality improvements by reducing the level of eutrophication. The data were collected by a postal survey, in which a hypothetical 10-year action plan was presented which would improve water quality and clarity. It was explained that to implement the plan, an increase in local taxes would be required, and respondents were asked if they were willing to pay a tax for this purpose and if so, how much they would be willing to contribute per month. After a pilot survey, the main questionnaire was sent to 1,510 individuals and 15% of them responded. Respondents were diverse in terms of personal characteristics (63% males, 68% in the 30–60 year age group, and from a wide range of income groups). Their uses of the fjord were mostly recreational (recreational hunting/fishing 11%; walking, jogging, and running 12%; bird/wildlife watching 9%; enjoying the views and the sounds of the water 22%). Only 43% of the respondents knew about eutrophication before the survey, and 36% had seen the effects of eutrophication in the fjord themselves. Results indicated that 70% of the respondents "definitely or probably" supported the proposed action plan that would return the Fjord to its pre-1915/1916 (pristine) conditions. Average maximum willingness-to-pay was €12.02 per month over the suggested 10-year period of the action plan. At the Arhus County level, this implies potential total benefits in the region of €5.5 million per month over the 10-year period. These survey findings indicate that the residents of Arhus County value reduced eutrophication in the Randers Fjord. Further statistical analysis of the survey evidence will provide greater insight into the preferences for such water quality improvements.

Nakivubo Swamp, Kampala, Uganda

Almost one sixth of the capital city of Uganda, Kampala, is covered by wetlands. Of these wetlands, the Nakivubo swamp is the largest one, covering an area of over 5 km^2 with a total catchment area of 40 km^2. It is dominated by papyrus (*Cyperus papyrus*) vegetation but also has cattails (*Typha* sp.), common reeds (*Phragmites* sp.), and *Miscanthidium* grass. Due to its location, Nakivubo swamp functions as a buffer through which most of the city's wastewater passes before being discharged into Lake Victoria. Over the last decade, Uganda has experienced rapid economic as well as population growth. The urban population, which now comprises over 14% of the total population, is increasing with 5% per year. This creates a pressure to convert Kampala's wetlands for residential and industrial development, with the argument that the "undeveloped" wetland is not worth much compared to industrial and residential areas. A valuation study was therefore done to estimate the economic value of the wetlands' wastewater purification and nutrient retention functions (Emerton et al. 1999).

The direct use values (wetland resources or provisioning ecosystem services) of Nakivubo swamp consisted mainly of agriculture, papyrus production, brick

Table 2 Economic valuation of Nakivubo wetland, Uganda. All values are in thousands of US Dollars converted from Uganda Shillings (using 1998 exchange rate) (Adapted by permission from Emerton et al. 1999)

Economic benefits	Current value ('000 USD/year)	Value per unit area ('000 USD/ha/year)
Wetland resources		
Crop cultivation[a]	88.8	0.90
Papyrus harvesting	14.0	0.92–3.46
Brick making	25.8	2.02
Fish farming	4.9	0.26–1.51
Total direct values	133.6	–
Wetland services		
Water treatment and purification[b]	1006–1866	2.44–4.06
Total indirect values	1006–1866	
Total wetland value	1140–2000	

[a]Excluding dryland cultivation value (71.9)
[b]Excluding reticulation (upkeep water network-small channels) costs (282.3)

making, and fish production of which agriculture provides over two-thirds of the economic value. By comparing the production value of irrigated crops (through the wetland) to rainfed crops as well as the additional savings on the purchase of fertilizer as would be necessary for areas outside of the wetland basin, the direct use value of the wetland can be calculated. For the indirect use values (or regulating ecosystem services) of the wetland, two valuation methods were used: valuing the avoided cost of replacing natural wetland functions with human-made alternatives; and valuing of foregone expenditures on mitigating or offsetting the effects of wetland loss. These methods estimate the costs of having to supply equivalent wastewater purification and nutrient retention services by other means and of moving the water intake for drinking water production, both in case Nakivubo wetland would no longer be there. The replacement costs included connecting Nakivubo channel to a sewage treatment plant which could absorb the wastewater load as well as the construction of elevated pit latrines to prevent sewage from low-cost settlements to enter the wetland. To compensate the effects of reduced water quality due to wetland loss would require the inflow from Kampala's water supply to be moved to a different location away from where the wastewater flows into Inner Murchison Bay. The National Water and Sewerage Corporation (NWSC) had already calculated the cost for these works for earlier projects which were never implemented. The cost of building pit latrines was obtained from a part of Kampala with similar circumstances.

