Agnieszka Ewa Latawiec, Dorice Agol
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Foreword

Indicators of sustainability are in vogue again, after a lull of a few years when a breakthrough in the drive for new and broader measures of progress inspired by Agenda 21 and the first Rio Conference appeared to elude science and the policy communities. This re-energized interest is driven in no small part by international politics that placed sustainable development goals (SDGs) at the center of the post-2015 development agenda. Taking SDGs and targets seriously requires tracking progress, and tracking progress requires sustainable development indicators (SDIs).

Even though the post-2000 years haven't resulted in a spectacular breakthrough in how society measures progress, the idea that progress is multi-dimensional and that it requires a more holistic approach to measurement quietly and steadily took hold. From corporate sustainability reports to community wellbeing projects, the reporting on international agreements, or thematic ecosystem assessment, not to forget the cottage industry around developing an ever-growing assembly of integrated indices, the sum total of these initiatives resulted in an increasingly rich and diverse 'indicator zoo' (Pinter et al., 2005).

The cases presented in this volume stand as illustration that the practice of developing alternative indicator systems has spread to countries as far apart as Poland and Brazil and to sectors as diverse as cattle ranching, wastewater treatment and *pronghorn* conservation. Complementing well-established measures of progress apparently does not only make sense under a wide range of contexts, but despite perennial data challenges it is also feasible. While undeniable data problems remain, as pointed out by the UN Secretary General's Independent Expert Advisory Group on the Data Revolution, among others, there is significant progress in the technologies of data acquisition, monitoring and sharing and in the 'outsourcing' of measurement to an ever growing community of social stakeholders (UN, 2014). This bodes well for the more recent interest in SDG indicators, underlining the feasibility of building such indicator systems on existing foundations (Pinter et al., 2014).

Beyond the spread of measurement practices, however, what has also evolved is our understanding of how measurement can make a difference. In contrast with an earlier common 'if we build it they will come' mentality, there is a more thorough understanding of what makes indicator systems useful in governance and accountability mechanisms. This calls more attention to the needs, interests and capacities of actors who are the target audience of indicators and whose decisions and actions turn sustainability from theory into practice and results. More emphasis on the role of indicators in governance, decision-making and strategic management can not only help improve the effectiveness of policy implementation, but it also makes clear the function of indicators. These functions may vary according to actors and their needs and essentially cover all stages of the strategic management cycle from monitoring and reporting to planning or exercising control (Lehtonen, 2015).

Bringing together cases of indicator use from a range of thematic, sector and geographic contexts in this volume shows that indicator use is ‘endemic’ to policy implementation, though also underlines that it is not without challenges and effective use cannot be taken for granted. The cases discussed here will be of interest to practitioners looking for analogues of indicator use to their own context. However, at a higher level, the more general lessons will also be of use to the broader community interested in making SDG planning, implementation and reporting more evidence based and accountable. With the statistical community paying increasing attention to developing the conceptual framework and data collection capacity for indicators that will accompany the SDGs, this volume makes an essential contribution by reminding readers that the endpoint of sustainable development indicators is not simply measuring progress, but navigating implementation to the point of sustainable and verifiable outcomes.

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Agnieszka Latawiec and Dorice Agol, November 2015.

Cover photo – Easter Island

History provides many examples of how unsustainable patterns of natural resource use have had drastic consequences. One classic example is Easter Island where, after decades of widespread deforestation, an island that was once a rich tropical forest was turned into the almost desert-like environment that we see today.

Introduction - Why Sustainability Indicators In Practice?

Agnieszka E Latawiec and Dorice Agol

1 From Rio To Rio – A Short History Of Sustainability Indicators At The International Level

1.1 The First Earth Summit And Aftermath

Sustainability indicators attempt to capture measures of economic, social and environmental processes in order to assist decision making to improve social and environmental outcomes. In other words, they are a means of gauging sustainability. Criteria that sustainability indicators should satisfy abound in the literature including, among others, the need to accurately reflect changes in the system, be transparent, measurable, verifiable, socially acceptable, adaptive, and easily communicated (see Meadows, 1998; Hak et al., 2007; Agol et al., 2014). Indeed, sustainability indicators are as complex and varied as the definition of sustainability itself and the source of this variety is discussed further in chapter 1.

Although sustainability indicators in one guise or another have been employed throughout human history, such as the use of soil colour by farmers as a simple indicator of soil fertility, they first started to become widely recognized following the first United Nations Earth Summit in Rio de Janeiro in 1992. Agenda 21, adopted at the Rio 1992 conference, for the first time explicitly emphasized the need for sustainability indicators for monitoring and fostering sustainable development via the action points of the Agenda (UNCED, 1992). The significance of the first Rio Summit was that the environmental concerns were suddenly no longer the obsessions of green pressure groups but part of global development solutions for humanity that were being sought by governments across the world.

During the United Nations Commission on Sustainable Development in 1994 in New York, concrete ideas for sustainability indicators were proposed, yet the political will to adopt them was lacking (Hak et al., 2007). As a consequence, the Scientific Committee on Problems of the Environment (SCOPE) and the United Nations Environment Programme (UNEP) were commissioned to undertake a joint project to foster the design and application of science-based sustainability indicators. The synthesis of sustainability indicators resulting from this project, SCOPE 58, was distributed to all delegations at the United Nations General Assembly Special Session in 1997. During the subsequent meetings of the UN Commission on Sustainable Development and other international forums it was highlighted and re-affirmed that indicators are widely accepted and recognized as critical tools for sustainable development. Ten years after Rio, in 2002 at the World Summit on Sustainable

Development in Johannesburg, voluntary use of sustainability indicators at the national level was encouraged. Both summits, which were held five and ten years after the first Rio Earth Summit, were considered disappointing by many observers (Dresner, 2008). Targets and timetables related to key sustainability issues such as overpopulation, overfishing and biodiversity loss were weakened, while targets to increase renewable energy were blocked by countries and industries closely linked to the fossil fuel-based energy sector. Moreover, no new commitments were made to increase international aid and relieve debt to tackle the commodity crisis.

In May 2004, a workshop was held in Prague in the Czech Republic, gathering experts from a range of countries in order to review and formulate the specific features of sustainability indicators in an attempt to resolve discrepancies and inconsistences in their use (Hak et al., 2007). Indeed, many indicators have been developed at the national and state levels, many organizations and communities have used sustainability indicators to measure their progress. There have been various recommendations on use of sustainability indicators following a number of different meetings and workshops (Bell and Morse, 2008). Bossel (2001) proposed a system-based framework where sustainability was perceived through a lens of a system and not an isolated set of unconnected attributes. Bossel's distinct approach also leaves plenty of scope for selecting a particular sustainability indicator, whilst providing context regarding its relationship to other aspects of the system. Furthermore, the approach recognizes the need to involve multiple actors in the development of sustainability indicators.

1.2 Advances Of Rio +20 – Promise And Disappointment

Twenty years after the first Earth Summit, the international community went back to Rio de Janeiro for The United Nations Conference on Sustainable Development (Rio+20) to reinforce political commitment for sustainable development, assess its progress, identify gaps and address its new and emerging challenges. Within the context of sustainable development, the Conference focused on three themes which include: the institutional framework for sustainable development; a green economy and poverty eradication; and the thematic areas that would comprise the Plan of Action.

One of the main outcomes of the Rio+20 Conference was the agreement by Member States to establish an intergovernmental process to develop a set of Sustainable Development Goals (SDGs), to be ´action-oriented, concise and easy to communicate, limited in number, aspirational, global in nature´ to help drive the implementation of sustainable development. Obviously, the progress towards these goals is to be measured by sustainability indicators. SDGs would build upon the Millennium Development Goals (MDGs) that were established following the Millennium Summit of the United Nations in 2000 and converge with the Post 2015 Development Agenda. There is broad recognition that the eight MDGs are unlikely to be achieved.

The main outcome document that came from the Rio+20 conference was entitled ´The Future We Want´ (UN, 2012). This document mandated the creation of an inter-governmental Open Working Group (open to all stakeholders) of the General Assembly that is tasked with preparing a proposal for Sustainable Development Goals for consideration and appropriate action to be agreed by the United Nations General Assembly. The Conference also adopted guidelines on green economy policies and member states decided to establish an intergovernmental process under the General Assembly to develop options on a strategy for financing sustainable development. Governments also requested the United Nations Statistical Commission to launch a process to assess development progress that could complement the gross domestic product, and could better inform policy decisions. The Conference also took forward-looking decisions on a number of thematic areas, including energy, food security, oceans and cities. The Rio +20 Conference galvanized the attention of thousands of representatives of the United Nations system and major groups. It resulted in over 700 voluntary commitments and witnessed the formation of new partnerships to advance sustainable development.

However, while for many the first Earth Summit of 1992 carried a strong message of hope, Rio +20 attracted widespread criticism with claims that it offered little beyond what the original Earth Summit delivered, and that it was short of real commitments to change and failed to establish better governance to tackle global challenges. Expectations were low yet critics maintained that a simple rephrasing of 20-year old statements would never result in ambitious outcomes and that the agenda was overly dominated by assumptions of technocratic solutions and the voice of the private sector. By the final day, speculation among the press corps went as far as to suggest the Rio Earth Summit process was on its deathbed, there were no negotiations – no bust ups – and less access for the press. The Non-Governmental Organizations sector also seemed exhausted and disenchanted with the process.

Nevertheless, despite this disappointment the Summit did bring wider attention to sustainability and stimulated a large number of smaller new initiatives and activities. The parallel People's Summit attracted 15,000 indigenous and land rights groups, environmental activists, and trade unionists, and reflected a more positive and hopeful perspective. It was reassuring to see that Rio+20 did welcome the UN/FAO Committee on World Food Security (CFS) to consider agricultural and food issues. The CFS includes a model for bringing together both UN and non-UN intergovernmental agencies to address common issues and the expanded participation of social movements and civil society in intergovernmental negotiations. When the food crisis became apparent in 2008, peasant organizations, among others, called for the renewal of the CFS in Rome as preferable to UN-New York proposals to create an alternative body for food and agriculture on the other side of the Atlantic. Rio+20 explicitly endorsed the CFS's work on assessment of sustainable food production and food security at the national level, as well as its work on land tenure, fisheries and forests in the context of food security. While the formal outcomes of Rio +20 provide little ground for optimism, the value of the Summit is more felt through longer-term projects and initiatives, changes in attitudes and understanding, and hopefully may reach beyond 2015. Moreover, the aforementioned development of Sustainable Development Goals has also already resulted in tangible outcomes. In July 2014, the United Nations General Assembly's Open Working Group on Sustainable Development Goals, at the conclusion of the Group's thirteenth and final session at United Nations Headquarters, presented to the Assembly its proposal for new 17 Sustainable Development Goals[1]. They consider social, environmental and economic aspects to promote people's wellbeing and protect the natural resources[2] and they set development priorities for governments and businesses. However, in order for the Goals to contribute to real changes, funding agencies and academia must support this process and the right expertise must be involved at international and regional levels (Stafford-Smith, 2014).

2 Sustainability Indicators Today

2.1 Indicators In Policy, Private Sector And Science

Discourse on sustainability is widespread across the public and private sector, from individual projects and companies, to municipalities, cities, regions, and countries. Sustainability has now reached a "buzzword" status in both science and policy, and although the reasons and the drivers for this popularity are different depending on the actors and circumstances, the concept is now embedded in political and scientific agendas worldwide. On the one side, multiplicity of approaches to sustainability is not necessarily bad, as it shows that some progress has been made since the Rio Earth Summit underpinned by a general recognition of the role of sustainability indicators in fostering sustainable development. However, sustainability indicators are often not used appropriately, are weakly grounded and subject to frequent misinterpretation and misuse (Bell and Morse, 2008).

There is little consensus on a common set of indicators and the criteria that should be used to determine what qualifies as a good indicator. There are few commonly recognized assumptions and underlying concepts on the use and evaluation of indicators and there is often little agreement on their scientific basis. Sustainability indicators therefore need to be flexible and adapted to the real world, and not give the impression of offering a black and white, silver-bullet approach (see also chapter 1). Although some claim that the complexities associated with the use of sustainability

1 http://www.foresteurope.org/news/final-proposal-sustainable-development-goals-and-targets-agreed

2 http://www.un.org/en/development/desa/newsletter/desanews/feature/2014/08/index.html#11715 indicators and sustainability itself justify lack of progress towards sustainability, others state that we already know enough about what is unsustainable and that this appreciation is widespread (Sir Robert Watson, broadcasted interview - BBC). But the concept of sustainability also may lead to misuses – everybody wishes to show that their work, company or department is sustainable leading to the risk of 'green washing'.

3 Why Sustainability Indicators In Practice?

> Oh please! Not again new indicators! I only want to see simple indicators that can be used by politicians and let the scientists stop with even more complicated stuff!
>
> A very high UNEP official (adopted from Meadows, 1998)

Currently, a multitude of different indicators exist and they are used in different contexts, for various purposes by inter and non-government organizations, national and regional authorities, private sector and in academia. It is not the purpose of this book to review the range of indicators currently in use, but rather to critically discuss their application in practice across a variety of environment and development projects and initiatives and how their use contributes to public and private sector decision making. It is also not the objective of this book to propose new indicators and reject others (although discussion on preferred indicators in certain situations is presented) but to look into their use and contexts in practice, and discuss the reasons for different applications.

A lot of criticism of the use and design of sustainability indicators has already been written (e.g. Wilson et al., 2007; Ramos and Caeiro, 2010; Agol et al., 2014) with many commentators finishing their critique by proposing a new set of indicators that are deemed to be preferable to the last. Practitioners commonly struggle to apply sustainability indicators in practice due to various, often contradictory pressures, such as requirements of funders, public perception, time and financial constraints to number just a few. In this book we take a closer look at the use and misuse of sustainability indicators in practice, and discuss what has and has not worked and why.

We invited specialists from different parts of the world who have experience with designing and implementing sustainability indicators in practice in a broad range of projects from conservation, reforestation, agriculture, water and wastewater management to air quality control. In drawing on these rich case studies and perspectives, the book identifies some of the most common challenges and opportunities presented in applying sustainability indicators to a diverse range of circumstances. Whilst we are restricted in what can be included in one volume we believe that the selected examples presented here illustrate a range of circumstances, approaches, their challenges and advantages.

We view a sustainability indicator as something that aims to capture the measure of sustainability – i.e. progress towards sustained social, environmental and economic outcomes. In doing so we embrace a systems approach, recognizing that different aspects of a system are tightly interlinked and that for an indicator to confer useful information about sustainability it has to provide a long-term perspective. For example, a number of planted trees in a reforestation project is a traditional indicator but survival rate would tell more about actual long term impacts of the project, thus can better represent sustainability.

Ultimately progress towards sustainable development depends on the combined decisions of individual people. Without understanding the choices of individuals sustainability actions at other levels have little effect. This is where the idea of this book was born and the ‘practicality’ of indicators is discussed throughout all chapters. We are aware that the topic of sustainability indicators is a daunting experience and the contributions of this book clearly illustrate that many challenges associated with practical use of indicators remain, and guidelines continue to be neglected.

Although indicators are only partial reflections of reality, they form a necessary part of the information we use to understand what is happening around us, make decisions and plan for the future actions (Meadows, 1998). Indicators do not guarantee results, but the desired results are unlikely to happen without appropriate indicators.

This volume provides a handbook of lessons learned from various case studies worldwide on practical use of sustainability indicators, and we hope that you find it useful.

4 Contents

Chapter 1 - What Are Sustainability Indicators For?

This chapter discusses the purpose of sustainability indicators, describes the features of good sustainability indicators, and highlights past examples of good uses of sustainability indicators. The chapter begins with a discussion on different definitions and understanding of “sustainability” that guides the discussion on the purpose, quality, and history of indicators. The chapter also discusses why progress towards sustainability should be measured, whether in quantitative or qualitative ways. Moreover, based on scientific literature, a set of examples of uses and misuses of indicators is provided. This is followed by a discussion of the challenges of measuring sustainability indicators.

Chapter 2 - Sustainability Indicators In Complex Socio-Ecological Systems

Chapter 2 outlines our current understanding of indicators and monitoring for sustainability in the context of complex social-ecological systems. The chapter first gives a general introduction to social-ecological systems thinking, then reviews the implications of social-ecological systems thinking for the design and interpretation of (any) indicator being used to measure and promote sustainability, and finally it explores ways in which sustainability indicators themselves, due to the complex, adaptive nature of the societal systems with which they interact, can change perceptions of values and goals (for better or worse).

Chapter 3 - Biodiversity Indicators And Monitoring For Ecological Management
This chapter presents a broad overview of some of the key features of any process to monitor and evaluate biodiversity. Selection of appropriate indicators is a central part of this. Yet as is the case for the assessment of any indicator, good biodiversity indicators represent only a necessary, yet insufficient condition for a monitoring process to provide the kind of support necessary to foster improvements in sustainability. The chapter briefly identifies ways in which biodiversity monitoring can be most effective in facilitating and guiding any management process. The chapter focuses on the importance of first thinking about the why and what of biodiversity monitoring, as well as the ways in which monitoring activities fit within a wider framework of the management system itself – whatever that management system may be. Following this an overview of different types of indicators that can be used to support a biodiversity monitoring program is presented, including different ways to assess the status and trends of biodiversity. The chapter ends with practical considerations regarding the human resources necessary for biodiversity monitoring to work on the ground.

Chapter 4 - Monitoring REDD+ Impacts: Cross Scale Coordination And Interdisciplinary Integration
The objective of this chapter is to examine possibilities for more integrated monitoring of the carbon and non-carbon impacts of reducing emissions from deforestation and forest degradation and enhancing carbon stocks (REDD+). Since the climate impact from reduced emissions (and increased removals) is the centerpiece of REDD+, countries are asked to set up systems to monitor changes in forest carbon stocks for reporting at the international level. The multidimensionality of REDD+ poses great challenges to identifying efficient trade-offs between in-depth, fully comprehensive monitoring and increasing complexity and costs, which is a serious problem given the limited funds available for REDD+ monitoring. Monitoring both the carbon and non-carbon impacts of REDD+ requires development of systems that are scientifically sound, yet simple enough to be implemented effectively. In this chapter, the authors first present key concepts in monitoring as related to REDD+. They then review available options for carbon monitoring, social monitoring and environmental monitoring, with particular attention to issues of scale. Finally, they present strategies for moving forward through a more integrated REDD+ monitoring across scales and between disciplines, which can go beyond REDD+ to inform approaches for measuring sustainability in landscapes. Integrated monitoring of REDD+ performance is not only

important for assessing adherence to safeguards, but can go well beyond REDD+ to inform indicators of sustainability towards promoting benefits for both people and the environment.

Chapter 5 - Measuring Indicators For Sustainable River Basin Management
This chapter discusses the complexities associated with measuring sustainability in river basins. With a case study of the River Nyando, which drains into Lake Victoria Basin, Kenya, the chapter highlights key concepts in sustaining the river basin such as ecosystem services, decentralization, multi-stakeholder participation and institutional arrangements. It identifies sustainability indicators for water quantity and quality, biodiversity and public participation and discusses the different approaches used to measure them, opportunities and shortcomings.

Chapter 6 - Sustaining Local Livelihoods Through Coastal Fisheries In Kenya
This chapter covers past and present strategies for managing coastal fisheries in Kenya. It discusses how coastal fisheries management has evolved in Kenya, where from the 1990s, there has been a paradigm shift from top-down to bottom-up approaches which embrace local community participation. It emphasizes the importance of the fishery sector in Kenya and the need to sustain the sector for improved food security and livelihoods of the dependent local communities. It discusses key strategic approaches used to sustain coastal fisheries in Kenya such as community conserved areas (CCA) and Beach Management Units (BMU) and highlights their strengths and weaknesses taking into account sustainability indicators. The importance of livelihoods diversification and local capacity building are also highlighted and key lessons learnt are outlined.

Chapter 7 - Peninsular Pronghorn Conservation: Too Many Paradigms, Too Few Indicators
In chapter 7 the various threats to pronghorn (*Antilocapra americana peninsularis*) are brought to light. Subsequently, short-term solutions were identified in a knowledge system experiment in relation to assisted reproduction. As to longer-term solutions, recent conservation literature points to rewilding and stewardship as two hitherto unconnected but possibly complementary wildlife management avenues. Wildlife conservation is one of the tenets of environmental sustainability. Efforts in recovering the population of *peninsularis* pronghorn however seem inconclusive. This is far from an isolated case and warrants a deeper examination than usually afforded in the course of practical animal conservation. Based mainly on fitness, food and habitat information gathered during the year posterior to introduction on an island of a captive and free-roaming population, it seemed that specialization in advisors had come with different conservation paradigms. Taken together they adversely affected individuals and population. Foremost were the zoo, veterinary, ranching and hunting paradigms. Perhaps more surprisingly, non-governmental

organizations’ activity also played a role, in a process possibly headed toward privatization and domestication.

Chapter 8 - Restoration Success Of Tropical Forests: The Search For Indicators
This chapter aims to discuss how restoration success is being measured and to find a role for functional ecology in providing reliable indicators for restoration ecology. The objectives of this chapter are threefold: to present the main ideas for the evaluation of restoration success and the indicators used; to discuss the main advantages and drawbacks of the main strategies of restoration - active and passive; and to emphasize the need for a more widespread use of functional approaches to evaluate success in restoring tropical forests. Given the difficulties associated with current indicators of restoration success based on species diversity, vegetation structure and ecological processes, it is extremely timely to consider that functional approaches play an important role in providing reliable and simplified indicators for restoration success. The use of such indicators can catalyze more restoration initiatives, because they offer insurance that such efforts will in fact accomplish their initial goals, as to provide ecosystem services, contribute to biodiversity conservation and increase ecosystem resilience in response to climate change.

Chapter 9 - Sustainability Indicators In Brazilian Cattle Ranching
Brazil is one of the largest agricultural producers worldwide and agriculture is one of the backbones of the country’s economy. The country also owns the largest commercial cattle herd with 211 million heads, responsible for about a quarter of the total volume of meat transacted in foreign trade supply. Opposite to western-style intensive agriculture that is often associated with biodiversity loss and environmental pollution, in Brazil extensive low productivity agriculture often leads to environmental degradation. Similarly, Brazilian pasturelands are characterized by low stocking rates and this low efficiency has historically led to deforestation and to other adverse effects on the environment such as soil erosion. In this chapter we discuss the reasons for unsustainability of Brazilian cattle ranching and indicators to measure progress towards sustainability.

Chapter 10 - Sustainability Indicators For Agriculture In The European Union
Chapter 10 presents a range of aspects associated with sustainability indicators used for agriculture: it highlights the need for monitoring of agriculture worldwide, describes beneficial and harmful effects of agriculture on the environment and society and discusses problems related to farmers´ activity towards sustainable agriculture in the European Union (EU). Different definitions of and approaches to sustainable agriculture including ecological, economic, social and political dimensions are discussed. The chapter also presents a set of agri-environmental indicators used by the Organisation for Economic Co-operation and Development (OECD) and within the EU – IRENA project (Indicator Reporting on the integration of Environmental

concerns into Agricultural policy). The origins of the concept of sustainable agriculture in Europe, historical and recent trends regarding agriculture and steps towards sustainability and ‘land sparing vs. land sharing’ are also discussed in the chapter. Finally, the reader can find a list of 28 different agri-environmental indicators proposed by the European Commission with the short explanation of domain and sub-domain that they represent. The chapter complements with a discussion on which agri-environmental indicators can be considered good indicators and why. A range of examples of sustainability indicators and the process for their selection in the EU are presented along with recommendations on their use.

Chapter 11 - Sustainability And Air Quality
The main objectives of the chapter are: (i) to highlight the main problems and main instruments of managing air quality in Europe, (ii) to present general reflections on linkages of air quality with sustainability issues and (iii) to analyze the case study of a specific air quality problem with domestic heating in Poland with special attention to sustainability indicators. Many air quality aspects are strictly linked to sustainability, such as (i) harmful impacts on people and ecosystems, (ii) material losses due to pollution, (iii) connection to the climate change policy, (iv) long range pollution transport, (v) control strategies, including links to energy policy and transport system, selection and optimization with cost-benefit analysis. In Europe air quality is one of the main threats to environmental and human health and air pollution is high especially for particulate matter (PM), nitrogen dioxide (NO_2) and ozone (O_3). Poland has problems with dust pollution (PM_{10}, $PM_{2,5}$ and benzo(a)pyrene) and it is estimated that every year about 80% of people living in the Polish towns/cities are exposed to the significant harmful impact of PM pollution. Large effort has been put into emission reduction actions but the air quality has not showed improvement. The chapter discusses further measures and recommendations that require implementation of indicators.

Chapter 12 - How To Measure Wastewater Systems’ Sustainability?
Wastewater collection by pipelines together with the proper treatment system is undoubtedly the most relevant way to deal with the environmental threats that could be caused by wastewater. Large-scale investments in construction of wastewater-systems are being currently realized in Poland. Wastewater investments are rather capital-intensive and therefore the areas to be covered by the pipelines and connections must be chosen very carefully in order to not produce exceeding costs. Although the methodology for selecting the areas that meet certain conditions is known, sometimes it is not implemented by the local authorities. This can lead to an increase of investment costs. The chapter discusses that the wastewater system could be considered sustainable only if all the costs (investment costs and running costs) are covered by the wastewater tariffs and that they are calculated and paid by the end users (society), and that the tariffs must be low enough for all the end users to bear

the costs of wastewater collection and treatment (tariffs). The chapter also discusses that in some cases the need for environmental protection stands opposite to economic and social aspects, for example in poorer regions and rural areas – where the unit costs of constructing wastewater systems are higher due to low-density housing. The chapter shows the case study of the single company operating on the territory of ten southern Poland Districts, covering mainly rural areas. The chapter also shows some methods to eliminate negative effects of costly investments on poorer parts of society and demonstrates implications for sustainable wastewater management.

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1 What Are Sustainability Indicators For?

Rachael Garrett and Agnieszka E Latawiec

1.1 Introduction

Indicators are critical to both scientific inquiry and policy development in complex systems. They are concise information systems that provide quantitative and qualitative information about the condition and trajectory of a system and why certain trends occur in specified contexts (Bell and Morse, 2008). To date a wide range of sustainability indicators have been proposed by different authors and organizations (Bell and Morse, 2008; Moldan et al., 2012). The selection and use of specific indicators from among these myriad choices depends on a range of factors, including values about the goals of such indicators and appropriate temporal and spatial scales of assessment. One cannot use every indicator potentially available, so an element of simplification, while maximizing unique and relevant information, is essential. Due to these value differences regarding objectives and scope, the selection of sustainability indicators will undoubtedly involve substantial discussion within an organization. The selection of indicators will also be influenced by the availability of resources, time constraints, and data. Due to these reasons there can be no a priori "best set" of sustainability indicators within a particular sector or region. Nevertheless, the goals of this chapter are to help improve the selection of indicators for sustainability science and policy by: i) Discussing the purpose of sustainability indicators, ii) Describing the features of *good* and *effective* sustainability indicators, and iii) Presenting examples of sustainability indicators that illustrate a range of trade-offs associated with their use in practice. Before embarking on this task we briefly contextualize sustainability and begin with a definition of "sustainability" that will guide our discussion on the purpose and quality of indicators.

1.2 Components And Interpretations Of Sustainability

Sustainability is a word used broadly in scientific and policy spheres to describe conditions that do not damage the environment or degrade ecosystem services (Parris and Kates, 2003). Over the last twenty years, numerous researchers have discussed the problematic nature of the word sustainability used in this broad sense, highlighting important questions such as *what exactly should be sustained* and for *whom, when, and why* (Costanza and Patten, 1995; Parris and Kates, 2003; Marshall and Toffel, 2004). More specifically these authors ask: i) who decides what should be sustained? ii) over what time frame should it be sustained?, and iii) for what purpose?

Almost every article or book on sustainability expresses disappointment that the concept of sustainability lacks consensus. For example, Lynam and Herdt (1989) state that sustainability is ‘the capacity of a system to maintain output at a level approximately equal to or a greater than its historical average, with the approximation determined by the historical level of variability’. Pearce and Turner (1990) claim that sustainability means ‘maximizing the net benefits of economic development, subject to maintaining the services and quality of natural resources over time’. More recently, Hak et al. (2007) defined sustainability as ‘the capacity of any system or process to maintain itself indefinitely’.

Coupled with the word development, however, the term sustainability provides a slightly clearer normative and anthropocentric goal of how to use resources. Using Arrow et al. (2012)’s definition: sustainable development is development that sustains, i.e. does not decrease, the wellbeing of the current generation as well as the potential wellbeing of all future generations. This definition helps clarify the ‘who, when, and why’ of sustainability. It also provides policy goals that are slightly more ambitious than just simply not ‘compromising the ability of future generations to meet their own needs’ (Bruntland, 1987), by specifying that policies intended to promote development should leave future generations with ‘as many opportunities as we ourselves have had, if not more’ (Serageldin, 1996).

The concept of wellbeing encompasses individuals’ capacity to achieve happiness, harmony, identity, fulfillment, self-respect, self-realization, community, transcendence, and enlightenment (Meadows, 1998). It involves access to security, health, material needs, good social relations, and freedom of choice (MEA, 2005). It is inherently relational, and takes into account equity, sufficiency, and quality (Meadows, 1998). To ensure non-decreasing intergenerational wellbeing it is necessary to maintain the assets and stocks that provide the goods and services essential to wellbeing (Arrow et al., 2012). Managing a stock to provide the continued satisfaction of our wants and needs inherently involves protecting the throughputs that replenish that stock (Daly, 1991).

We can divide the assets that must be maintained into five major categories:

- *Natural capital* is the quantity and quality of environmentally provided assets (such as soil, atmosphere, forests, water, wetlands, mineral resources, biogeochemical cycles, etc.) that provide a flow of useful goods or services (Serageldin, 1996). The “ecosystem services” provided by natural capital include provisioning of food, water, timber, and fiber; regulating climate, floods, disease, wastes, and water quality; culturally related recreational, aesthetic, and spiritual benefits; as well as soil formation, photosynthesis, and nutrient cycling processes that support other natural capital services (MEA, 2005). Natural capital can also be perceived as the ultimate, non-substitutable stock underlying all other capital stocks (Daly, 1991; Meadows, 1998). Humans can build a water filtration plant to provide the same services as a forest, but we cannot create water out of nothing.

- *Human capital* is the quantity of the human population (size, age structure and geographic distribution), and the quality (health and capability) of that population (Serageldin, 1996).
- *Knowledge capital* includes collective public awareness of how and why things are as they are (formal scientific knowledge) as well as how to fulfill human purposes in a specifiable and reproducible way (experiential technological and managerial knowledge) (Brooks, 1980; Raymond et al., 2010). The components of human and knowledge capital defined here are often combined under the heading of human capital.
- *Social capital* encompasses norms and institutions and emerges from interactions between people or between people and organizations or the market. Institutions include official policies as well as informal rules, while norms include expectations about behavior, such as reciprocity and trust (Ostrom, 1986; Roseland, 2000; Ostrom, 2009).
- *Manufactured capital* is the quantity and quality of physical stock that is created by humans, to provide goods and services, such as roads, houses, machinery, cars, and medicine (Serageldin, 1996).

The economy is in a “steady state” when natural, human, and manufactured capital are non-decreasing (Daly, 1991). Development is not sustainable when wealth, measured as the sum of all assets, weighted by their marginal contribution to wellbeing, is decreasing (Arrow et al., 2012). An economy that is “developing” is one in which natural, human, and manufactured stocks are non-decreasing, while social and knowledge capital are increasing (Daly, 1991), so long as increases in social and knowledge capital are contributing positively to human wellbeing.

In the selection of relevant indicators of sustainability it is important to note that some assets are substitutes, some are complements, some are both (Serageldin, 1996). Manufactured capital is undoubtedly the most substitutable stock since we create this capital from other asset groups, predominantly natural (energy), human capital (labor), and knowledge (technology). Natural capital is perhaps the least substitutable of all assets. Not only is it impossible to replace the natural capital that provides services that are directly essential to our wellbeing, such as healthy food, clean water, and clean air, but it is also impossible to replace the underlying ecosystems services that support the natural capital that provides these essential services (MEA, 2005). For example, a fishery policy that contributes to sustainable development would not only restrict harvesting, but also would protect the marine ecosystem of that fishery from damage that might harm the fish populations capacity to reproduce.

According to the capital asset theory, instantaneous and intergenerational wellbeing will move in the same direction when the economy is in a steady state (Arrow et al., 2012). For more discussion on intergenerational relations and wellbeing see chapter 2. It is also assumed, implicitly, that increases in all assets will be distributed equally. In reality it is quite likely that the total asset base for a country could stay

Figure 1: The 'Daly Triangle': relates natural capital with human wellbeing as the ultimate human purpose through science, technology, politics and ethics. Adapted from Meadows (1998).

constant while the distribution between individuals within that country changes substantially. It is even possible that some people could see their access to certain assets decreasing even as the total asset base increased. In this case, it becomes less clear that the total wellbeing of the country would be constant. Thus, distribution of assets also matters in the selection of sustainability indicators and evaluations of sustainable development (Valentin and Spangenberg, 2000).

While we have focused on a wealth-based definition of sustainability, it is worth noting that not all uses of sustainability indicators need focus on wealth accounting approaches. It may be equally functional, and less redundant to focus the study of sustainability on specific sectors and regions and to select clear indicators within these sectors and regions that can measure a clear deviation from sustainable pathways within the larger context of sustainable development (Kaufmann and Cleveland, 1995).

1.3 Why Do We Need Sustainability Indicators?

Indicators serve two major roles in the field of sustainability science. First, the selection of good sustainability indicators (or metrics) can help clarify causal relationships between specific capital assets and intergenerational wellbeing, improving knowledge about social-ecological systems as an end in and of itself. Second, the creation of good sustainability indicators can greatly aid policy and management decision-making. These roles are highly interconnected since the proper identification of causal relationships between capital assets and wellbeing in social-ecological systems can help elucidate trade-offs in wellbeing from enhancing or depleting different capital stocks.

Sustainability indicators can be drawn from a wide range of economic, social or environmental sources (Hak et al., 2009) and may contribute to all five stages of policy analysis: i) Clarifying goals, ii) Describing trends, iii) Analyzing conditions, iv) Projecting developments, and v) Inventing, evaluating, and selecting alternatives, so long as they are concise and easy to interpret (Clark, 2002). Nevertheless, there is a variety of challenges associated with selecting and using sustainability indicators. Some of these challenges mimic the definitional ambiguities of sustainability itself, such as what is the right time scale over which to collect or apply indicators and who should select these indicators. Sometimes the 'right' indicators are used in the 'wrong' context, a situation described frequently throughout the chapters of this book.

The selection of indicators is inherently driven by values about the who, when, and why questions outlined above; values that can differ substantially across stakeholders (Meadows, 1998). The selection of indicators is also influenced by conceptual understanding of the connections between the stocks to be sustained and human wellbeing. Therefore, negotiating indicators within a group early on in the policy evaluation process is particularly important for clarifying conceptual frameworks and goals across groups with differing scientific backgrounds and values. One major conceptual difference that will likely influence the selection of indicators is whether stakeholders believe that all capital groups are substitutable (Getzner, 1999).

Sustainability indicators are useful in describing trends when they capture variation in both time and space about changes in the quantity or quality of capital assets and human wellbeing. In that respect, sustainability indicators provide a measure of the effectiveness of actions and policies at moving a system towards a more sustainable state (McCool and Stankey, 2004). Complementary to evaluating the *magnitude* of a stock, an indicator can also be designed to measure the *rate of change* in that stock (Bossel, 1999). Demonstrating rates of change may aid understanding of the system dynamics (and most of the systems that indicators assess are dynamic ones).

Indicators may also be selected to estimate future changes. This is especially relevant given that social-ecological systems tend to be characterized by temporal and spatial delays and nonlinear dynamics (see also chapter 2). Many complex natural resource systems also present delays between the occurrence of an event (such as a policy initiative or project intervention) and the effect, which leads to both advantages and disadvantages. For instance, long delays between actions and the result make it more difficult to draw cause-effect connections. Indicators that can offer insights to future threatening conditions (such as the size of the ozone layer over Antartica) can provide important lead time during which mitigation policy interventions can be proposed and initiated.

Along these lines, sustainability indicators are now increasingly used to perform project impact assessments (Agol et al., 2014). Project impact assessments focus on the effects, rather than project management and delivery, and typically occur after project completion. Project impact assessments, if performed adequately, may provide useful information to project executors, funders, and the target community to monitor and evaluate the effects of their actions towards sustainability. Sustainability indicators may also be incorporated into assessments that evaluate the potential impact of a project before it is funded to assess which projects are likely to lead to the largest overall improvement in intergenerational wellbeing.

Indicators can also be used for strategic environmental assessment. Donnelly et al. (2007) showed an interesting approach to evaluate performance of indicators for strategic environmental assessment during a workshop gathering a multi-disciplinary team to incorporate differing viewpoints and to ensure less bias in the decision-making process. The indicators included biodiversity (e.g. number of sites with habitat enhancement), air (number of exceedances of air quality limits), water (minimize culverting of watercourses) and climatic (insurance claims due to flooding) indicators. Although the degree to which those indicators were able to show trends and provide early warning mechanisms varied, most of the indicators were found to be policy relevant, cover a range of environmental receptors, were adaptable and understandable (Donnelly et al., 2007). The following section of this chapter extends the discussion on features of a good indicator identified in literature.

1.4 What Characterizes ‘Good’ And ‘Effective’ Sustainability Indicators?

It is impossible to definitively categorize individual indicators as good or effective in all settings; some indicators might be useful at certain times and scales, but not useful in others. Furthermore, the definitions of good and effective are highly subjective. Nevertheless it is still possible to highlight some of the features that indicators should have if they are going to improve scientific understanding of complex systems and the selection of policies for sustainable development.

Generally speaking, indicators of sustainable development must capture information about the quantity and quality of the underlying asset base that is to be sustained for the ultimate goal of ensuring human wellbeing (Meadows, 1998). Good sustainability indicators should also assess whether the relative contributions of different assets to wellbeing are changing over time. Since the effectiveness of an indicator in sufficiently capturing this information may change over time as the context of the system changes, it is necessary to continually monitor, review and evaluate selected indicators over time (Ramos and Caeiro, 2010).

More specifically, indicators should be simple, measurable, feasible, flexible, dynamic, and user-inspired.

- Simple: easily communicated. Reducing the volume and complexity of information is often required by decision makers (Donnelly et al., 2007). While the use of simple indicators may sometimes be perceived as a reductionist approach to sustainability science, this critique is really only valid if these indicators are ultimately used in isolation. Simple indicators can be used in complex combinations that capture much more information about the system.
- Measurable: capable of being quantified.
- Feasible: able to be collected (Bell and Morse, 2008). This is a slightly different requirement than being measurable, since something can technically be measured, but collection would require time and money beyond the capacity of the organizations or individuals involved.
- Flexible: to allow replacing with new available data (Ramos and Caeiro, 2010).
- Dynamic: capturing changes in stocks and flows over time. This is necessary to capture trends, but also non-linearities and causal processes within a system. Sustainability intrinsically involves the maintenance or continuity of outcomes over time. Any indicator that just looks at the present flows, without talking about the future, and thresholds or changes in the stocks that produce those flows is not really capturing intergenerational wellbeing (Merkle and Kaupenjohann, 2000).
- User-inspired: indicator properties should align with the goals of its users and be co-produced by these users when possible (Mitchell, 2006). Do the users care about diagnosing progress toward sustainable development, communicating progress, or assessing cause and effect within a system?

Spangenberg (2002) also proposes that indicators should be: i) general, i.e. not dependent on a specific situation, culture or society; ii) indicative, i.e. truly representative of the phenomenon they are intended to characterize; iii) sensitive, i.e. they have to react early and sensibly to changes in what they are monitoring, in order to permit monitoring of trends or the successes of policies, and iv) robust, i.e. directionally safe with no significant changes in case of minor changes in the methodology or improvements in the data base. There is also extensive discussion on validity of indicators to guarantee their credibility (Bockstaller and Girardin, 2003).

Indicators are not objective and, in fact, they do not need to be (as long as they are adequate and reflect assumptions behind sustainability). Indicators based on numbers are however usually considered more valuable and reliable than qualitative assessments, and they can be more easily communicated and validated. As Meadows (1998) notes however paying attention to only what is measurable is itself a subjective choice.

Unfortunately, some of the characteristics of 'good' indicators outlined above may present contradictory goals. For example, indicators that are easy to measure directly and easy to communicate may not adequately reflect complexity (Agol et al., 2014). Indicators that are *dynamic* (capture changes over time), might be *infeasible* given money and time limitations. The use of secondary data aggregated to the higher levels may allow for the feasible capture of dynamic indicators, enabling sophisticated modeling that captures complexity, interactions and feedbacks over a long term (e.g. Dizdaroglu and Yigitcanlar, 2014). Yet, these data may not reflect the intricacy of real-world factors at a more fine scale (e.g. individual household). Complex and rigorous indicators are rarely replicable (their appropriate application requires time, financial resources and often skilled staff to gather data and perform modeling).

The 'Ecological Footprint' indicator provides a good example of these tradeoffs. The Ecological Footprint measures environmental impact by the amount of land that a person, city, industry or a country requires for its maintenance (Rees, 1992). It converts the flows of energy and matter used to produce an item into corresponding quantities of land and water required to support these flows, expressed in area units. It captures many useful ideas within one number to express sustainability, however it requires considerable scientific review to codify its calculation (consumption of biomass, energy, water and other resources are converted into a normalized measure of land area). Researchers have used this indicator to demonstrate that current consumption practices are not sustainable and to argue that major transformative changes in the global economy are necessary to reduce society's collective ecological footprint (Hoekstra and Wiedman, 2014). However others (Van Den Berg and Grazi, 2013) have argued that the Ecological Footprint is simultaneously too complex and too simple since it involves large-scale analysis of energy and matter flows across whole nations, yet focuses on only two inputs (total land and water), thereby ignoring toxic substances, noise pollution, and fragmentation of ecosystems. Furthermore it has been argued that the Ecological Footprint indicator does not provide information about the linkages between institutions, technologies, and ecological outcomes, so it cannot contribute to better understanding of what policy interventions might be undertaken to reduce footprints (Van Den Berg and Grazi, 2013).