Results showed that the wastewater purification and nutrient retention services of Nakivubo swamp had an economic value of between US$ 1 million per year (using the replacement cost method) and $1.75 million per year (using the mitigative expenditures methods) (Table 2). Even taking account of the costs of managing the wetland to simultaneously optimize its waste treatment potential and maintain its ecological integrity (some US$ 235,000) resulted in a significant net benefit. The high economic value of Nakivubo wetland for nutrient retention and wastewater purification provides a strong argument for protecting the wetland from further draining and development. In fact, the wetland saves the NWSC a large amount of money each year as it is less expensive to maintain the wetland than to build and maintain human-made infrastructural works.

Although Nakivubo wetland already greatly improves the water quality of water entering Inner Murchison Bay, its wastewater treatment capacity is not even fully utilized. Currently the two outlets which carry most of the wastewater and between 75% and 85% of the nutrients only pass through a limited area of the wetland and the wastewater is only retained for 0.5–2 days at most (COWI/VKI 1998). To optimize wastewater treatment as well as ecological integrity of the wetland, channels would need to be dug and maintained for optimal dispersion of the wastewater across the whole wetland area. This option would be low cost and easier to maintain than a more costly option of constructing artificial wastewater treatment facilities which require higher level of maintenance and operation.

Shrimp Farming in Mangroves in the Coastal Areas of Southern Thailand

Between 1961 and 1996, Thailand lost over 2,050 km^2 of mangrove forests, equivalent to about 56% of the total pre-1961 coverage. Over half of this loss was due to conversion mainly for shrimp farming and other coastal developments. Mangrove loss has called attention to two main ecosystem services provided by mangrove ecosystems: their function as a nursery and breeding habitat for fisheries and their role as natural “storm barriers” for periodical coastal storms, tsunamis, and typhoons. Additionally, many coastal communities rely directly on the mangroves for a variety of products such as fuel wood, timber, raw materials, honey, resins, crabs, and shellfish.

The production function approach and the expected damage function approach were used to value nursery and breeding habitat for fisheries and the storm protection services of the mangroves. These values were then compared to the value provided by shrimp farming, both before and after subsidies are taken into account. The benefits of mangroves in Southern Thailand were estimated at about 10,821 US$/ha for coastal protection against storms, 987 US$/ha for fish nurseries, and 584 US$/ha for collected wood and non-timber forest products (all in net present value terms; Barbier 2007; Fig. 1).

In contrast, the benefits of commercial shrimp farming were estimated at 9,632 US$/ha with government subsidies contributing the equivalent of 8,412 US$/ha (Fig. 1). Hence shrimp production without subsidies created benefits of only 1,120 US$/ha which was dwarfed by the monetary value of the ecosystem services provided by intact mangroves (Hanley and Barbier 2009). While the benefits of mangroves are provided continuously, shrimp production declines after 5 years, and shrimp farms are abandoned when turning unproductive. The costs of restoring