In sum, it is more useful to think of the tradeoffs between different indicators, rather than attempting to address all features of 'good indicators' simultaneously. Therefore, the goals and tradeoffs among different indicators should be discussed explicitly among stakeholders. Explicit consideration and discussion among stakeholders of tradeoffs of different indicators, and the definitions and objectives of sustainability themselves, can enhance the potential utilization of such indicators (Binder et al., 2010). No less important are normative decisions about how to weight indicators, which should also be explicitly discussed (Binder et al., 2010). Ultimately a matrix of indicators sorted under different types of competing objectives can help overcome the limitations of single approaches and promote policy analysis of tradeoffs (Palm et al., 2013). Additional discussion on these tradeoffs can be found in literature (e.g. Utne, 2007) as well as in this book (chapter 6: Sustaining Local Livelihoods through Coastal Fisheries in Kenya).

1.5 Conclusions

Sustainability *indicators* help us understand whether or not the capital assets on which intergenerational wellbeing depend are decreasing in quantity or quality over time (Daly, 1991). They can also tell us whether the marginal contributions of these stocks to wellbeing are changing over time. Indicators are only partial reflections of reality, based on imperfect models loaded with uncertainty (Meadows, 1998), yet they are still necessary for policy evaluation. The world is too complex to make decisions without some level of simplification to direct us to the right decisions. Despite discrepancies in definitions and interpretations, as present in many other scientific areas, we can learn from sustainability indicators use (and misuse) in practice to improve the assessment process. The following chapters will illustrate many more examples of sustainability indicators from many different sectors and scales.

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2 Understanding Indicators And Monitoring For Sustainability In The Context Of Complex Social-Ecological Systems

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2.1 Introduction

It is widely accepted that ecosystems across the world are increasingly affected by humans. Many earth system scientists contend that we have entered a new geological epoch, the Anthropocene, in which humans exert a dominating influence on many key earth system processes (Crutzen, 2002; Steffan et al., 2007; Rockström et al., 2009). Research on the characteristics of this new era emphasizes that a) the world is interconnected and thinking of a given study system as being made up of both social and ecological attributes that interact in complex and adaptive ways can help us make sense of these interactions; and b) the speed of environmental change introduces novel institutional challenges, such as the need to grapple with cross-scale interactions where the activities of one community or society can have far reaching effects on another, thousands of kilometers away (Galaz, 2014 pp: 15). This perspective provides the starting point for why and how, in our view, sustainability indicators should be developed in a way that takes account of the complex and continuously changing nature of the systems they are trying to assess.

This chapter begins with a general introduction to some of the key concepts that have emerged from thinking about complex adaptive systems. These concepts highlight some of the considerations that should underpin any attempt to monitor changes in a set of focal attributes that cannot be disentangled from the wider system within which they exist. We then provide a brief introduction to social-ecological systems thinking that explicitly recognizes the highly interdependent and cross-scale nature in which social and ecological attributes of a system are often connected. We posit that social-ecological systems thinking can provide invaluable guidance in designing monitoring and evaluation systems for assessing how different (interconnected) social and ecological attributes of a system are changing as we monitor progress towards, or away from, sustainability. In adopting such a systems approach we conclude with a discussion on the ways in which sustainability indicators themselves, as interdependent parts of the system they are designed to measure, can ultimately change perceptions of values and goals (for better or worse) regarding how that system should be managed.

2.2 Complex Adaptive Systems: The Whole Is Larger Than The Sum Of Its Parts

How does a framing of complex adaptive systems help us manage systems for change? First, by framing the 'things' we study as systems, we can set a boundary on what interactions we wish to analyse, monitor or change. A systems framing implies that cross-scale interactions are interdependent and understanding only one interaction in isolation from others will not provide an adequate explanation of the dynamics we seek to understand and influence. Further, adopting a systems perspective can help identify ways in which an isolated focus on monitoring and managing a single, or small number of attributes (e.g. human wellbeing, environmental health), can result in undesirable or even perverse outcomes, such as poverty traps (Carpenter and Brock, 2006).

A complex (versus a simple) system means that there are many working parts, and that many of these parts are connected through positive (reinforcing) or negative (balancing) feedbacks. Moreover, a complex system interacts across multiple spatial and temporal scales and is constantly changing and adapting in response to new pressures and influences, whether they be endogenous or exogenous. Complex adaptive systems (CAS) therefore have emergent non-linear properties, which make them unpredictable and subject to runaway feedback processes. Such a perspective reveals a number of properties that can help characterize the complex and interdependent relationships between humans and the environment in which we live, and which are typical of many landscapes around the world. Recognizing these general properties can help in design, implementation and interpretation of monitoring and evaluation processes to help foster progress towards sustainability.

a) CAS are highly unpredictable: Component parts of a system are connected in diverse ways and relationships are non-linear. This means that systems can approach and pass critical thresholds (or tipping points), beyond which system characteristics fundamentally change. For example, Lenton et al. (2008) and Lenton and Williams (2013) have identified nine potential policy-relevant tipping points in the Earth's climate system. One specific example is the dieback of the Amazon rainforest, which will have global effects on moisture recycling and carbon sequestration, and there is great uncertainty about how this may have far reaching effects on global climate patterns.
b) CAS can be contagious: what happens in one part of the system can have cascading, reinforcing or balancing effects on other parts of the system across various scales. For example, West Africa's monsoon shift may lead to a change in the greening of the Sahara desert.
c) CAS demonstrate modularity and redundancy: parts of the system are more connected within, rather than between nodes, and some repeat themselves. This means that CAS possess cross scale dynamic properties and indicators must be appropriately aligned to capture these processes. For example, some species may

play the same ecosystem function, and therefore have functional redundancy. This is incredibly important in the face of an unexpected event, a pest outbreak for example, and the existence of various species that carry out the same function can help maintain the function of the ecosystem.

d) CAS can self-organise: the direction of the trajectory depends on the system's history and initial conditions. Birds flocking is a typical example of a self-organising system.

These properties together help make up the general *resilience* of the system. Resilience is the capacity of social-ecological systems to continually change and adapt yet retain the same basic structure, composition and function (Folke et al., 2010). Resilient social-ecological systems have the potential to sustain development by responding to and shaping change in a manner that does not erode the possibility of exploiting future opportunities (Berkes et al., 2003). Resilience used in this general way has taken a prominent role in sustainability discourse. It is important to note that resilience is also characterized as the way in which a particular system responds to a perturbation. In this case, resilience is not inherently a good thing, and requires a normative judgment on which aspects of a system should persist in the face of change, adapt to unexpected surprise or actively transform into a new system configuration. Meanwhile, notions of sustainability emphasise the importance of fostering improvements in social wellbeing without undermining the critical environmental processes and services upon which the wellbeing of future generations depends. As such the two concepts should be seen as compatible rather than contradictory, as an understanding of social-ecological resilience can help in understanding the challenges and opportunities facing more sustainable development trajectories.

Any given study system is made up of multiple subsystems, and measuring or understanding changes in attributes of just one subsystem will not allow for an adequate representation of the wider system, and may lead to inappropriate or even maladaptive management practices. The characteristics of complex adaptive systems have profound implications for how we manage them and management processes can be improved by making them sufficiently adaptable and flexible that they can deal with uncertainty and surprise (Berkes et al., 2003).

2.2.1 Social-Ecological Systems

Thinking about social-ecological systems as a type of complex adaptive system arose from the growing understanding that social and ecological systems are inherently linked (Berkes et al., 2001). Coral reefs which are seemingly undisturbed by any direct interaction with humans might be affected by climate change, ocean acidification and sea level rise – all driven, in part, by the activities of humans in distant places and earlier time periods. Or likewise, urban dwellers, who have little direct contact with many natural resources, are critically dependent on ecosystem services like the provision of clean and plentiful water from upstream watersheds, clean air and cultural services such as the conservation of urban biodiversity in parks and recreation areas. Seen like this, nearly everything can be viewed as a social-ecological system, and understanding and monitoring the interactions between interconnected and interdependent systems is essential for achieving lasting and large-scale sustainable development goals.

The framing of social-ecological systems has also helped create a common meta-theoretical framework for scholars from across disciplines to engage in a meaningful way about the management of natural resources (e.g. Ostrom 2009; Poteete et al., 2010). The most recent work by Elinor Ostrom for example (2007; 2009) has adopted a social-ecological approach, tying together institutional scholars, sociologists, ecologists and practitioners with a framework that spans across disciplines and fields.

2.3 How Can Social-Ecological Systems Thinking Help In The Design Of Sustainability Indicators?

The proposition put forward by this chapter is that a systems approach is extremely useful in providing the perspective and context necessary for designing indicators that can provide transparent information on progress towards, or away from sustainability. Nevertheless, the very nature of complex social-ecological systems, with defining non-linear and cross-scale dynamics makes this task far from trivial. Here we posit four overarching and interrelated principles that can help guide researchers and practitioners in the design of indicators for sustainability.

Principle 1. Indicators are integral parts of a wider monitoring and management system

Indicators provide the key tool by which different elements of the monitoring and evaluation process can be logically connected to a given management regime. Discussions around indicators are often held out of context to the monitoring and management system to which they are intended to contribute. The connections are often implicit. The information provided by indicators is only of value if it feeds into a monitoring process managed by credible, legitimate and empowered individuals or institutions that are, in turn, connected to a system of resource management or policy design that is responsible for fostering progress towards sustainability. This is true whether the system is the entire world, such as the case of the Living Planet Index (managed by WWF and used as a key indicator of the status of biodiversity for a range of UN processes and conventions), or a specific conservation area, urban park, or conditional agricultural aid program. The ability to gather information that captures how valued attributes are changing over time, and in response to different interventions or pressures, is widely viewed as a key condition for the sustainable management of common pool resources, including water, fisheries, many timber and fuel-wood resources (Ostrom, 2009; Danielsen et al., 2013).

Principle 2. Indicators should be designed and used in combination with a suite of other indicators as a coherent part of a wider monitoring system

Despite the tantalizing appeal of a single magical index that can provide an adequate barometer of progress towards sustainability, the complex nature of social-ecological systems means that such an index does not, nor will ever, exist (see also section 4 below, and chapter 1). Instead, indicators can only have relevance to management and decision making processes when they are used in conjunction with other, related indicators, and in the context of a particular aim or objective (e.g. Niemi and McDonald, 2004 in the case of biodiversity). That said, the way in which indicators are combined is central to determining the value of the information they provide. Many environmental monitoring programs, for example, combine various types of indicators into uncoordinated simple lists, suggesting that they should be measured and reported together, and with little hierarchical or interactive structure (Kneeshaw et al., 2000; Rempel et al., 2004; Gardner, 2010). This is partly due to the superficial surveillance function that many environmental assessment programs play, with few if any incentives or hard laws motivating or obligating that they are designed in a more coherent and useful way (Lindenmayer et al., 2010). Some outcome level variables are assessed in isolation of any other indicators as a form of surveillance monitoring. This can provide a valuable function as an early warning system or “canary in the coal mine” that something is wrong. However, unless they are also linked to both indicators of how the system is managed, and background pressures and drivers, then it can be impossible to decipher why an observed change has taken place. In turn, it is also then impossible to determine how the information gained from a change in indicator value can be used to alter how the system is managed in some way, and protect the attributes that are the underlying source of concern.

To be effective in contributing with useful and coherent information for management and policy design, it is useful to consider two main types of indicators; those whose function is *prescriptive*, and those whose function is *evaluative*. Management policy and process indicators are both prescriptive in that they are used to measure or verify the existence or implementation of certain policies and management strategies. Whilst they only serve to prescribe a rule or norm they are nevertheless essential in ensuring that measurements of actual change in outcome level variables are linked to formal or de facto decision-making bodies and processes. These may include indicators like the cutting schedule of a managed forest, the sewage processing requirements for discharge into natural rivers, or social safeguards requiring the employment of a certain number of local people in an oil palm plantation.

By contrast, performance indicators are evaluative in the literal sense that they are used to evaluate changes in management performance and are made up of outcome level variables that are proxies of changes in the actual valued attributes of a system. These may include measures of social wellbeing such as education, health and income distribution, or environmental measures such as water quality indices or proxies of biodiversity. It is important to note that performance indicators can also be qualitative, such as an expert based scoring of air quality or worker satisfaction, where quantitative data are either unavailable or inappropriate.

Principle 3. It is essential to understand how different indicators relate to the wider system that is being monitored

Nothing exists in isolation and it is only ever possible to assess and monitor a small number of indicators that, in turn, relate to a tiny number of attributes and processes in the wider study system. Faced with such complexity it is very useful to develop a conceptual model to visually depict how the system of interest (be it a city´s water supply, managed wetland or system of national environmental accounts) is understood to be structured and function. Part of this conceptual model are the indicators of management interest, while the remainder depicts other system attributes and processes that may not be of direct concern, and may not all be measured, but which may influence or mediate changes in the valued attributes and indicators. These influences should encompass both immediate pressures – "fast" variables" – but also drivers that can play out over much longer timescales but which may ultimately have an overriding effect on system dynamics – so-called "slow" variables (Gunderson and Holling, 2002).

As an example we can think of the mechanism for land-based climate change mitigation activities that is REDD+: Reducing Emissions from Deforestation and Degradation. Distinct from some other climate mitigation mechanisms, REDD+ interventions act upon a complex and dynamic social-ecological system that operates across different interconnected land-uses, scales, time horizons, management sectors, governance arrangements and types of authority (Fig. 1). Although REDD+ is focused on the protection and enhancement of terrestrial carbon stocks (Fig. 1) – the primary indicator of concern – any REDD+ intervention will be both influenced by, and have unavoidable impacts on the provision of other (non-carbon) ecosystem services, in addition to having other social welfare implications. Ignoring these first and second order relationships and feedbacks would risk serious perverse outcomes that have plagued conservation interventions in the past (Parotta et al., 2012). Risks may include the displacement of deforestation to carbon poor but biodiversity rich regions and the potential for elite capture of any social and financial benefits provided by REDD+ (Parotta et al., 2012). These unintended consequences also have the potential to undermine the long-term sustainability of emissions-reduction targets by degrading social and ecological resilience. For example, forests that are protected from direct human impacts but host an impoverished biodiversity due to over-hunting of game animals may experience a shift in species composition towards more wind-

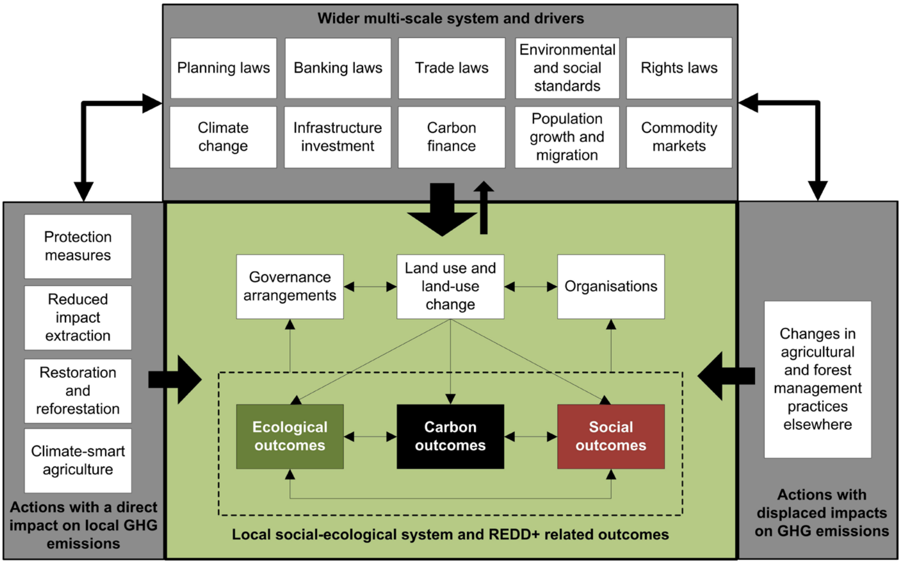

Figure 1: Conceptual figure of REDD+ consequences and effects operates across different interconnected land-uses, scales, time horizons, management sectors, governance arrangements and types of authority.

dispersed and low-density trees. Socially unjust REDD+ interventions are likely to suffer reductions in community engagement, undermining the long-term viability of an emissions reduction program and potentially leading to social unrest. For more information on indicators for REDD+ implementation see chapter 4.

Principle 4. Indicators, and the monitoring and management systems to which they are linked, should be designed through a participatory process that involves the key stakeholders who are responsible for, or influenced by the system attributes that sustainability indicators are trying to represent

In all the debate about the purpose and practicalities of sustainability indicators and monitoring it is easy to forget the importance of people in every part of the equation. It is people who design and implement indicators and monitoring activities, people who draw conclusions from sample data, people who decide which findings will be listened to and incorporated into new policies and management approaches on-the-ground and which will be discarded, and people who decide whether efforts to measure changes in a particular indicator are ultimately worthwhile and should be sustained into the long-term.

Participatory approaches to monitoring sustainability indicators are particularly relevant in developing countries, where meaningful engagement in the design and execution of monitoring programs by local people can empower them to take responsibility for the management of their own resources. It can also encourage a culture of learning which is key to the success of adaptive management (Cundill and Fabricius, 2009). In turn this empowerment and enhanced proximity between monitoring and management activities can often lead to more rapid and effective decision-making (Danielsen et al., 2007). A successful example of participatory indicator development and monitoring is Tebtebba, a global indigenous organization that works to incorporate indigenous rights, social justice and environmental sustainability into international processes. Their work on indicators is particularly impressive as diverse groups of indigenous peoples identify what they value and this is later aggregated at a higher scale (Chavez and Tauli-Corpuz, 2008).

2.4 Sustainability Indicators As Interdependent Components Of The Social-Ecological Systems They Are Designed To Measure

No sustainability indicator exists independently of the social-ecological system it is designed to assess. As a consequence, differences in context in which an indicator is assessed can result in markedly different outcomes (Meadows, 1998). For example, a recycling initiative in Poland based on leaflets and TV commercials directed at individual households had relatively little positive effect on individual behavior and recycling levels remained low. In an effort to increase recycling, incentives were introduced at the level of entire condominiums. Under this new monitoring regime the entire community was rewarded if everybody recycled, and through increased visibility among members of the community and concerns that individuals who failed to recycle would be socially ostracized, the recycling rates increased.

Over-emphasis on achieving changes in the value of a particular indicator or set of indicators can detract attention away from the underlying societal values it is intended to measure. As such, efforts to optimize management and decision-making processes to improve the value of an indicator can result in the indicator becoming the target in its own right. Further, because indicators can only ever provide a simplified and limited representation of the complex and dynamic social and environmental conditions they are designed to measure, placing excessive emphasis on changing the values of an indicator can lead to important changes being masked and allowing potentially perverse and unintended outcomes to emerge. As discussed above there are many reasons why indicators, for reasons of practicality are often highly simplified proxies of the attributes that managers and policy makers are really trying to change. An example discussed elsewhere in this book (chapter 8) is in the case of ecological restoration. The past few years have seen a marked increase in recognition of the need to restore large areas of native habitat if ecosystem services critical to human wellbeing – including hydrological, pollination and pest control services – are to be maintained and restored. The Aichi Target 15 of the United Nations Convention on Biological Diversity together with the ‘Bonn Challenge’, a global initiative on restoration, have established the goal to restore 150 million hectares of degraded and deforested land globally by 2020 (Menz et al., 2013). However, a target based only on hectares is not sufficient to indicate the successful restoration of the ecosystem services that underpin political support for this action – instead it is also necessary to take account of the condition of regenerating and planted forests, including aspects of their structural, compositional and functional diversity (see chapter 8) (Van Bellen, 2002; McGlade, 2009).

If employed for long enough, indicators can ultimately change underlying perceptions of values, becoming valued attributes in their own right often without a clear understanding of why changes in their value are more or less desirable for society. As eloquently stated by Meadows (1998), although indicators are formulated to measure what we value, in practice the opposite also often happens – we come to value what we measure. One of the most famous and instructive examples of this phenomenon can be found in the case of Gross Domestic Product (GDP) and how its use has resulted in entire economies being continuously shaped and reshaped to increase their value, often at the expense of critically important economic, social and environmental attributes (Daly, 2005).

2.4.1 Indicators And The Changing Nature Of Value: The Case Of Gross Domestic Product (GDP)

As discussed earlier in this chapter, an indicator is not inherently good or bad: its influence on how we manage social-ecological systems depends on the context in which it is measured and interpreted as well as the underlying objectives. Problems emerge when an indicator is interpreted as providing more meaning than was originally intended or than it is capable of providing. For example, a country's development standing is perceived principally not through indicators of equitable distribution of wealth, high literacy or low infant mortality rates, but rather on account of a high GDP. Marked changes in GDP readily make headline news of national newspapers around the world and can have a significant effect on determining the success or failure of a political party at election times. Many national and international (e.g. European Union) government policies are explicitly designed to achieve increases in GDP (Costanza et al., 2014).

GDP is designed to measure economic productivity and can be simply defined as the total monetary value of everything that has been produced in a given period (Stiglitz et al., 2010; Veiga, 2010). It is this intuitive and simple meaning that underpins its attraction to policy makers the world over. On the one hand GDP itself is not a useless measure – understanding the total value of a countries´ economic output is extremely useful as it is a proxy for a host of factors that most societies care about – such as the amount of tax revenues that are being generated. However, there are fundamental limitations to its application and use that continue to remain poorly appreciated despite its growing notoriety.

Whilst on the one hand GDP calculations are supported by detailed economic data and transparent methods (Tayra and Ribeiro, 2006; Bauler et al., 2009; Moldan and Dahl, 2009), the inclusion of certain economic activities and goods into its calculations is subject to country-dependent choices, and decisions about what should be valued. As such, critiques of GDP have a ready list of undesirable economic activities that do or could have a marked influence on levels of GDP. For example, a recent study pointed out the United Kingdom's GDP would be up to 5 percent higher if activities such as illegal prostitution and drugs were counted (Pilling, 2014). Another example is where the cost of crime protection is taken into account in GDP, which therefore rises when levels of violence increase across the country (Van Bellen, 2002; Tayra and Ribeito, 2006; Costanza et al., 2014).

In addition researchers and policy makers have long recognized that a single indicator such as GDP is inherently incapable of capturing the myriad attributes that contribute towards the sustainability of a country's development, and is blind to many negative social and environmental changes (Bauler et al., 2009). In particular, the focus of GDP on consumption rates means that it is naive to the many environmental and social benefits that contribute towards human wellbeing yet are rarely, if ever, quantified in monetary terms (Tayra and Ribeito, 2006; Stiglitz et al., 2010), and that this naivety underpins the increasingly widespread assumption that the current pattern of economic growth is unsustainable (Dasgupta and Ehrlich, 2013).

2.4.2 Towards Alternative Measures Of Sustainable Human Prosperity

There is a large number of efforts to complement or entirely replace GDP with indicators that can offer a more nuanced and transparent barometer of progress towards, or away from sustainable development. Set against these efforts is the fact that the very concept of sustainable development is, to the persistent frustration of many sustainability practitioners and decision makers, hard to reach a consensus on, much less measure progress towards. The most enduring definition can be traced to the Brundtland Report (WCED, 1987) where sustainable development is defined as human development that ''meets the needs and aspirations of the present without compromising the ability of future generations to meet their own needs.'' Central to the common interpretation of this concept is the need to reconcile environmental, social and economic demands, and recognition that the existence of biophysical limits or boundaries, together with imperatives of social equity, justice and wellbeing that cannot be breached if economic growth is to be considered sustainable (e.g. Ehrlich and Ehrlich, 2013).

Many alternative indicators of human prosperity to GDP are biased towards only one or two of the three "pillars" (social, environmental, economic) of sustainable development. These can be divided in three groups, of which the first reflects social and environmental factors; the second includes subjective measures of wellbeing; and the third includes dimensions of wellbeing related to housing, leisure, life expectancy, among others (Costanza et al., 2014).

The Human Development Index is one of the most widely recognized measures of development that considers many of the same variables included in GDP but also includes measures of human longevity and education (Wilson et al., 2007; Guimarães and Feichas, 2009). By contrast, the Ecological Footprint index is focused on measuring environmental impacts, and the sustainability of human consumption patterns by calculating the amount of resources that are required to support consumption (and the production of waste) (Wilson et al., 2007; Stiglitz et al., 2010). The Genuine Progress Index (GPI) tries to go one step further by considering not only economic (spending, consumption), but also social and environmental criteria (aspects of wellbeing, deduction of environmental factors such as pollution from consumption, and loss of natural resources, among others) (Stiglitz et al., 2010). Despite its more inclusive approach GPI remains problematic in a number of key aspects, with the valuation of externalities considered to be too speculative (Veiga, 2010), and many of the criteria being difficult to collect data for (Guimarães and Feichas, 2009).

While all of these measures seek to capture some aspect of human prosperity the fact that they emphasize different aspects, and entirely exclude others, often leads to contrasting trends, confounding interpretation and leading to conflicting guidance for policy makers (Veiga, 2010). Disparities are clear when comparing patterns of GPI and GDP over time, revealing that GDP can rise at the same time that GPI is falling, revealing ongoing environmental degradation (Guimarães and Feichas, 2009) and widening social inequality (Tayra and Ribeiro, 2006).

As most of the alternative indicators to GDP are biased toward one or two pillars of sustainability and not on the links between them (Moldan and Dahl, 2009) they continue to fall short of providing a clear and robust picture of changes in the overall system. Appreciation of the need to draw on a suite of indicators rather than one single "magical index" (see section 2 above) underpin the significant efforts that are being invested in the crafting of the United Nations Sustainable Development Goals (SDGs) in the aftermath of the Rio+20 conference and to drive the post 2015 international development agenda (see Introduction to this book). The SDGs are designed to be "action-orientated, concise and easy to communicate" whilst also being comprehensive enough to address all major sustainability concerns.

2.5 Conclusions

A systems perspective can be very helpful in designing indicators and monitoring programs. We propose four principles in designing and monitoring sustainability indicators for social-ecological systems: i) indicators are integral parts of a wider monitoring and management system, ii) indicators should be designed and used in combination with a suite of other indicators as a coherent part of a wider monitoring system, iii) it is essential to understand how different indicators relate to the wider study system, iv) indicators, and the monitoring and management systems to which they are linked should be designed through a participatory process that involves the key stakeholders who are responsible for, or influenced by the system attributes that sustainability indicators are trying to represent. Indicators will always remain a subjective reflection of the varying goals and values that different societies and groups of stakeholders hold. Indicators will also inevitably become part of the complex system themselves, and as such changes in their values may result in changes to societies' values. There is an urgent need for systems thinking to become much more mainstreamed in both research and policy spheres if the indicator and monitoring systems we depend upon are to provide a reliable barometer of progress towards a more sustainable world.

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3 Biodiversity Indicators And Monitoring For Ecological Management

Toby Gardner

3.1 Introduction

The fate of much of the world's terrestrial biodiversity depends upon our ability to improve the management of ecosystems that have already been, or are currently being, modified by humans (Gardner et al., 2009; Wright, 2010; Pereira et al., 2012; Malhi et al., 2014). Monitoring, as a means of detecting the changing state of an ecosystem, and identifying ways in which existing management approaches can be made more sustainable, is a central part of any strategy to safeguard biodiversity in the long-term.

This chapter presents a broad overview of some of the key features of any process to monitor and evaluate biodiversity. Selection of appropriate indicators are a central part of this, yet as is the case for the assessment of any indicator, good biodiversity indicators represent only a necessary, yet not sufficient condition for a monitoring process to provide the kind of support necessary to foster improvements in sustainability.

The chapter briefly identifies ways in which biodiversity monitoring can be most effective in facilitating and guiding any management process – whether it is the management of a protected area, a multiple-use reserve, managed forest or wetland or urban park. A lot of existing texts on biodiversity indicators and monitoring focus primarily on the technical details of *how* to survey biodiversity in the field. Here I take a few steps back and focus on the importance of first thinking about the *why* and *what* of monitoring, as well as the ways in which monitoring activities fit within a wider framework of the management system itself – whatever that management system may be. Following this, I present an overview of different types of indicators that can be used to support a biodiversity monitoring program, including different ways to assess the status and trends of biodiversity.

The chapter is based heavily on the work of Gardner (2010a), which provides a more comprehensive overview of the status of biodiversity monitoring (with a focus on forest ecosystems) and presents a detailed operational framework of ways in which the process of collecting biodiversity data and indicators can make a more effective and meaningful contribution to the way in which we manage and conserve our natural heritage for future generations.

3.2 The Context And Purpose Of Biodiversity Monitoring

3.2.1 Why Should We Be Worried About Biodiversity Monitoring In The First Place?

The straightforward answer to this question is that monitoring is generally done badly yet remains the only way by which we can assess the state of our ecosystem and improve on our ability to conserve biodiversity into the long-term.

Despite its theoretical importance, monitoring is often trivialised as being a simple "tick the box" exercise, necessary to satisfy auditing requirements. The focus is often only on which indicators to choose and how they should be sampled and recorded. Yet poorly conceived monitoring programs can often do more harm than good – resulting in a waste of precious resources, and an undermining of the credibility and value of monitoring in the eyes of management authorities and decision makers (Sheil et al., 2004; Lindenmayer and Likens, 2010).

A large number, if not the majority, of existing biodiversity monitoring programs are centred on providing a "surveillance style" record of how biodiversity (e.g. the population size of a particular species or the area of a specific vegetation type) changes over time. Such information is often used as a form of early-warning system. For example, information on population or species declines can be used to kick start conservation action, both in the form of a regulatory mechanism (e.g. as is done commonly in the management of fish stocks) and as a way of raising public and political awareness about environmental issues. Long-term monitoring of biodiversity across a network of sites can also help in developing an improved understanding of background levels of variability in natural systems, as well as capture information on hitherto unperceived threats (e.g. the impacts of climate change and disease on amphibians; Pounds et al., 2006). Surveillance style monitoring can also be an effective way to engage non-scientists in conservation. Good examples of this are the long-term, nation-wide bird surveys ran in Britain and North America that involve thousands of volunteers while also feeding information into national indicators of biodiversity loss (e.g. as developed by the British Trust for Ornithology on behalf of the UK government http://www.bto.org/research/ indicators/index.htm).

Nevertheless, there are serious limits to a surveillance approach as a practical aid to ecosystem management. The main shortcoming is that *it is disconnected from the management process*. By this I mean that the design of the monitoring program has an isolated focus on the biodiversity of interest (e.g. are there more or less individuals of an endangered species in the management area?) and not on assessing the impact of the ongoing management activities themselves (e.g. importance of variability in logging cycles, road building or the design of nature corridors for effectively conserving the biodiversity of interest). Surveillance type approaches presume that a clear and workable plan of action is already available and that this can be launched into place once the warning bells start ringing. Unfortunately this is rarely the case.

3.2.2 A Simple Framework For Biodiversity Monitoring As A Practical Aid To Ecosystem Management

In an ideal world we would have a perfect understanding of how different human activities and management interventions impact biodiversity, and we could use this understanding to dictate a clear code of practice (e.g. legal or certification standards) that guarantees responsible and more sustainable use. Assessments of management compliance could be made simply by monitoring the implementation of previously agreed management activities. This is often termed *implementation monitoring* (Noss and Cooperrider, 1994; Gardner, 2010a). However, this is clearly not the case. The biodiversity consequences of human activities are unpredictable, many threatening processes remain poorly understood, and in the vast majority of cases we have a poor understanding of how generic guidelines can be most effectively adapted to fit the context of particular site. To overcome this problem biodiversity monitoring is needed to satisfy two interrelated purposes that are central to any management process (Fig. 1):

- To ensure that recommended management practices do indeed translate into minimum levels of performance and biodiversity conservation on the ground. This is often termed *effectiveness monitoring*. Effectiveness monitoring should represent an integral part of any auditing or compliance process, e.g. for a certification standard or environmental regulatory framework.
- To evaluate the extent to which existing management standards are adequate and how they can be further refined to ensure continued progress towards long-term conservation goals. This is often termed *validation monitoring*. This is essentially the same as applied research and provides a valuable mechanism for learning about how to improve opportunities for biodiversity conservation within any management process

Done well, monitoring should provide a linchpin between ultimate management goals and the ongoing management process - the guiding hand by which conservation objectives can be translated into improved on-the-ground management. To achieve this broad purpose monitoring serves two specific and inter-related functions.

First, it is an assessment tool that is used to assess the status or condition of biodiversity in a managed system (wherever conservation management strategies may already exist). In so doing it can provide an assessment of management performance and compliance against pre-determined standards. Without reliable information on the status and trends of the ecosystem it is impossible to expect that managers can ensure the conservation of viable populations of native species and the maintenance of key ecological processes (Noss, 1999).

Second, it is an evaluation tool that is used to compare the effectiveness of alternative management actions, whether existing or potential - thereby providing a means to both validate the adequacy of existing management approaches and identify

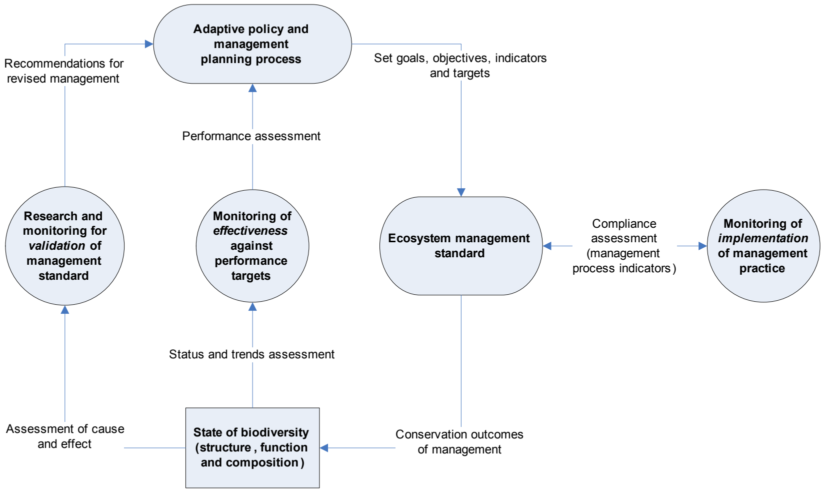

Figure 1: A conceptual framework of an integrated biodiversity monitoring program for adaptive ecosystem management. To be effective in both assessing and evaluating performance a monitoring program should comprise three tiers: implementation monitoring of management practice compliance, effectiveness monitoring of the system state against predetermined performance indicator values, and validation monitoring to evaluate how best to achieve continued progress towards long-term conservation goals. Adapted from Gardner (2010a).

directions for future progress towards more sustainable systems of use. In essence this is the philosophy of adaptive management, which although much discussed (and required – at least on paper - by many management authorities) has rarely been implemented effectively on the ground. Key to successfully integrating biodiversity monitoring within the wider ecosystem management process is recognition of the complementary role played by these different types of monitoring approaches and their associated indicators (Fig. 1).

Once the purpose of a monitoring activity has been established (e.g. effectiveness or validation monitoring or both) there is a logical series of steps in developing the rest of the program (Fig. 2; Green et al,. 2005; Gardner, 2010b). If the only purpose is to provide an audit function then the task is relatively straightforward – indicators and minimum standards are determined by the relevant authority and monitoring data are collected to ensure that standards have been met (Fig. 3). By contrast, validation monitoring is a much more involved process that requires measuring changes across different levels of cause and effect, from changes in management practices (ultimate drivers), through changes in ecosystem structure and function (proximate drivers), to changes in biodiversity - with the end goal of generating recommendations for how to improve management (Fig. 4).

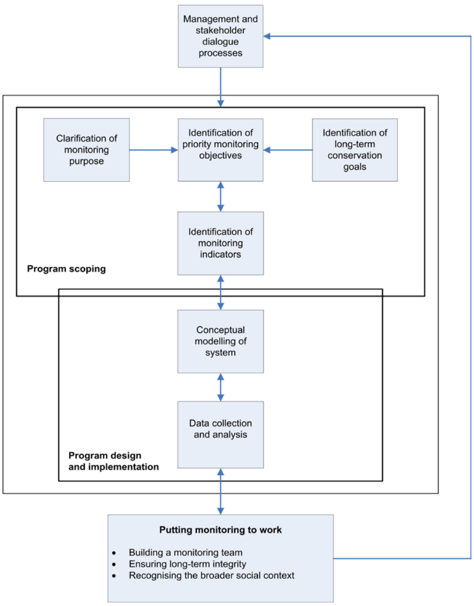

Figure 2: Overview of a biodiversity monitoring program, comprising of inter-related scoping, design and implementation stages. Although some choices are inevitably made before others the process of developing and implementing a biodiversity monitoring program should not be thought of as strictly linear. Instead the development of different stages often requires joint consideration (e.g. the process of selecting objectives and indicators) and a flexible approach is needed to accommodate feedbacks arising from constraints associated with indicator selection, data collection and analysis. Redrawn from Gardner (2010a).

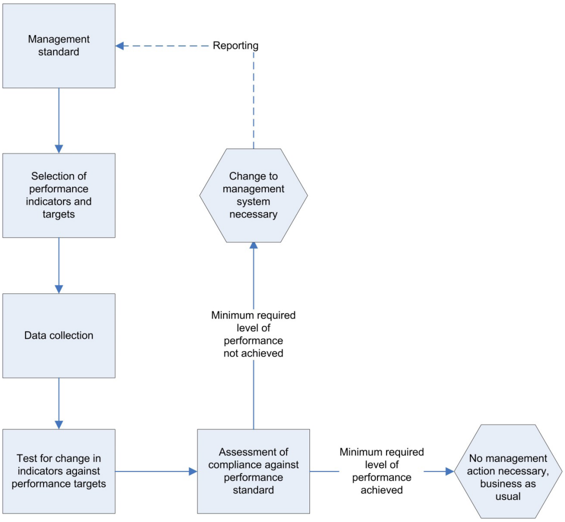

Figure 3: The process of effectiveness monitoring to assess compliance against minimum performance standards in forest management. Redrawn from Gardner (2010a).

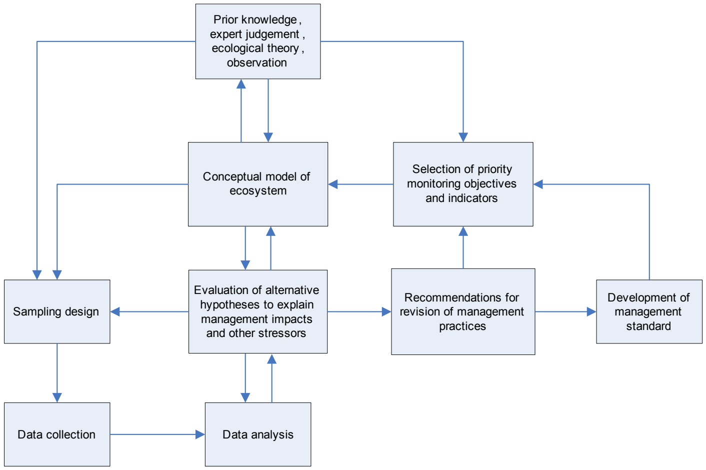

Figure 4: A conceptual framework of a validation monitoring program as a research exercise designed to evaluate the adequacy of existing management standards and make recommendations for continued improvement in achieving progress towards long-term conservation goals. Adapted from Gardner (2010a).

Biodiversity conservation goals provide a reflection of societal values and political or institutional intent in management, and create the entire context and sense of purpose of biodiversity monitoring, as well as the basis for selecting individual monitoring objectives and indicators. Conservation goals can be focussed on safeguarding individual species of conservation concern (e.g. threatened species or species of particular functional importance in ecosystems such as key seed dispersers and pollinators), or may adopt a broader, ecosystem-wide perspective to ensuring the protection (or restoration) of ecological integrity across whole management areas or landscapes (as determined by deviations from an appropriate reference condition such as a neighbouring reserve or set-aside area). Both goals are complementary (Lindenmayer et al., 2007; Gardner, 2010a) yet the maintenance and restoration of ecological integrity invokes a much broader conservation challenge than that which is focussed on preserving a particular set of species. Assessments of ecological condition or integrity are very well developed for aquatic systems (see Linke et al., 2007), yet despite offering much promise have received comparatively little attention in the terrestrial world.

Clear objectives are essential to ensuring that the time, money and expertise are not wasted in any monitoring program. Because no monitoring program has sufficient resources to address all possible objectives it is necessary to prioritise investment so as to deliver the greatest benefits with respect to long-term conservation goals. This includes identifying which areas of management have the greatest impact on the biodiversity of concern, where the greatest areas of scientific uncertainty lie, and what is possible with the funds and human resources available.

3.3 Indicators For Biodiversity Monitoring

In discussing environmental indicators in general Hammond (1995, pp:1) define an indicator as "something that provides a clue to a matter of larger significance or makes perceptible a trend or phenomenon that is not immediately detectable. [. . .] Thus an indicator's significance extends beyond what is actually measured to a larger phenomena of interest''. Indicators provide the practical tool by which changes in management practices, system attributes and ecological processes can be measured, and minimum performance standards or thresholds evaluated. Consequently the choice of indicator is fundamental in defining the approach to both monitoring and management (Lindenmayer and Burgmann, 2005). It is important to clearly distinguish the concept of an indicator from that of an attribute: attributes are all encompassing and relate to any quantifiable element of concern, whereas indicators only represent the subset of attributes or attribute features that are used as surrogates of other valued attributes (see Chapter 8 for examples related to forest restoration).