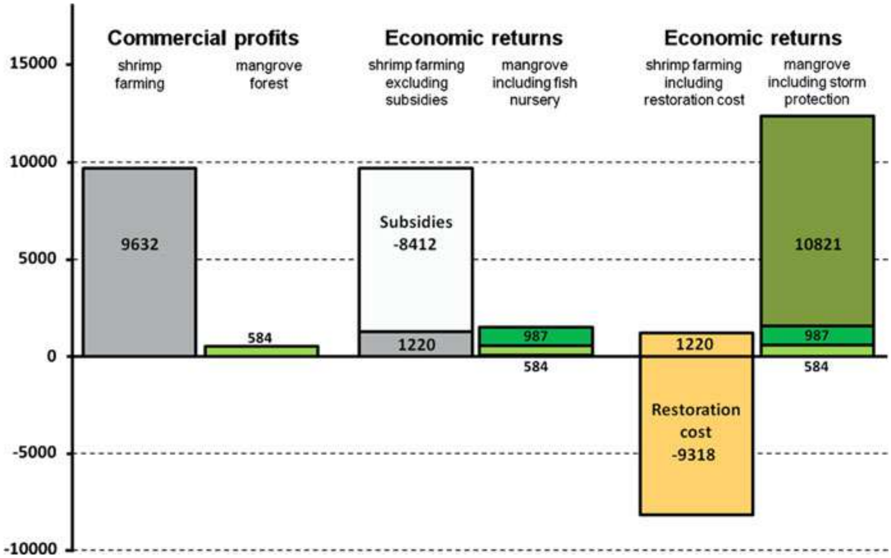

Fig. 1 Benefits of intact mangroves and shrimp farms in southern Thailand with and without subsidies. All values are net present value in US$/ha/year (Figure adapted from Russi et al. (2013) and based on Barbier (2007) and Hanley and Barbier (2009); used with permission)

mangroves are 9,318 US$/ha beyond the private profits from shrimp and have to be borne by the public.

Wise use of wetlands, including the conservation and restoration of the wetlands' hydrological functions, is essential in maintaining services such as storm protection and nursery and breeding. In many cases, natural ecosystems can provide ecosystem services at a lower price than hard engineering approaches. According to these value estimates, the main economic benefit with the highest value provided by the mangrove is protection against storms (Russi et al. 2013).

Conclusion and Future Challenges

Based on the cases studies presented, it becomes clear that estimation of the indirect use values and nonuse values (equivalent roughly to the regulating, cultural, and habitat ecosystem services of the Millennium Ecosystem Assessment and the TEEB studies) is much more challenging than estimation of the direct use values (provisioning services). Goods produced by wetlands (often food or other materials) are traded on markets using market prices that can serve as a basis for monetary valuation. For the regulating and cultural ecosystem services, more indirect methods for estimation are used, and the range of uncertainty around these values is much higher (see, e.g., Table 2) or the estimation can only be done in a qualitative way (Table 1). Despite these challenges of quantitative estimation, inclusion of regulating and cultural services in a valuation exercise can still be very useful as valuation of different options allows preference ranking, trade-off analysis, and decision making.

Another observation is that in most valuation studies, the value of the regulating services is much higher than the value of the provisioning services. For example, Table 2 shows that the water treatment functions of Nakivubo wetland were valued an order magnitude higher than the food production. Figure 1 demonstrates the same phenomenon when comparing the values of shrimp production with the value of the storm protection function f mangroves. Similar results can be observed in many other valuation studies (see, e.g., Stuip et al. 2002). This implies that protection and sustainable management of wetland ecosystems and their functions is economically much more viable for governments than conversion of wetlands for food production (and probably also for other commercial uses). Very often, decision makers are not aware of the enormous economic value they lose when permission for wetland conversion is granted. These losses are generally not borne by the proponents of the “development”, but by the general public who suffer the consequences of water quality degradation or the loss of their livelihoods options. In many cases, it may be possible to have “the best of both worlds” by using wetlands responsibly and wisely, developing food production and other activities in such a way that the ecosystem functions are preserved. More research into such “wise use technologies” is needed.

Future challenges also include further research into valuation methods, especially for the indirect use and nonuse values. Hydrological and ecological modeling will increasingly help in estimating quantitatively the water, nutrient, soil, and plant processes that are the basis for many wetland functions related to water storage, flood protection, and water quality regulation. These material processes can then be valued using avoided cost or replacement cost methods. Contingent valuation methods will also be developed further. While monetary valuation is not strictly needed for determining trade-offs between different wetland benefits, it will be particularly useful for applying certain market-based policy approaches such as payment for environmental services (PES) or mitigation banking. Most of all, wetland valuation will help to increase awareness of the importance of wetlands as natural capital that is needed for human development and needs to be protected. Global-scale studies will mainly help to achieve that awareness, where the valuation studies on more local scales (river basin or wetland site scale) can play an important role in actual decision making about use and management.

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