To assist in relieving the significant confusion that surrounds the indicator concept, Table 1 provides a typology of indicator types that helps reveal problems of synonymity and identifies the role of different indicators within a simple hierarchical pressure-state-response framework. In general terms – whether for biodiversity or any other kind of sustainability indicator - it is useful to consider two main types of indicator; those whose function is prescriptive, and those whose function is evaluative (Kneeshaw et al., 2000; Rempel et al., 2004 - although these concepts can be confounded depending on the perspective of the observer, see Table 1). Management policy and process indicators are both prescriptive in that they are used to measure or verify the existence or implementation of certain policies and management strategies. Management practice indicators are termed such because they are used to measure the implementation of management practices. In a similar sense they are also referred to as "driver" or "pressure" indicators (e.g. Hagan and Whitman, 2006), as well as indicators of management response to some earlier observation or warning sign. In contrast, performance indicators are evaluative in the literal sense that they are used to evaluate changes in management performance. I distinguish the concepts of indirect versus direct performance indicators, and recognise that both are commonly used to measure management performance (e.g. for certification standards, FSC, 2002; and see Newton and Kapos, 2002). Because of the difficulties in reliably linking management impacts to changes in the distribution and abundance of actual species, direct species-based performance indicators are often more useful as evaluators of performance standards rather than direct measures of compliance.

3.3.1 Management Practice Indicators

Management practice indicators directly measure management practices, including local and landscape level interventions. Together they detail which aspects of a given ecosystem are managed (i.e. as distinguished from un-managed activities and external threats). The assessment of management practice indicators forms the basis of an implementation monitoring program that is used to evaluate compliance against an agreed management standard.

3.3.2 Management Performance Indicators

When the consequences of management are uncertain (a condition that is almost always true) performance-based indicators provide a more transparent and comparable assessment of management accountability than is possible with process-based indicators.

Ecosystem managers very rarely have the capacity or expertise necessary to directly manage populations and species. Instead they manipulate structural and functional aspects of the ecosystem, which in turn have a variety of consequences for biodiversity (Lindenmayer and Franklin, 2002; Lindenmayer and Fischer, 2006; Gardner et al., 2009). To accommodate this fact ecosystem managers address the problem of biodiversity conservation through a combination of coarse and fine filter management approaches (Hunter, 1990).

Coarse filter management is focused on creating or maintaining the ecosystem structures and ecological processes required for the persistence of a wide range of species. By contrast, species specific or fine-filter management addresses the direct resource or habitat needs for particular target species that are not addressed through coarse filter approaches. The extent to which coarse filter approaches ensure adequate resource provision for a large number of species is poorly understood for much of the world, and requires testing through validation monitoring (Table 1). Coarse and fine-filter management approaches can be evaluated through monitoring programs using a combination of what I term here indirect and direct performance indicators.

Indirect performance indicators provide the foundation for a performance-based standard and are intended to represent the proximate drivers of any observed changes in biodiversity (Table 1). For example in a managed forest system indirect performance indicators are commonly made up of stand and landscape-level indicators of forest structure (sensu Lindenmayer et al., 2000) which provide a readily accessible and quantifiable indication of ecologically relevant forest management impacts. Some indicators of ecological processes such as major disturbance regimes (e.g. fire, flood, pest-outbreaks), and changes in soil or water contaminant loads, can also represent valuable indirect indicators of performance (although at the same time they also provide direct measures of performance if the management goal is concerned

Table 1: A typology of indicators for biodiversity monitoring to support ecosystem management. Adapted from Gardner (2010a) and compiled with reference to Stork et al., (1997), McGeoch (1998), Caro and O'Doherty (1999), Kneeshaw et al., (2000), Lindenmayer et al., (2000), and Rempel et al., (2004).

Indicator type	Synonyms	Description	Examples	Key advantages	Key disadvantages
Policy indicator	*Planning indicator*	To assess the existence of an adequate institutional framework to support planning, research and development, and reporting. Can be relevant to both local and landscape-scale management issues.	– Existence of clearly defined management objectives – Existence of a comprehensive management plan – Availability of trained individuals for monitoring purposes	– Fundamentally important for the reliable implementation of best-practice guidelines and adaptive management – Easy and cheap to assess – Fully comparable across different sites	– Provides no assessment of actual management practices or performance.
Management practice indicator	*Management process indicator, pressure indicator, control indicator*	To assess the extent of intervention of an explicit management *practice* that forms part of an overall management strategy. It is *prescriptive* in that it defines the type and level of *treatment* or *control* applied by a management intervention. The use of appropriate process indicators reflects a proactive approach to promoting responsible management. Process indicators are used as the basis of systems-based standards for ecosystem management.	– Logging intensity and length of logging rotation cycle – Restoration planting strategy – Techniques to prevent soil erosion – Reserve allocation and design	– Easy to define – Available for immediate implementation based on knowledge of "best-practice" management – Provides the most suitable proxy for the assessment and early detection of changes to ecological resources – Cheap to monitor – Readily transferable among different sites – Requires no a-priori ecological knowledge	– Only represents the application of a minimum or best practice standard and doesn't measure management performance or evaluate the effectiveness of alternative strategies.

Continued **Table 1:** A typology of indicators for biodiversity monitoring to support ecosystem management. Adapted from Gardner (2010a) and compiled with reference to Stork et al., (1997), McGeoch (1998), Caro and O'Doherty (1999), Kneeshaw et al., (2000), Lindenmayer et al., (2000), and Rempel et al., (2004).

Indicator type	Synonyms	Description	Examples	Key advantages	Key disadvantages
Performance indicator	*Condition indicator, health indicator, response indicator, environmental indicator, monitoring indicator, outcome based indicator*	To assess the *performance* of ecosystem management toward meeting long-term conservation goals. In general they are considered to be *evaluative* in that they assess the *response* of a system attribute to a particular management practice. Biodiversity performance indicators can provide both indirect (i.e. measures of ecosystem structure and function) and direct (i.e. measures of biodiversity) assessments of the target criteria and depending on the perspective of the stakeholders they can be considered as end-points in themselves (e.g. target species). The use of appropriate performance indicators in the evaluative sense reflects a reactive approach to promoting responsible ecosystem management. Performance indicators are used as the basis of performance-based standards for ecosystem management. Individual target species can sometimes provide a complementary indicator function (e.g. keystone species) or may just be worthy of monitoring in their own right (e.g. endangered species and pest species).			
– Indirect performance indicator	*State indicator, eco-physical indicator*	Indirect assessment of ecological condition through the measurement of structural (landscape and habitat) and functional ecosystem attributes. Reflects a coarse-filter approach to assessing progress towards sustainability based upon implicit hypotheses of the relationship between the indicators and valued conservation attributes. Can be either evaluative or prescriptive depending on the context	– Habitat structural complexity – Area of native vegetation cover – Habitat fragmentation – Disturbance regimes (e.g. fire and grazing) – Soil contaminant loads	– Transparently linked to management actions – Directly evaluates ecosystem heterogeneity and structural diversity – Relatively cheap and simple to monitor	Assumes an adequate understanding of the relationship between structural/functional ecosystem attributes and the persistence of biodiversity

Continued **Table 1:** A typology of indicators for biodiversity monitoring to support ecosystem management. Adapted from Gardner (2010a) and compiled with reference to Stork et al., (1997), McGeoch (1998), Caro and O'Doherty (1999), Kneeshaw et al., (2000), Lindenmayer et al., (2000), and Rempel et al., (2004).

Indicator type	Synonyms	Description	Examples	Key advantages	Key disadvantages
– Direct performance indicator	*Biological indicators, bioindicators, response indicator, taxon-based indicator, predictor taxa*	Direct measurement of attributes of interest as defined by a particular conservation goal. Varyingly operate as surrogates of other attributes of the target criterion (e.g. other species, or a measure of ecological integrity). Reflects a fine-filter approach in assessing progress towards responsible management. Direct performance indicators can be thought of as purely evaluative.	– Environmental indicators – Cross-taxon response indicators – Ecological disturbance indicators – Target species	– Provides a direct measurement of change in actual conservation values	– Not always clear how they are linked to management activities – Often captures only a subset of a much broader conservation goal

with soil contamination, and not biodiversity). The key requirement of any indirect performance indicator, whether structural or process-based, is that any changes in biodiversity can be clearly linked to actual management impacts through a logical chain of cause and effect.

Direct performance indicators differ from indirect performance indicators because they directly measure changes in the valued attributes rather than their perceived ecological requirements (e.g. native species rather than habitat availability). For biodiversity monitoring programs, direct performance indicators can be considered as synonymous with biological indicators (and their myriad forms) and target species (Table 1 and see below). A comprehensive ecological monitoring program may also encompass indicators of ecological processes that play important roles in the maintenance of biodiversity (e.g. leaf litter decomposition, Ghazoul and Hellier, 2000; or soil properties, Curran et al., 2005), yet whose links to management activities are poorly understood.

3.3.3 Biological Indicators

Chosen carefully, biological indicators can make an invaluable contribution to monitoring because they are the only method of synthesising the overwhelming complexity of ecological systems, and are therefore the most effective tool for linking conservation science to policy (UNEP, 2000). They also provide the highest level of information quality because biological indicators are valued attributes in their own right. Financial and logistical constraints mean that it is impossible to measure all elements of biodiversity (Lawton et al., 1998; Gardner et al., 2008), and biological indicators that can operate as surrogates for changes in ecosystem health or condition or the distribution and status of other species provide a practical solution to an otherwise intractable problem (Margules et al., 2002, Niemi and McDonald, 2004).

As surrogates of change in the condition and/or diversity of ecosystems, biological indicators can be used in a regulatory sense to provide an early warning signal of impending environmental change, as well as in a diagnostic sense, as an aid to interpreting the ecological consequences of alternative management strategies (Dale and Beyeler, 2001; Niemi and McDonald, 2004). The two fundamental requirements for all biological indicators are; (i) that they reflect something that cannot be measured directly, while also providing more information than that which relates only to the indicators themselves, and (ii) their measurement is logistically and financially feasible. Beyond this, the concept of an indicator species or species group can adopt a myriad of different meanings (Caro and O'Doherty, 1999; Lindenmayer et al., 2000). Although the semantics of biological indicators has a highly confused history in the ecological literature (Caro and O'Doherty, 1999; Caro 2010), I follow McGeoch (1998, 2007) in recognising three broad and overlapping categories of biological indicator that each correspond to conceptually different applications, namely; environmental indicators, biodiversity indicators and ecological indicators.

3.3.3.1 Environmental Indicators

Environmental indicators are species, or groups of species that provide a predictable and quantifiable measure of an environmental state or impact on some abiotic or physical parameter of interest that may be difficult or expensive to measure directly. They have most commonly been applied to indicate levels of pollutants and toxins in water, but also other measures such as soil fertility (McGeoch, 1998, 2007). Related terms include bioassays, accumulator species and biomarkers.

3.3.3.2 Biodiversity Indicators

Biodiversity indicators operate, as the name suggests, as surrogates of biodiversity, and are described as species or groups of species whose distribution or level of diversity reflects some measure of diversity of other taxa (i.e. their distribution is highly congruent with the distribution of other, unrelated species) (Noss, 1990; McGeoch, 1998, 2007). Despite often having weak theoretical and empirical support (Lindenmayer et al., 2000, 2002) this concept has dominated much of the discussion surrounding biological indicators. The term biodiversity indicator is commonly employed in reference to large-scale conservation planning assessments where understanding spatial patterns of species congruency is central to developing an effective network of protected areas. However, the concept also encompasses what I term here “cross-taxon disturbance response indicators” (and see Caro, 2010), which are applicable to monitoring systems and relate to those individual species or species groups that are used to capture the impacts of disturbance on other species or species groups (e.g. Barlow et al., 2007; Gardner et al., 2008).

3.3.3.3 Ecological Indicators

Ecological indicators are species that demonstrate the effect of environmental change and degradation on biota or biotic systems (Kremen, 1992; McGeoch, 1998, 2007; Howe et al., 2007). As some researchers have employed a more general usage for ecological indicators (e.g. Noss, 1999; Niemi and McDonald, 2004), I use the more specific term of “ecological disturbance indicators” with respect to their application for evaluating the ecological consequences of human disturbance in modified systems (see also Caro, 2010).

Ecological indicators may or may not capture the specific responses of other species to disturbance, but their primary utility is in providing a species-based gauge of the otherwise difficult to quantify holistic concepts of ecological condition and integrity, where measurements are made as some form of deviation from a reference or minimally disturbed state. The implication of a loss of integrity as signalled by a decline in ecological disturbance indicators is derived from what we know about what such species do. Indicator species groups that are both sensitive to environmental change, and are known to perform important ecological functions make excellent ecological disturbance indicators as they provide the most reliable inferences about the ecological and functional implications of disturbance.

Because a primary goal of biodiversity conservation is to improve our understanding of the consequences of anthropogenic-induced stressors on native biota, ecological disturbance indicators as defined here represent arguably the most critical objective in the field of biological indicators (McGeoch, 1998, 2007; Pearce and Venier, 2005). An important way in which ecological disturbance indicators are quite distinct from environmental indicators is that they assess the effects or ecological consequences of environmental change (the indicator itself is of intrinsic interest), whereas environmental indicators are used more simply as a gauge of change in a particular abiotic environmental parameter (McGeoch, 1998). Because ecological disturbance indicators indicate functional changes to an ecological system (as measured by a deviation from a reference condition), they reveal insights into the consequences of ecosystem management that cannot be gained from direct measurement.

3.3.3.4 Focal Species

In addition to the indicator concepts as described above, there are additional types of indicator that operate as partial surrogates for biodiversity yet have been defined to have a more specific usage and fall under the general category of "focal" species (e.g. Lambeck, 1997; Caro and O'Doherty, 1999). Focal species have been developed largely in response to criticisms of the biodiversity indicator concept as based on patterns of species congruency, and are characterised as species that have particular ecological requirements, the protection of which can help ensure the conservation of other species, encompassing the concepts of umbrella species, keystone species, and resource or process-limited species (Mills et al., 1993; Lambeck, 1997; Noss, 1999). Under a framework developed by Lambeck (1997, see also Noss, 1999) focal species are used to identify specific threats, and the species most sensitive to each threat are then used to define the minimum acceptable level at which that threat can occur. Consequently, they encompass elements of both the biodiversity and ecological indicator species concept, yet as noted by Niemi and McDonald (2004), focal species tend to differ from ecological disturbance indicator species because they do not necessarily serve to measure ecological condition, nor do they convey a clear stress-response relationship.

3.3.3.5 Target Species Of Particular Conservation And Management Concern

There are a number of individual species which, for a variety of reasons, may deserve to be monitored for their intrinsic interest, and not because they necessarily indicate patterns of any other species or ecological processes, or condition. Here I term all such species “target species”. Many legal and voluntary ecosystem management guidelines give particular emphasis to the role of target species of conservation concern (i.e. endemic, threatened and endangered species) as a focus for biodiversity monitoring and evaluation programs. Identification of threatened and endangered species may be made on the basis of regional, national and global listings, each of which involves a distinct set of selection criteria. However, the globally accepted standard for classifying extinction risk and identifying threatened species is the IUCN Red List (Mace et al., 2008). In addition, other species that are deserving of particular management attention include invasive and pest species that may have significant impacts on local biodiversity, as well as species that are of particular economic or cultural importance for local people (e.g. non-timber forest products such as palms, nuts and game meat, Godoy and Bawa, 1993) or “flagship species” that are used to motivate conservation action, Caro and O’Doherty, 1999).

3.4 Putting Biodiversity Monitoring Into Practice

Ultimately theoretical arguments concerning the purpose, design and implementation of monitoring programs can only go so far towards ensuring success. Many biodiversity monitoring programs either fail or fall short of their original intentions because insufficient attention is given to the factors that determine viability in the real world - in particular the role of people in monitoring.

Deciding on the appropriate blend of people to be responsible for designing and running a biodiversity monitoring program depends on both the desired level of detail as well as who the data are intended to benefit. In many ecosystems an integrated approach to monitoring that combines expert guidance and management from professional scientists with a close involvement of local people (whether they be local management authorities or representatives of local communities) is likely to provide the most attractive solution. The contribution of professionals ensures scientific rigour in program design and data analysis, while the involvement of local people facilitates the process of implementing any management recommendations - providing a cost-effective and sustainable means of data collection as well as a potentially rich source of local knowledge to aid interpretation of results. In addition to efforts to improve cost-effectiveness, the viability of biodiversity monitoring can be further enhanced by increasing the relevance and utility of monitoring products to as wide an audience as possible, including relevant management authorities responsible for standard development, government agencies responsible for national biodiversity assessments, the scientific community and environmental educators.

Biodiversity monitoring and management should be viewed not as strictly scientific activities, but instead as inherently social processes that are influenced and guided by science. Without clear recognition of the broader societal context within which the monitoring process and the collection of information about the changing status of biodiversity and the environment, is situated, as well as the underlying conservation values that define the ultimate purpose of monitoring, even the most technically robust monitoring programs will be committed to failure. The challenge of putting biodiversity monitoring to work will ultimately depend, more than anything else, on human behaviour and our capacity to change. As John Meynard Keynes so astutely put it "*The difficulty lies not so much in developing new ideas as in escaping from old ones*".

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4 Monitoring REDD+ Impacts: Cross Scale Coordination And Interdisciplinary Integration

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4.1 Introduction

Results-based compensation for reducing emissions from deforestation and forest degradation and enhancing carbon stocks (REDD+) is one promising way to help mitigate global climate change. Since the climate impact from reduced emissions (and increased removals) is the centerpiece of REDD+, countries are asked to set up systems to monitor changes in forest carbon stocks for reporting at the international level (Herold and Skutsch, 2011; Romijn et al., 2013). Yet, REDD+ monitoring goes beyond carbon for at least three reasons. First, REDD+ activities can promote a host of social and environmental co-benefits or entail risks that should be considered in their design and implementation. Second, the United Nations Framework Convention on Climate Change (UNFCCC) Cancun Agreement articulates seven safeguards (Decision 1/CP.16) for REDD+ programs to: 1) complement national forest programs and international conventions and agreements; 2) maintain transparent governance; 3) respect knowledge and rights of indigenous people and local communities; 4) obtain effective participation in REDD+ design and implementation; 5) promote forest conservation and other environmental and social co-benefits; 6) address risks of reversals; and 7) reduce leakage (UNFCCC, 2011a). Countries must set up Safeguard Information Systems to be eligible for results-based payments (UNFCCC, 2014). Also, jurisdictions and projects engaged with multi- and bilateral donors and third-party certifiers may need to consider additional standards and/or guidance for demonstrating high social and environmental performance, such as those of the World Bank Forest Carbon Partnership Fund (FCPF, 2013), the UN-REDD Programme (UN-REDD, 2012), the Climate Community and Biodiversity Alliance (CCBA, 2013) and REDD+ Social and Environmental Standards Initiative (REDD+ SES, 2013). Third, forest monitoring is becoming an important national policy tool for countries to assess and understand drivers of forest change, underpin REDD+ and related climate-friendly land use strategies, track implementation, and form the basis for the distribution of benefits generated through climate finance (De Sy et al., 2012; Kissinger et al., 2012). The multidimensionality of REDD+ poses great challenges to identifying efficient trade-offs between in-depth, fully comprehensive monitoring and increasing complexity and costs, which is a serious problem given the limited funds available for REDD+ monitoring. Monitoring both the carbon and non-carbon impacts of REDD+ requires development of systems that are scientifically sound, yet simple enough to be implemented effectively (Gardner et al., 2012). Resolving this challenge is critical to operationalizing REDD+.

One of the primary challenges for REDD+ monitoring systems is the issue of scale. To date, most monitoring of REDD+ performance has occurred at the subnational level. Since the Bali Road Map of 2007, hundreds of subnational REDD+ initiatives have emerged throughout the tropics, which range from localized projects to broader jurisdictional REDD+ programs (Simonet et al., 2014, Sunderlin et al., 2014). Many of these initiatives include a combination of forest law enforcement and implementation of both conditional and non-conditional incentives to promote more sustainable land use practices (Sunderlin and Sills, 2012). While these initiatives conform to various third-party accounting and verification systems, many have struggled to implement sustained and effective monitoring (Joseph et al., 2013). This difficulty is partly due to limitations in capacity and resources, and because the role of subnational monitoring systems becomes less clear as national REDD+ systems develop. For instance, some subnational REDD+ programs are pilots of the Verified Carbon Standard (VCS) Jurisdictional and Nested REDD+ Framework for carbon accounting and crediting. These rules may eventually differ from those for accounting and reporting of national forest monitoring systems to the UNFCCC using Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidance (GPG). For non-carbon, several subnational REDD+ programs are part of the REDD+ SES Initiative for demonstrating high environmental and social performance, which may or may not dovetail with national Safeguard Information Systems. In addition to the issue of reporting across scales, the issue of scale of measurement is central to monitoring. Coarse- versus fine-scale monitoring of the carbon and non-carbon impacts of REDD+ may lead to different conclusions about its results-based performance, making it key to find the right balance between precision/accuracy and effort (Romijn et al., 2013). This issue surfaced in the reporting of Annex 1 countries for land use, land-use change and forestry (LULUCF) activities under the Kyoto protocol.

A second challenge is the disconnect between carbon and non-carbon monitoring efforts in REDD+, which are often pursued in disciplinary isolation. On the one hand, there are remote sensing and forest carbon scientists focused on improving systems and approaches for carbon monitoring through activity data (i.e. human activity resulting in emissions or removals), emission factors (i.e. emissions or removals of all greenhouse gases in all carbon pools), and assessing impact against robust reference emissions levels (i.e. counterfactual benchmark against which actual emissions and removals can be measured) (Herold et al., 2012; Verchot et al., 2012). On the other, there are social scientists, ecologists and advocates focused on minimizing social and environmental risks associated with REDD+ and enhancing benefits, with further subdivision into social and environmental camps. On the social side, the focus has been on protecting and enhancing local governance and wellbeing (Brown et al., 2008), along with securing local rights to land and resources (Sunderlin et al., 2009), which are often considered key to REDD+ effectiveness (e.g. secure tenure as a pre-requisite for application of regulatory and incentive-based REDD+ mechanisms; Duchelle et al., 2014). On the environmental side, the focus is on conserving the environmental services provided by natural forests to avoid a pure focus on carbon. The fear is that a sole-carbon focus could lead to displaced destruction from high biomass to low biomass forests, replacement of native ecosystems with monoculture tree plantations (Stickler et al., 2009), or silvicultural interventions to increase carbon stocks in forest management areas that negatively affect biodiversity (Putz and Redford, 2009). Calls for biodiversity conservation, as an integral part of REDD+ planning, stem from the perception that biodiversity is instrumental to long-term stable ecosystem service provision (Phelps et al., 2012a). There are also warning calls that too narrow of a focus on carbon could overlook negative feedbacks to human wellbeing through negative impacts on the environment at the landscape scale (Lindenmeyer et al., 2012; Phelps et al., 2012b). Divisions between carbon and non-carbon monitoring are reinforced through international and national reporting frameworks. While there are exceptions to these divisions in practice, we argue that better integration across scales and between disciplines is crucial to long-term cost-effectiveness and performance of REDD+ and its monitoring systems. These same issues of scale and disciplinary divides are pertinent to the design and application of sustainability indicators towards fostering sustainable development more broadly.

The objective of this chapter is to examine possibilities for cross-scale coordination and interdisciplinary integration in monitoring the carbon and non-carbon impacts of REDD+ (Fig. 1). We first present key concepts in monitoring as relate to REDD+. We then review available options for carbon monitoring, social monitoring and environmental monitoring, with particular attention to issues of scale. Finally, we present strategies for moving forward through more integrated REDD+ monitoring across scales and between disciplines, which can go beyond REDD+ to inform approaches for measuring sustainability in landscapes.

4.2 Key Concepts And Objectives In Monitoring

Monitoring is tracking key elements of program performance (inputs, activities, results) on a regular basis. Monitoring differs from impact evaluation, which is the episodic assessment of the change in targeted results that can be attributed to an intervention through understanding the counterfactual (i.e. what would have happened in its absence). Importantly, data gathered through the monitoring process can feed into impact evaluation. Although the discourse for monitoring in REDD+ is largely driven by the need to conform with requirements set up by the UNFCCC, the approaches employed can certainly draw on previous experiences in status assessments and effectiveness measurements, which have been widely used in the fields of conservation and international development for decades (Stem et al., 2005).

There are some generic issues for the way monitoring works in practice. First, clearly defined objectives, users and uses are essential for efficient monitoring,

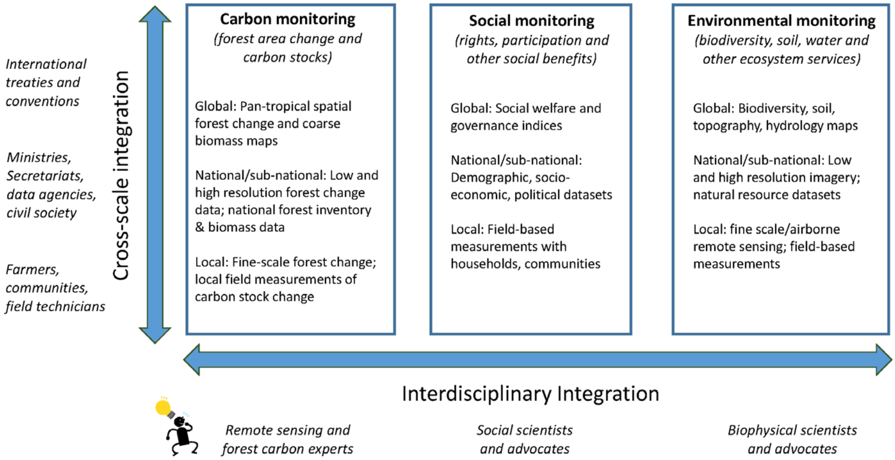

Figure 1: Scalar and disciplinary components of REDD+ monitoring.

particularly if used as basis for improved decision-making and resource management across a variety of sectors. Second, monitoring assumes that phenomena are measured and assessed at multiple points in time to track them. This temporal component requires consistency and stability in data acquisition and has often led to a focus on areas of change that are smaller than the overall area to be monitored. Third, not every phenomenon can be monitored with the same degree of effectiveness. There is a non-linear relationship between increases in monitoring precision and accuracy, and related costs. At a reasonable cost, one may be able to reach a good level of certainty, but going from good to near-perfect can increase such costs exponentially. This is exemplified by the increasing costs for acquiring and processing of satellite data with higher spatial and temporal detail (GOFC-GOLD, 2013), or the growing number of field plots and observations needed to reduce errors in carbon inventories. In related accounting, the focus on getting the big things right is inherent. For instance, in the IPCC GPG, the use of tiers that reflect different levels of certainty and comprehensiveness in estimating carbon stocks, focusing on priority emission sources through key source category analysis and the use of conservative adjustments, is a common approach to dealing with uncertain or incomplete data.

The objectives and reporting rules for countries in measuring and reporting the carbon impacts of REDD+ activities are rather clearly defined in UNFCCC decisions and the IPCC GPG. With these objectives in mind, the technical community has developed dedicated guidelines and training materials to support countries in these efforts (i.e. GOFC-GOLD, 2013). There are two stages of monitoring, which correspond to the REDD+ design and implementation phases, respectively. In the first stage, the goal is to develop a baseline or reference level (i.e. counterfactual) based on existing or new data. In the second stage, the goal is to monitor changes against the baseline. These two stages can also relate to monitoring the social and environmental impacts of REDD+, with *ex ante* impact analysis helping provide the necessary data to develop REDD+ strategies, and *ex post* impact evaluation used to measure the causal effects of REDD+ interventions. Importantly, evaluation of impacts during REDD+ implementation can help inform modifications needed through learning and adaptive management (Lawlor, 2013).

When compared to carbon monitoring, the objectives of social and environmental monitoring in REDD+ are less clearly and less strictly defined internationally. Aside from the international requirement that Safeguard Information Systems should be “transparent, consistent, comprehensive and equitable” and “build upon existing systems, as appropriate” (UNFCCC, 2011b), countries are not given much guidance on the use of appropriate indicators, data collection methods and reporting frameworks. While minimal guidance supports national ownership and provides space for independent experimentation in complex country-specific contexts, it also creates uncertainties and very high transaction costs if each country is “re-inventing the wheel.” Additionally, while the notion of the counterfactual is intrinsic to carbon monitoring through reference level setting and additionality requirements (i.e. showing that the intervention results in lower emissions than the baseline scenario), there is little use of counterfactual scenarios for understanding socioeconomic or other environmental outcomes of REDD+ (Caplow et al., 2011).

Beyond the international negotiations, there is a broader set of objectives for national REDD+ monitoring, which present clearer pathways and opportunities for linking carbon, social and environmental monitoring. These objectives are not beholden to the UNFCCC process, but reflect the need for national forest monitoring to evolve to: i) underpin and stimulate strategies and priorities for REDD+ implementation; ii) track REDD+ activities and both carbon and non-carbon impacts; and iii) support the generation and sharing of benefits. For all three objectives, a greater understanding of common concepts between different monitoring approaches can enable more harmonization among them. Increased integration can also help make REDD+ monitoring more cost-effective.

4.3 Options For Monitoring The Carbon And Non-Carbon Impacts Of REDD+

4.3.1 Carbon Monitoring

Robust data and methods for estimating greenhouse gas emissions from and removals by forests are crucial for REDD+ (UNFCCC, 2009; UNFCCC, 2011b). Countries have been encouraged to establish national forest monitoring systems based on the IPCC

Guidelines (IPCC, 2006). These guidelines have been agreed upon internationally and have been used for many years for Kyoto reporting and to generate UNFCCC national communications. Measuring and monitoring forest carbon emissions at the national level involves estimating and monitoring changes for two key variables: i) area of deforestation and degradation (activity data); and ii) terrestrial carbon stock densities per unit area (emission factors; Verchot et al., 2012; GOFC-GOLD, 2013). Many REDD+ countries are starting with large gaps in capacity for carbon monitoring and have concrete plans to improve this capacity as part of REDD+ readiness activities (Romijn et al., 2013).

While the IPCC GPG provides the framework for emissions estimation and reporting, there are several tools and approaches for carbon monitoring, some of which may be more appropriate in different contexts (Table 1). The IPCC methods are particularly suitable for evaluating the impacts of forest clearing for commercial agriculture and infrastructure expansion, which commonly lead to large-scale permanent conversion and can be accurately monitored through a combination of remote sensing and forest inventories. In contrast, monitoring deforestation associated with subsistence agriculture poses a greater challenge, since the disturbances are smaller and the long-term net carbon outcomes less certain (Ziegler et al., 2012). Small-scale deforestation therefore requires investigation at a finer scale, such as through the use of very high resolution imagery, or through other innovative spatial techniques, such as classifying change processes using “landscape mosaics” (Hett et al., 2012). Conversely, forest degradation processes and their specific drivers are more difficult to detect through remote sensing. The changes in carbon stocks vary greatly in space and time, and thus require more frequent ground surveying. Monitoring industrial/commercial extraction of forest products can build upon the combined use of archived satellite data, forestry concession data, and forest inventories. For forest degradation associated with local markets and subsistence, however, proxy data may be needed as historical field data sources are generally rare, and remote sensing approaches have limited ability to provide information based on archived data, which results in the lack of a proper reference level for many small-scale forest degradation processes (Skutsch et al., 2011).

Proponents of every jurisdiction or project planning to estimate the emissions impact of their REDD+ activities should do so based on appropriate data measured within the area of implementation. The IPCC has suggested a concept of different tiers for estimating emission factors, commonly measured through forest field sampling and repeated forest inventories (and reported as MgC ha^{-1} yr^{-1}). Changes in emission factors should be calculated for each of the five forest carbon pools: aboveground biomass, belowground biomass, deadwood, litter, and soil organic matter. The IPCC provides three tiers for estimating emissions with increasing levels of data requirements, analytical complexity and increasing accuracy. Tier 1 uses IPCC default values, Tier 2 uses country-specific data (i.e. collected within the national boundary), and Tier 3

Table 1: Options for monitoring approaches and data sources of the main forest change activities and drivers at the national level beyond the use of default data (adapted from Herold et al., 2011; Kissinger et al., 2012; GOFC-GOLD, 2013; Pratihast et al., 2013).

Activity/driver of deforestation and forest degradation	Indicator for mapping	Common sources for activity data (at national level)	Common data sources for emission factors/estimations (at national level)	Examples of other data on proxies and for assessing underlying causes
Commercial agriculture; clearing for cattle ranching, row crops etc.	Large clearings; post-clearing land use	– Historical satellite data (i.e. Landsat-type data time series) for deforestation area and land use following deforestation	– Traditional national forest inventories/ground measurements	– Commodity prices – Agriculture census, agricultural GDP, exports etc.
Subsistence agriculture; smallholder farming and shifting cultivation	Small clearings, often rotational fallow cycles	– Historical satellite data (i.e. dense Landsat time series and high-resolution data) for determining area and rotation pattern	– Traditional national forest inventories, ground measurements and targeted surveys – Efforts to assess long-term net emissions	– Population growth in rural and urban areas – Agricultural imports/exports – Land use practices (e.g. rotation cycles etc.)
Infrastructure expansion (roads, mines, settlements etc.)	Road networks; new mines; built-up areas	– Historical satellite data (i.e. Landsat time series) to measure deforestation area and land use following deforestation	– Traditional national forest inventories and ground measurements	– Growth in urban/rural population – Infrastructure/development programs – Mining: commodity prices/exports
Industrial/commercial extraction of forest products, such as selective logging	Small-scale canopy damage; Logging roads / infrastructure	– Historical satellite data (i.e. Landsat time series) analyzed with concession areas	– Regular national forest inventories, ground measurements, and harvest estimates from commercial forestry	– Timber prices and demand (nationally, internationally) – Timber import/exports

Continued **Table 1:** Options for monitoring approaches and data sources of the main forest change activities and drivers at the national level beyond the use of default data (adapted from Herold et al., 2011; Kissinger et al., 2012; GOFC-GOLD, 2013; Pratihast et al., 2013).

Extraction of forest products for subsistence and sale in local and regional markets (e.g. fuel wood and charcoal)	Very small-scale canopy damage; understory impacts; footpaths	– Limited historical data – Information from local scale studies or national proxies – Only long-term cumulative changes may be observed from historical satellite data	– Limited historical data – Information from local scale studies – Emission factors can be measured today and applied as consistent factors for historical periods – Important role for community-based monitoring – Besides direct forest carbon stock changes, more indirect methods (such as head loads of fuel wood) may be useful	– Rural/urban population growth – Energy use/fuel sources (% of population) – Consumption patterns and changes
Other disturbances, such as (uncontrolled) wildfires	Burn scars and associated impacts	– Historical satellite-based fire data records (since 2000) to be analyzed with Landsat-type data	– Regular emissions estimates can be applied consistently for historical periods with suitable activity data	– Land use practices (e.g. agricultural fires) – Links to other activity data to attribute fire emissions – Fire prevention – Natural wildfire events

uses actual inventories with repeated measurements to directly measure changes in forest biomass and/or well-parameterized models in combination with plot data.

The concept of tiers emphasizes how different kinds of data can be useful for carbon monitoring in REDD+. Ideally, both activity data and emission factors should be measured with sufficient precision and accuracy (fine-scale monitoring), but this is sometimes not achievable due to a lack of capacity and resources. Thus, questions arise about using available, coarser-scale datasets as supplementary or complementary sources. For example, if a local REDD+ project is able to build on a strong national forest monitoring system, including suitable remote sensing-based activity data and emission factors based on detailed national inventories, the estimates obtained can be robust with only a limited amount of refinement or additional data needed. Alternatively, since many national monitoring systems are still evolving, regional or global datasets can be used. More large-area or pan-tropical datasets on forest change (Hansen et al., 2013) and biomass (Saatchi et al., 2011; Baccini et al., 2012) are becoming available that can provide data on scales that matter for REDD+. These datasets, however, often have an intrinsic requirement of a consistent (global) definition and method to ensure large area consistency, which often implies a trade-off in local precision and accuracy. This trade-off can be exemplified by the use of remote sensing for REDD+ monitoring (De Sy et al., 2012), which is shown here as the operational ability of different forest information products at multiple scales (Table 2). Commonly, remote sensing research starts from the local experimental level to develop and test technologies and methods, and if suitable, moves towards larger demonstration areas or even global level analysis. While monitoring forest area change is operational at all scales, approaches for mapping forest types or biomass are not yet used by many REDD+ countries. Given that the most appropriate and suitable methods for generating forest information products often depend on national and local circumstances (e.g. types of forest changes, data costs and availability, technical capabilities, size of forest area, drivers, etc.), coarser-scale products often show less suitability for use at national and subnational scales without additional calibration or integration. Yet, as more coarse-scale datasets become available with increasing degrees of precision and accuracy, their usefulness for REDD+ monitoring at national and subnational levels also increases and should be evaluated by dedicated research at multiple scales.

Aside from the need to acquire appropriate data, different frameworks are available to estimate and report on the carbon impacts of REDD+. At the national level, the IPCC GPG provides the rules and tools for international reporting. At local and subnational levels, other reporting frameworks, such as VCS, are more commonly used. Importantly, these frameworks are designed for different users and uses; the first is for reporting to the UNFCCC, while the second is to feed into the voluntary carbon market. It is thus not uncommon that reporting to the different frameworks, even when based on similar data (i.e. activity data and emission factors), will lead to different results due to different definitions, time frames, accounting rules, approaches for developing reference levels, activities to include, use of conservative

Table 2: Operational ability of different forest information products in REDD+ context (black = high, dark grey = intermediate, light grey = low and white = limited or no operationality). Adapted from De Sy et al., 2012.

Forest information product	Local pilot and research sites	Large research demonstration areas	National level
- Forest area change monitoring			
- Near real-time deforestation detection			
- Land use change patterns and tracking of human activities			
- Forest degradation monitoring			
- Monitoring of wildfires and burned areas			
- Biomass mapping			
- Subnational hotspot monitoring			
- Forest type mapping			

adjustments, etc. Currently, the differences between estimates derived from different accounting methods are often greater than the actual difference in the data, and comparability is often limited. Therefore, cross-scale integration of national and subnational estimations will require agreement on the level of data and fundamental approaches used.

4.3.2 Social Monitoring

It has been widely accepted that REDD+ must minimize social risks and maximize social benefits to be effective and to support countries' rural development goals. Following the logic of social safeguards, social monitoring can focus on three main categories: i) respect for knowledge and rights of indigenous people and local communities; ii) full and effective participation of local stakeholders; and iii) enhancement of other social benefits. For the first, while respect for local rights is a broad concept, much of the REDD+ literature to date has convened on the importance of tenure security, or clear and enforceable local rights to forests and carbon (e.g., Corbera et al., 2011; Larson et al., 2013). For the second, full and effective participation requires high levels of engagement by local stakeholders throughout REDD+ design and implementation. It begins with access to information, which is reflected in the requirement of free, prior, and informed consent (FPIC), as target communities choose whether or not to participate in REDD+. It also links to broader multi-level governance issues with mechanisms needed to promote local engagement in higher-level REDD+ processes (Agrawal et al., 2011). For the third, enhancement of other social benefits can be conceptualized as improving human wellbeing, assuring equitable benefit sharing and increasing the adaptive capacity of local people (Lawlor, 2013). There are important interconnections among these social dimensions; for instance, secure tenure can be considered the basis for improving local livelihoods and increasing local adaptive capacity (Chhatre et al., 2012), while greater local participation in REDD+ decision making may result in more equitable benefit sharing and long term support of the activities (Cromberg et al., 2014).

The issue of scale is quite relevant for social monitoring, since the determined social outcomes of REDD+ will likely differ based on scale and level of aggregation of analysis. For instance, while protected areas may have substantial socioeconomic effects (both positive and negative) on local people, a global study of 136 countries showed that such effects were not discernable at the national scale (Upton et al., 2008). Social outcomes will also vary among and within social groups, and net benefits may be distributed unevenly. In Thailand, while protected areas contributed to economic development and reduced poverty, they may have increased overall local inequality (Sims, 2010). Disaggregation into social groupings (i.e. along gender, age and ethnicity lines) is needed to understand uneven social impacts, and is most critical in places with greater inequality (Daw et al., 2011). Given the complexity of social monitoring, the key challenge is developing simple, yet adequate methods and performance indicators that are appropriate to the scale of analysis.

To select and monitor social performance indicators, countries can draw on existing national socio-economic monitoring programs, and leverage both secondary and primary datasets. A variety of national-level secondary datasets are publicly available, such as the World Bank Living Standards Measurement Study (LSMS, 2014) and USAID Demographic and Health Surveys (DHS, 2014), which have been applied in many REDD+ countries in partnership with national statistical agencies. These secondary datasets can be used in REDD+ monitoring and complemented by primary data collection in the field. For social monitoring at the local level, more expensive primary data collection would include extensive household surveys, whereas a less expensive approach would be based on participatory methods at the village level. The World Bank's Poverty Mapping technique provides an interesting example of combining census and household-level data towards informing policies that are better tailored to local conditions (Bedi et al., 2007). The application of mixed methods at multiple scales in social monitoring can help provide a more accurate understanding of the results-based performance of REDD+, which could be misinterpreted through the use of one dataset or method alone (Jagger et al., 2010). In all REDD+ monitoring, engagement with relevant stakeholders throughout the process can help address issues of legitimacy of data and results. Such engagement is also required to address social safeguards and ensure local participation and ownership of the process.

To be able to attribute social outcomes to specific REDD+ interventions, impact evaluation is needed in addition to monitoring. There have been detailed reviews of specific methods and indicators that can be used in social impact evaluation (e.g. Schreckenberg et al., 2010), along with guidebooks for conservation practitioners (Wongbusarakum et al., 2014), with distinct mixed methods approaches favored depending on the amount of time, funds and capacity available (Lawlor, 2013). The Participatory Theory of Change approach involves broad stakeholder consultation in the REDD+ design stage to provide a road map for expected changes that a given intervention will have, focusing on selection of indicators that can most strongly inform attribution (Richards and Panfil, 2011). Multiple theories of change are created to establish attribution and eliminate rival explanations. The strength of this approach is that it is highly participatory and relatively inexpensive; its main weakness is that its robustness depends on how indicators are selected, measured and analyzed. Participatory approaches can be complemented with rigorous social impact evaluation at the site level, which involve the application of experimental (e.g. randomization) or quasi-experimental methods (e.g. Before-After-Control-Intervention, BACI) to evaluate REDD+ impacts (Jagger et al., 2010). Experimental approaches, such as randomization, can only be used if REDD+ participants are selected randomly (e.g. through a lottery system) allowing for no bias between treatment and control groups. Quasi-experimental approaches that employ matching techniques to create controls and measure conditions before implementation of REDD+, such as BACI, are more rigorous in establishing attribution, but also more time-consuming and difficult to implement. Importantly, these same concepts apply to environmental monitoring. While countries will need to report on the social performance of REDD+ at relatively coarse scales, fine-scale monitoring of local processes can help inform of national-level indicators for respecting local rights, ensuring local participation and enhancing social co-benefits in an iterative process.

4.3.3 Environmental Monitoring

Environmental monitoring in REDD+ focuses on the need to promote forest conservation and other environmental co-benefits, which loosely translates into biodiversity conservation and ecosystem services provision. The Cancun safeguards propose that REDD+ activities should take into account the multiple functions of forests and other ecosystems, be consistent with the conservation of natural forests and biological diversity, and not be used for the conversion of natural forests but instead to incentivize their protection.

The biodiversity component of environmental monitoring in REDD+ has foreseeably received the most international attention. Biodiversity monitoring at national or global scales has been a concern of conservation science pre-dating REDD+ (Stoms and Estes, 1993; Innes and Koch, 1998). In recent years, the UN Convention on Biological Diversity has recognized the potential opportunities and risks of REDD+, including leveraging REDD+ as a tool for biodiversity conservation in their post-2020 targets (CBD, 2012). There has been a growing policy focus on the environmental co-benefits of REDD+, along with practical information on biodiversity monitoring for REDD+ (Latham et al. 2014). Several unresolved issues, however, stand in the way of a faster uptake of environmental safeguarding in subnational and national REDD+ designs.

Monitoring of biodiversity and other ecosystem services in the tropics is historically hindered by a shortage of data (Martinez et al., 2011) deriving from chronic underfunding of conservation science, in general, and more evidently so for taxonomic work in biodiversity-rich tropical rainforests (Balmford and Whitten, 2003). This situation is compounded by the high cost of multi-taxa field studies (Margules et al., 1994; Lawton et al., 1998). Moreover, biodiversity and ecosystem services are distributed unevenly within forests and between forests and other ecosystems, and the lack of a common measure, such as metric tons of CO_2 in carbon monitoring, poses a challenge in how to compare results both between habitats within a country or landscape, and between countries and landscapes (Dickson and Kapos, 2012).

The issue of scale therefore becomes a centerpiece of the debate on environmental monitoring in REDD+. Fine-scale field measurements provide important but spatially limited information at high costs (as an exception, see Bassett et al., 2004), and efficient pathways for scaling-up to national and international monitoring systems are largely lacking. On the other hand, at higher geographic scales, biodiversity (or specifically gamma diversity; Hunter, 2002) is usually measured through remote sensing and expressed as changes in land cover type. Although this approach is key to carbon accounting in REDD+, it is still unable to translate into actual changes in species and populations, and importantly, the related consequences of these changes on ecosystem functioning. Without this information, our understanding of the environmental risks and benefits of REDD+ will remain largely inadequate to effectively inform its design.

Environmental monitoring in REDD+ is reinvigorating a long-standing challenge in ecology and conservation. Some authors are calling for the development of effective, flexible biodiversity indicators to maximize field-monitoring efficiency (Gardner et al., 2008), while others argue that ecological indicators must reflect the health of a landscape or water catchment (Stickler et al., 2009). Although the relationships between potential indicator species and total biodiversity are not well established (Lindenmeyer and Franklin, 2002), it has been proposed that ecological indicators should be easily measured, sensitive to change and respond to stress in a predictable manner, anticipatory, and have a known response to disturbances with low variability (Dale and Beyeler, 2001). In tropical settings, bats (Waldon et al., 2011) dung beetles (Rodriguez et al., 1998), butterflies (Beccaloni and Gaston, 1995), and several arthropod groups (Kremen et al., 1993) represent taxa that are common, diverse and sensitive to change. Focusing on such taxa relies on evidence that many taxonomic groups respond similarly to habitat modification (Schulze et al., 2004). Nevertheless, there are concerns about depending on a small number of species without considering the full complexity of the ecological system (Carignan and Villard, 2002). There are also concerns with choosing ecological indicators that are not clearly informed by long-term goals and implementing monitoring programs that lack scientific rigor in identifying suitable target organisms (Dale and Bayeler, 2001). Alternative models have been proposed that place more emphasis on community assembly metrics, such as (relative) abundance, richness, composition and (a-) symmetry (Dufrene and Legendre, 1997). Diversity indices rather than species count are widely used in ecology, as they provide a common-standard, comparable measure as well as capturing ecological complexity beyond species richness (Scholes and Biggs, 2005). Such aggregated indices have been developed and widely used in plant community composition and structure. It is also recognized that animal diversity is often closely linked and predicted by vegetation diversity; therefore, vegetation surveys are and remain one of the most efficient tools for biodiversity monitoring (Noss, 1990; Noss, 1999). On the other hand, the use of novel tools such as camera traps to inexpensively obtain distributional and abundance data over time (Ahumada et al., 2013; Rendall et al., 2014), are emerging as additional tools that can prove especially useful where large-territory species (which may not reflect local vegetation trends) are integral part of the conservation effort. This wealth of knowledge indicates that rigorous biodiversity monitoring is possible, albeit not necessarily technically easy or inexpensive. Thus, further advances such as the identification of “high performance indicators” as part of a framework that includes assessing the costs of monitoring different taxa (Gardner et al., 2008; see also chapter 3), are needed, but monitoring in REDD+ can rely on a solid scientific base that can be tailored for its purposes.

Ecosystem-level monitoring is also faced with challenges as to what should be measured. Despite a clear interdependence between biodiversity and ecosystem function (Loreau et al., 2001; Hooper et al., 2005), this relationship cannot be used a priori to serve as a proxy for monitoring purposes. Ecosystem services can derive from biodiversity-independent processes and factors (e.g. a single or few plant species can provide soil erosion control on a riparian bank) or can operate at a landscape scale (i.e. encompassing multiple habitats with distinct biodiversity values). On the other hand, there are also clear opportunities for maximizing monitoring efficiency wherever biodiversity and a target ecosystem service are in spatial, functional and temporal synchrony. A recent assessment, however, has noted how “the relationship between biodiversity and the rapidly expanding research and policy field of ecosystem services is confused and is damaging efforts to create coherent policy” and calls for caution in oversimplifying a complex relationship (Mace et al., 2012).

The above challenges make it difficult to devise a clear pathway for environmental monitoring without further research, which likely contributes to the lack of explicit environmental co-benefits in the national strategies of most REDD+ countries. While social safeguards are seen as strictly necessary to obtain stakeholder support, even before considering any humanitarian and development benefits, environmental co-benefits beyond the "do no harm" principle are less central to the success of a market-driven mitigation scheme such as REDD+ (Phelps et al., 2012b). While environmental safeguards are well anchored in the discourse, the extent to which REDD+ should achieve additional co-benefits is less clear. Ecosystem services, watershed and species protection all have the potential to harness consumer support and willingness-to-pay or, in some cases, even be the primary motive for establishment of a REDD+ project (Cerbu et al., 2011), yet can also increase design and implementation costs making their inclusion in REDD+ less appealing to investors with a primary focus on carbon (Phelps et al., 2012b). Although extreme scenarios of a REDD+ scheme devoid of environmental co-benefits versus a scheme that prioritizes environmental co-benefits over carbon are unlikely, a satisfactory middle ground is yet to be reached (Dickson and Kapos, 2012).

4.3.4 Possibilities For Integrated Monitoring?

REDD+ countries currently follow entirely separate reporting frameworks for carbon Measurement, Reporting and Verification (MRV) and for Safeguard Information Systems. Yet, there can be no holistic understanding of the impacts of REDD+ without integrated monitoring of its carbon and non-carbon impacts, or at least integrated analysis of observation data from different sources. While overall integrated monitoring would likely be difficult to achieve, coherence between data sources can help in understanding and balancing the trade-offs and synergies between reducing emissions, enhancing local rights, participation and wellbeing, and conserving biodiversity and other ecosystem services. Given limited funding for REDD+ monitoring, it is also a potential way to make it more cost-effective. The key is to identify pathways for integration, through complementary data collection methods at multiple scales, and to generate empirical evidence that demonstrates the relationships between the carbon and non-carbon impacts of REDD+.

Clear opportunities exist for integrating carbon and environmental monitoring (Fig. 2). As highlighted in the previous section, and in a recently-proposed framework for biodiversity monitoring integration in REDD+ (Gardner et al., 2012), there is considerable existing knowledge from ecology and conservation that could be integrated into the strategic planning of REDD+. Combined carbon and biodiversity analysis can be conducted at various scales to identify either carbon neutral solutions that offer varying benefits for biodiversity, or opportunities where minor sacrifices of carbon effectiveness can deliver disproportionate environmental co-benefits (Venter et al., 2009; Thomas et al., 2013). Similar datasets can be leveraged to

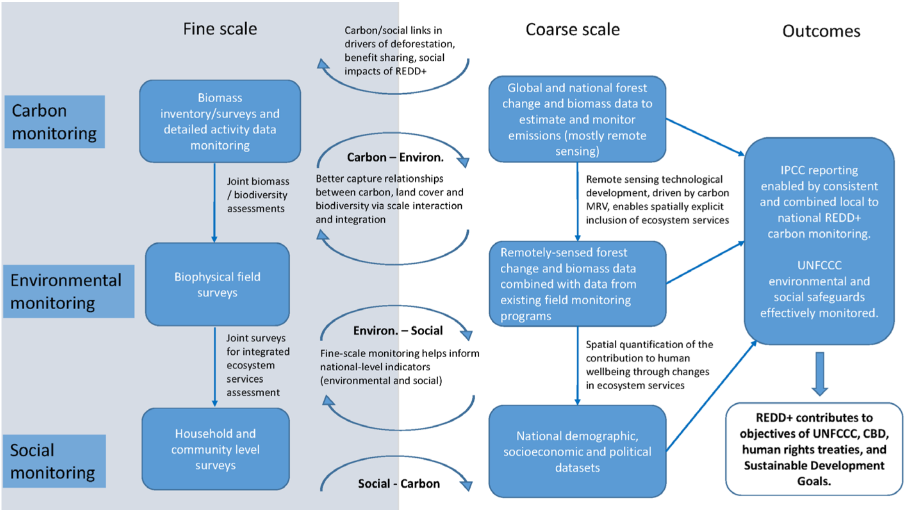

Figure 2: Possibilities for integrated carbon monitoring, environmental monitoring and social monitoring in REDD+.

measure the carbon and environmental impacts of REDD+ at multiple scales. Remote sensing, widely used to estimate forest cover and area change, can deliver a great deal of ecological information, including percent forest cover of water catchments, fragmentation of terrestrial and aquatic habitats, stream continuity, fire incidence, and soil erosion susceptibility (Stickler et al., 2009). Rapidly increasing imagery resolution and analytical processing capabilities are now being combined with terrestrial and aerial biomass measurements, which could be further developed to capture and integrate biologically relevant indicators in a spatially explicit way. For instance, adding data on the distribution of species and threats, along with known responses of ecosystem-level variables to change in forest cover and forest management strategies can be included in REDD+ prioritization processes (Gardner et al., 2012; Thomas et al., 2013). Biologically meaningful monitoring of ecosystem function at a coarse scale (Stickler et al., 2009) can help overcome the funding barrier to global biodiversity monitoring and associated ecosystem services. REDD+ presents an enormous opportunity for scaling up environmental monitoring to a global level from its current local and regional focus. It is only very recently that remote sensing science, ecology and conservation have started to coordinate efforts (Pettorelli et al., 2014), providing the first steps in informing current and long-term trends in carbon, biodiversity and other ecosystem services. Such improvements, however, will only be harnessed when plans for monitoring the environmental impacts of REDD+ (and ensuring the necessary institutional coordination) are incorporated early on in REDD+ design, and environmental co-benefits are considered as a centerpiece of REDD+ beyond “do no harm” requirements (Thomas et al., 2013).

There are also opportunities for integrated monitoring of the carbon and social impacts of REDD+ (Fig. 2). Both are mediated by human behaviors, and the viability of REDD+ depends on understanding and managing the relationship between emissions reductions and improvements in human wellbeing. While the data sources for social monitoring are quite different from those for carbon monitoring, the conceptual links become clearer when drivers of deforestation, benefit sharing systems and measurement of the social impacts of REDD+ are considered. For instance, understanding the socioeconomic drivers of deforestation and forest degradation is fundamental to the creation of effective REDD+ strategies, including the justification and prioritization of REDD+ interventions that address key drivers (Hosonuma et al., 2012; Salvini et al., 2014). There are instances when REDD+ strategies to address the drivers of deforestation and forest degradation could adversely impact local livelihoods (e.g. strategies to curb swidden agriculture) if no alternatives are provided. Closely linked carbon and social monitoring systems are needed to highlight such tradeoffs to be able to inform policy in an iterative way. Additionally, while carbon monitoring will help determine the flow of benefits, more integrated monitoring could help provide the foundation for benefit sharing systems that focus on activities and changes in land use practices that go beyond forested areas (Salvini et al., 2014). Finally, social monitoring is needed to understand the equitability of benefit sharing mechanisms and can guide the adaptation of REDD+ interventions, since the social impact of any intervention (e.g. support for land tenure regularization, fuel-efficient cooking stoves, agricultural intensification to reduce pressure on forests etc.) will ultimately determine its cost-effectiveness.

Finally, the concept of ecosystem services provides a platform for linking social and environmental monitoring, since these services are the benefits that people derive from ecosystems (Millennium Assessment, 2005). Aside from the global public goods of carbon sequestration and biodiversity, the value of ecosystem services depends on the location of forests in relation to beneficiaries (i.e. whose values are counted). For example, forests up-stream of drinking water supply systems generate more valuable watershed services than remote forests. Many studies have explicitly attempted to account for ecosystem services through in-depth analysis of their contribution to human wellbeing, using monetary valuation of ecosystem services as a tangible measure (Ferraro et al., 2012; Ninan and Inoue, 2013). That said, moving from a research intensive, one-point-in-time valuation to long-term monitoring remains a considerable challenge, suggesting an urgent need to advance the interdisciplinary science that investigates the full ensemble of processes and feedbacks. Synergies and trade-offs between human welfare and ecosystem services (including carbon sequestration) as related to REDD+ will be best understood through the application of monitoring and evaluation methods that use similar approaches to constructing the counterfactual (Caplow et al., 2011) and flexible systems that best reflect the context.

4.4 Lessons Learned And Way Forward

Given the lack of capacity and funds for REDD+ monitoring in many countries, greater integration of carbon, social and environmental monitoring – both across scales and between disciplines – could help make the process more cost-effective. To promote such integration, advancements are needed in three key areas.

First, there is a need for cross-scale coordination in measuring, reporting and verifying the carbon and non-carbon impacts of REDD+. The challenge of applying international guidelines at the national level can be seen in countries' responses to the sustainable forest management criteria and indicators, which stem from the Forest Principles defined at the UN Conference for Environment and Development (UNCED) in Rio de Janeiro in 1992. These criteria and indicators consider social, economic, environmental, and cultural dimensions, are to be applied at regional, national and local (i.e. forest management unit) levels, and are commonly accepted as appropriate tools for defining, assessing and monitoring progress toward sustainable forest management (Castañeda, 2000). A recent assessment of the Montreal Process, which includes 12 temperate and boreal countries that in 1995 agreed to report on a common set of criteria and indicators, showed a lack of harmonization in reporting. The majority of countries had not reported on the agreed upon criteria and indicators, likely due to data collection difficulties or lack of commitment to the agreements, and the assessment highlighted clear areas for improvement in communication and consultation with stakeholders (Chandran and Innes, 2014). For REDD+ monitoring to work, it is critical to understand how monitoring systems can be elaborated from existing national policies, indicators and data so that monitoring requirements are a source of support and not a burden. REDD+ country experiences in establishing Safeguard Information Systems and advancing with monitoring efforts should be widely disseminated and contribute to the international policy process in a "bottom up" fashion. Additionally, as national REDD+ frameworks develop, countries can learn from and incorporate advances already made at subnational levels, so that the hard lessons learned by subnational jurisdictions and projects are not lost as national carbon monitoring systems and Safeguard Information Systems are consolidated. In this context, there is the opportunity to think beyond forests and forest monitoring towards the engagement of multiple sectors and stakeholders in measuring sustainability more broadly. Considerable needs for research and action lie in this area.

Second, there is a need to resolve the issue of coarse- versus fine-scale monitoring methods and datasets to facilitate the choice of appropriate performance indicators for REDD+ monitoring. Performance indicators should be: i) easy to understand; ii) applicable at multiple scales; iii) applicable to any location; iv) efficient to measure and monitor; v) sustainable in providing data; and vi) able to be improved over time. There is a disconnect between the widely-available coarse-scale data on forest cover change derived from remote sensing, and the fine-scale data needed to monitor forest degradation processes and changes in social and environmental conditions. Fine-scale data are much more limited, are costly to obtain and generally lack historical measures. The constraints associated with fine-scale monitoring highlight the need for higher levels of aggregation, especially since monitoring efforts ultimately need to align with broader UNFCCC reporting guidelines. Yet, such aggregation threatens the loss of important information on local processes. Consequently, there is a need to establish clear pathways through which local-level information can inform and update any attempts at aggregation. This represents the rationale behind the call for establishment of robust sustainability indicators to evaluate the impacts of conservation and development projects, which can inform efforts to measure sustainability more broadly (Agol et al., 2014). That said, the robustness of indicators is ultimately dependent on the amount and quality of field sampling for development and testing. Since effective monitoring is hampered in many tropical forest countries by lack of capacity and funds for even the simplest monitoring efforts, creative ways to reduce the high costs associated with local-level data collection should be explored. For instance, collection could be partly (but not exclusively, to avoid sampling bias) directed towards sampling potentially more vulnerable populations to create a baseline against which future data collected could be measured (Lawlor, 2013). Although not without its own set of challenges, there are also important opportunities to involve local people in community-based monitoring to address some of the smaller-scale processes, and make links to higher-level monitoring efforts in both environmental and social fields (Bassett et al., 2004; Pratihast et al., 2013).

Further technical work can help understand the differences between the results of coarse- versus fine-scale monitoring of both carbon and non-carbon impacts. Information on the early impacts of pilot subnational REDD+ initiatives is beginning to be consolidated with clear opportunities to compare methods used for assessing performance. For instance, there are opportunities for the Center for International Forestry Research (CIFOR) and The Nature Conservancy (TNC) to compare REDD+ monitoring and evaluation efforts at two subnational sites in Brazil (São Félix do Xingu) and Indonesia (Berau). At these sites, CIFOR's REDD+ impact evaluation is based on a quasi-experimental BACI approach using village, women and household surveys, along with fine-scale spatial and biomass data. While this approach is considered very rigorous for measuring impacts, data collection is limited to relatively small areas within the larger sites and is expensive and time-consuming to implement. In contrast, TNC is using focus group discussion, key informant interviews and secondary data to monitor a larger set of indicators of human wellbeing, an approach that allows for a broader coverage at lower costs, but may sacrifice data quality and depth. Empirical, multidisciplinary analysis of the results-based performance associated with these different monitoring systems can help in the development of coarse-scale indicators that can capture typical outcomes from aggregated fine-scale mechanisms to be used in future REDD+ monitoring efforts.

Finally, there is an important opportunity to promote more interdisciplinary integration in monitoring systems to reduce costs and advance our understanding of synergies and trade-offs between carbon and non-carbon benefits. As discussed earlier, many of the same remotely sensed and field-based datasets that are being leveraged to measure changes in forest carbon emissions can be used to assess changes in biodiversity, hydrology and water resources, and soil resources. There are also key linkages to social benefits. Although most countries report carbon and non-carbon benefits separately, there are interesting examples of bridging this divide. For instance, the Food and Agriculture Organization of the UN and the government of Finland jointly support the Peruvian National Forest Inventory, which is making steps to integrate biophysical and socioeconomic monitoring across the country. In addition to learning from such initiatives, there is an opportunity to promote more interdisciplinary research at the local level. Results can be scaled up to inform the creation of national and global indicators, test their robustness and iteratively update the current set of indicators towards achieving coarse scale, relatively inexpensive monitoring that does not miss the implications of critical local processes. To achieve this, scientific disciplines that remain largely isolated will need to increasingly work together and develop common protocols and frameworks to achieve true interdisciplinarity. Integrated monitoring of REDD+ performance is not only important for assessing adherence to safeguards, but can go well beyond REDD+ to inform indicators of sustainability towards promoting benefits for both people and the environment.

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5 Measuring Indicators For Sustainable River Basin Management

Dorice Agol

5.1 Introduction

River basins are complex socio-ecological systems that reflect interactions between humans and their environment as well as multiple interests and activities. Subsequently, sustainable river basin management calls for an integrated approach requiring a balance between social equity, economic efficiency and environmental sustainability (e.g.Gallego-Ayala and Juízo 2012). This is to maintain the various ecosystem services that river basins provide, which range from provision of water supplies for domestic, hydropower and irrigation to regulatory, supporting and cultural/aesthetic benefits (Grossman, 2012; Comino et al., 2014; Fu et al., 2014). However, experiences across the world have shown that optimizing these ecosystem services is a huge challenge due to the need to reconcile trade-offs that are often inherent at their interfaces (e.g.Swallow et al. 2009). For instance, the negative consequences of wetland drainage and deforestation with the aim to foster agricultural development are well documented and evidence shows that such practices can adversely reduce ecological functions and integrity of natural ecosystems (Izac and Swift, 1994; Urama, 2005; Swallow et al., 2009; Guo et al., 2014). This holds true across Sub-Saharan Africa where it is often challenging to realize socio-economic development and environmental sustainability simultaneously due to issues of poverty, unemployment and inequality and poor health.

The basis for socio-economic development is a healthy river basin system with adequate supplies of natural resources such as land, water, energy, timber, food and minerals. Safeguarding adequate supplies of these resources requires robust sustainable river basin management approaches. Sustainability Indicators (SIs) are being increasingly used to monitor and appraise river basin management strategies because they can help to quantify and/or qualify the impacts of programmes, projects, policies and institutions within different contexts (Madulu, 2005; Harmancioglu et al., 2008; Owens et al., 2008; Li and Yang, 2011). Since the Rio Earth Summit the use of SIs has been widespread in fostering sustainable development (UNCED, 1992). There is a range of SIs that is embedded in socio-economic, cultural, political and environmental spheres (e.g. Moldan et al., 2012). SIs are defined within different sectors, including water, forestry, energy, land use, transport, tourism and mining (Hezri, 2004; Strezov and Evans, 2009; Blancas et al., 2011; Shaheen et al., 2011; Popovic et al., 2013). Examples of commonly used sustainability indicators in river basins include water quantity and quality, level of access, richness in biodiversity, efficiency in policy, institutions and regulatory frameworks. These SIs can be measured directly and/or indirectly and the success of their appraisal depends on the level of available technical skills and financial resources. This chapter discusses opportunities and challenges associated with measuring SIs for sustainable management of River Nyando (RNB) in Western Kenya, East Africa. It highlights commonly used river basin management concepts and theories such as decentralization, multi-stakeholder participation, institutional arrangements and ecosystem services.

5.2 Background and Context

The River Nyando, which drains into Lake Victoria on the Western Kenyan side, East Africa, is just over 150 kilometres long and covers an area of approximately 3618 km^2. It runs through three distinct hydrological zones namely; upper, middle and lower. The upper zone and parts of the mid zone are cooler with altitudes of approximately 3,000 meters above sea level (asl). The lower and some areas in the mid zone are hotter and drier with lower altitude (~1000m asl). Rainfall in the Nyando River Basin is bimodal with long events in March/April and short ones in October/November. Temperatures can drop up to five Degrees Celsius in the cooler upper zone while in the low lying areas, more than 30 Degrees Celsius have been recorded in the past.

Currently, there are more than one million people living in the River Nyando Basin and many are rural dwellers whose livelihoods depend directly on its natural resources. Some parts of the river basin are so densely populated that they support more than 1,000 people per square kilometre. Subsistence farming is commonly practised and crops such as maize, millet and vegetables are grown. There are also commercial activities in various parts of the RNB mainly growing and processing of tea, rice and sugarcane (Agol, 2010).

Since the last 50 years, the River Nyando Basin has experienced intense environmental problems such as deforestation, soil erosion, pollution and climate change (LVEMP, 2005; Raburu and Okeyo-Owuor, 2006). The River Basin has undergone dramatic land-use changes due to intensive agricultural activities. Deforestation is widespread and has claimed much of the forest cover especially in the upper zone (GoK, 2010; API, 2010). Soil erosion accounts for over 50% of land degradation in the RNB and is caused by intensive forest clearance, over-grazing, sand harvesting, land fragmentation and quarrying (Swallow et al 2003; Shepherd and Walsh 2002). The lower zone is prone to floods during extreme rainfall events and during the *El Niño* events of 1990/ 2000, many parts of the lower zone were inundated with water levels exceeding three metres high (JICA, 2007; Olang and Furts, 2011). Pollution is also a common problem in the RNB and in the rainy seasons, sediments from the catchment usually get deposited into the river. These sediments include untreated agro-chemical and municipal wastes which are key sources of pollutants (e.g. Raburu and Okeyo-Owuor, 2006).

Many of the environmental problems in the River Nyando continue to threaten its socio-economic and ecological functions (LVEMP, 2005). For example, previous research estimated that more than 3 million tonnes of soil has been eroded away from the catchment leading to depletion of key nutrients mainly phosphorus, nitrogen, organic matter and soil moisture and subsequent poor crop yields (Swallow et al 2003, Shepherd and Walsh 2002). Common weeds mainly *Striga*, tend to take advantage of these nutrient-poor conditions by outcompeting food crops (Atela et al., 2008).

Sustaining the functions of the RNB requires strategic and practical actions that are multifaceted in nature. It requires the sustenance of the various ecosystem services that the River system provides including adequate water quantities and good water quality and high levels of biodiversity. It also requires active participation of multi-stakeholders across the River Basin.

5.3 Sustaining River Basin's Ecosystem Services

Securing a range of benefits that people derive from ecosystems – from provision of food, fibre, and water, to regulation of climate, pests and diseases as well as opportunities for recreation, cultural and spiritual experience- is important for human wellbeing (MA, 2005). The River Nyando Basin is an important socio-ecological system which provides these ecosystem services which are derived from its rich upland forests, wetlands and flood plains. The river is an important source of food such as fish and wild vegetables, and provides water supplies for domestic, agricultural and industrial use. It regulates the local climate around the Lake Victoria region and filters pollutants from multiple sources. The River Nyando is aesthetic with its course meandering across different landscapes and within its catchment are inhabitants of multiple ethnic communities with distinct cultures and traditions.

However, threats to these ecosystem services such as deforestation, soil erosion, pollution and climate change continue to disrupt the river's ecological and socio-economic functions. For example, deforestation in the upper catchment has caused reductions in biodiversity loss including indigenous tree species (GoK, 2010; API, 2010) and is threatening local livelihoods which are dependent on them. Over the years, many wetlands in the RNB have disappeared due to activities such as wetland drainage and use of agro-chemicals and this has destabilized their ecological functions such as regulation of flood and pollutants. Since the early 1990s, efforts have been made to minimize these environmental problems in the River Nyando Basin in order to sustain its ecosystem services. Key strategies have focussed on maintenance of adequate water supplies, good water quality and high levels of biodiversity.

5.3.1 Securing Adequate Water Supplies For All

Securing adequate water supplies is an important perquisite for socio-economic development and environmental sustainability (Lawford et al., 2013). The rationale for sustaining adequate water quantities in a river basin is to improve water security for socio-economic activities and environmental sustainability. The RNB has multiple water sources which are used for various purposes such as domestic, irrigation and industrial activities. They include the river and its associated wetlands (streams, springs, ponds), groundwater and rainfall. The river is the main source of water for irrigation and industrial activities. The mid and upper zones are endowed with numerous springs many of which provide water for domestic purposes.

To sustain adequate water quantities in the RNB, there have been efforts by governmental and non-governmental organizations, private sector and local communities to improve rainwater harvesting, stream flow diversion, spring protection and groundwater abstraction. During rainy seasons, some households can harvest water from rooftops while small dams (water pans) are used to divert stream flows for small-scale irrigation and livestock watering. Abstracting groundwater via shallow wells and boreholes for domestic and small-scale irrigation activities is also common especially due to external funding from the donor community. Many communities are protecting spring water to improve their productivity and quality while piped water systems are used by well-established farms, industries and institutions (e.g. hospitals and schools).

The variety of water sources across the RNB and their productivity levels are obvious indicators of water availability. The Water Resources Management Authority (WRMA) is legally mandated to monitor water use across the whole basin. WRMA's key responsibility is to do water accounting through surveys which involves identifying all water sources, determining their level of productivity and their spatial and temporal distribution. This requires large investments in capital and human resources which unfortunately are limited. Thus although past surveys have revealed that there are multiple water sources across the RNB, one cannot be confident that the multiple water sources are a reliable indicator for water availability because there lacks a systematic monitoring to establish their productivity. For example, there is no up-to-date information on water supply against demand especially owing to the fact that water insecurity level in river Nyando is rising due to a growing human population combined with problems such as pollution and prolonged drought.

5.3.1.1 Determining Water Productivity, Promoting Equitable Allocation

Measuring productivity of the variety of water sources in the RNB can help to determine the water balance, that is supply versus demand. Some efforts are being made by the WRMA to carry out direct measurements of stream flows (cubic metres per second), discharge, evaporation rates (evaporation pans) and water levels (rain gauges). These measurements are done at designated sites along the main course of the river and in some seasonal streams. However, to determine the full situation that truly reflects water productivity across the River Basin is challenging due to limited systematic research and monitoring processes. For example, groundwater production and stream flows/dynamics in the river basin are less well understood (Agol, 2010). The lack of knowledge limits efforts especially by key stakeholders such as conservationists, development practitioners and the local water resources to manage water resources of the Nyando River sustainably.

Sustaining adequate water supplies requires allocation strategies that promote water equity between competing users (Cai et al., 2008; Dai and Li, 2013; Li et al., 2015). The fact that water is now commonly valued as an important economic good (e.g. Agol, 2010), has created opportunities to apply economic instruments such as permits to allocate water in the many river basins across the world. In the River Nyando, the WRMA allocates water through permit mainly to large-scale water users such as irrigators and industries. Permits are a form of water use control mechanism and can be used as indicators of trends in water abstraction, in practice however, there are illegal abstraction activities across the River. Thus a meagre record of the number of permit holders is not sufficient to infer trends in water use and subsequent demand by the population. In addition, there are hundreds of small-scale water abstractors across the River Nyando many of which are not regulated by the WRMA. According to the Water Act 2002, small-scale water abstractors do not necessarily require permits and this means that the WRMA cannot monitor their water use activities across the River Basin. Without adequate information on trends in water use, it is difficult for the WRMA to allocate water efficiently to different users across the River Nyando Basin. Certainly, there needs to be a systematic data collection and analysis, for example on the different water sources, their productivity levels, type of users (illegal and legal water users), population densities, household size to determine water use patterns.

5.3.1.2 Access To Water Resources

The level of access to water resources by different users can be a useful measure of sustainability (Gustavson et al., 1999; Joziasse et al., 2007; Milman and Short, 2008; Dai and Li, 2013) in river basins. In measuring access to water as an indicator of sustainability, it is important to consider what sorts of formal and informal laws prevail in a given context. In the River Nyando Basin, property rights and land tenure significantly influence access to water. In communally owned land, access to water sources is better compared to private property where formal laws tend to prevail and anyone who trespasses can be prosecuted. In communal land settings, the use of customary laws is commonly used to negotiate access to water. In recent years, communal water supply systems (e.g. drilling boreholes and wells) have become popular in the River Nyando Basin. Where a water source is located on private land, an agreement, usually in the form of a verbal consent, is drawn between the land owner and the community group involved. In many cases, land owners have failed to honour such verbal agreements by denying group access to the water after a water resource has been developed. This is common in the dry season when competition for water intensifies due to high levels of shortages. To ensure access, community groups who manage communal water systems in private land are being advised to sign formal and written agreements (Memorandum of Understanding- MoU) with private land owners where a particular water resource has been developed. MoUs and level of commitment can be useful indicators for communal access to water in private land.

Technological innovations for water abstraction although commonly used as indicators for improved access to water can cause conflicts between users. Experience in the River Nyando has shown that the installation of water infrastructure for stream flow diversion and spring protection in one community (e.g. household or population) upstream can simultaneously reduce the flow of water to another community downstream (Agol, 2010). In the mid- 2000s, a sugarcane farmer who was located upstream, channelled stream flow onto his land which inhibited water flow downstream to rice irrigators. This led to conflicts between the sugarcane farmer and the rice irrigators. Similar conflicts have been recorded across the River Basin, for example, where spring protection by one community has excluded another from access to water (Agol, 2010). In these examples, it can be observed that action and technology used by one actor can reduce opportunities for another to abstract water. These are key issues which should be analysed carefully when measuring equitable access to water.

5.3.2 Sustaining Good Water Quality

Good water quality is a key indicator of a healthy river system (Chilundo et al., 2008; Schmitter-Soto et al., 2011; Li et al., 2012; Dahm et al., 2014). One of the basic steps for sustaining good water quality in rivers is to safeguard them from the effects of harmful events such as pollution (Ajeagah et al., 2010; Bouraoui and Grizzetti, 2011; Guo et al., 2014). In many river basins across the world, it is common practice to identify potential sources of pollution, their spatial and temporal distributions and risk levels. In the River Nyando knowledge about point and non-point pollution is known and information is clearly documented. The main pollution sources are farm wastes (e.g. use of agro-chemicals in sugar, tea and horticulture), industrial activities (e.g. effluent from sugar and tea processing) and municipal wastes (e.g. Swallow et al., 2003; Raburu and Okeyo-Owuor, 2006).

5.3.2.1 Water Quality Monitoring

Different aspects of water quality are measured in various points across the RNB by use of physical, chemical, biological and epidemiological parameters. Water quality monitoring involves direct and indirect measurements which are usually done by different stakeholders including research institutes, government authorities (e.g. WRMA), NGOs, private establishments (e.g. industries,) and some local community groups such as the Water Resources Users Associations (WRUAs). Direct measurements of chemical water quality is common for phosphorous and nitrogen as well as monitoring physical aspects such as turbidity, conductivity and temperature. Monitoring results have shown that the water quality in the River Nyando is generally poor due to the effects of pollution in the catchment (e.g. Raburu and Okeyo-Owuor, 2006). During flooding, large quantities of sediments are deposited into the River basin containing traces of phosphorous and nitrogen from agricultural and municipal sources. Due to its connection with Lake Victoria, a high level of nutrient loading in the River Nyando tends to cause eutrophication in the Lake. The consequence is colonization of water hyacinth, an invasive plant species which proliferates quite fast under nutrient-rich (eutrophic) conditions (e.g. Odata et al., 2004; Raburu and Okeyo-Owuon, 2006). Measuring levels of nutrient loads from point sources of pollution such as farmlands and municipalities is a more direct way of determining water quality in the river basin. This is in contrast to non-point sources such as run-off from farmlands, leaching from drainage and drifts from spraying of agro-chemicals which quite often are difficult to measure.

Biological indicators (bio-indicators) such as fresh water invertebrates and macrophytes (water lilies) are also used for monitoring water quality in the RNB. Past observations have shown that some open wetlands and parts of the River channel are devoid of these plants and animals especially in areas close to human settlements (Agol, 2010). Due to limited resources, monitoring bio-indicators of plants and animals is not systematic in the River Nyando. In addition the monitoring activities of the stakeholders in the RNB are not coordinated and this is a drawback in understanding trends in water quality in the River Basin.

Since the 1990s there has been a steady growth and expansion of informal settlements (urban and peri-urban) and associated trading centres. Many of these settlements lack proper water and sanitation facilities and quite often, people are seen bathing and washing directly on the River. Large dumps of municipal waste are also common sights around human settlements especially in the lower zone in Ahero, Muhoroni and Nyamasaria towns. Unfortunately, much of these wastes gets deposited directly into the River and associated streams.

5.3.2.2 Awareness Creation And Capacity Building

Government authorities and NGOs are making efforts in awareness creation and capacity building in sustainable waste management and safe water, hygiene and sanitation (WASH). While some have given support in building communal toilets in schools and public establishments (e.g. trading centres), others have funded protection of water sources from contamination (e.g. spring protection). Supporters of such activities have asserted that incidences of water borne diseases (e.g. cholera) have reduced, for example, within households which use water from protected springs. Such claims are based on personal and public testimonials and less attention is often paid to important indicators such as public attitude and awareness of the dangers of water pollution and capacities and willingness of the people in the catchment to change destructive behaviours (e.g. dumping of waste). Socio-cultural acceptability of WASH facilities is also an important sustainability indicator since it has been shown that some modern WASH facilities (flush toilets) are not fully embraced by certain local communities (Agol, 2010). Hence water quality sustainability indicators such as level of acceptability and use of proposed interventions are also important and should be considered.

5.3.2.3 Compliance With Regulatory Measures

In Kenya there are several regulatory measures that protect water quality in river basins and examples include the Water Act (2002), Water Quality Regulation (2006) and the Water Rules (2007). Tool such as Environmental Impact Assessments (EIAs), Polluter Pays Principles complement the regulatory measures in protecting water resources in Kenya. In accordance with these frameworks, any establishment/activity with a likely potential to pollute the environment (including water courses) should implement pollution abatement measures such as installation of waste treatment and recycling plants. They are also required to adopt a protocol of Good Practice including keeping records of all chemical used in their operations and their levels of danger. A systematic monitoring of effluent quality and solid waste generated from the establishment is also a requirement.

Thus the different establishments across the RNB whose operations have the potential to cause harmful effects on quality of the water resources are required by law to implement pollution control measures. Some establishments have complied with legal requirements; others still evade them due to reasons such as lack of technical capacities, ignorance and corruption. Some large-scale industrial and agricultural establishments which operate in the RNB (e.g. the tea and sugar farms and factories) have waste water treatment and recycling plants, solid waste incinerators and constructed wetlands. Some establishments have contractual agreements with third parties such as a certified body or government authority to take the necessary pollution abatement measures such as analysis of effluent in the laboratory and safe disposal of hazardous wastes. But some of the third parties contracted do not comply with legal procedures and tend to take harmful waste from the source and dump it carelessly on the River Basin. Contracting a third party works if they are disciplined, efficient and committed to undertake the assigned tasks. Thus in measuring water quality indicators, it is important to scrutinize the different roles of the key players involved, their capacities and integrity. Knowledge on the destination of waste is necessary when evaluating water quality indicators in a river basin.

Experiences in the RNB have shown that many government authorities who are charged with the enforcement of environmental laws lack financial and technical capacities to do so. As a result, they are unable to monitor all the activities across the River Basin on regular basis. Consequently, knowledge on the level of compliance with regulatory measures for catchment protection in the River Nyando is largely incomplete. In addition, some government officers are not motivated to carry out their duties due to low wages and delays in their payments, inadequate compensation for carrying out fieldwork, etc. This should be taken into consideration when interpreting indicators of water quality for they are important in determining whether the responsible parties are fulfilling their mandate.

5.3.3 Protecting And Conserving Biodiversity

Intensive forest clearance, over-grazing, land fragmentation, sand harvesting and quarrying have caused habitat loss and threats to biodiversity in the Nyando River Basin (Shepherd and Walsh, 2002; Swallow et al., 2003; SANA, 2008). Deforestation is widespread particularly in the Upper catchment where more than ten thousand acres of forest area have been cleared in the past (GoK 2010, API 2010). Agricultural intensification characterized by drainage of several wetlands and use of agro-chemicals have led to loss of natural habitats and biodiversity in the River Nyando Basin.

Current legal measures for protecting and conserving biodiversity in Kenya are encapsulated in the national legislative frameworks such as the Environment Management and Coordination Act (1999), Water Act (2002) and the Forest Act (2005). Gazettement is a common proposed legal measure of these legislative frameworks and aims to conserve biodiversity through physical protection of threatened habitats such as forests.

5.3.3.1 Area Enclosure (Gazettement) To Protect Biodiversity

Some parts of the River Nyando have been officially enclosed through fencing (gazetted) to prevent further degradation. For example, encroachment into the Mau forests in the upper zone and subsequent illegal activities such as logging, charcoal burning and collecting firewood have led to the demarcation of more than 300,000 acres of forestland since the 1930s (API, 2010; GoK, 2010). The need to protect the forests from further degradation through gazettement has led to eviction of local inhabitants from the forests and surrounding areas by the responsible government authorities. Yet indigenous communities such as the Ogiek and the Dorobo have used forests' resources for food, water, fuel wood, grazing and cultural rituals for over one hundred years. Eviction from the Mau forests has caused tension between government authorities and local people. Some local groups have been resistant to eviction from the forests and have protested against gazettement.

The amount of area gazetted is a commonly used indicator for habitat protection. However, this does not guarantee that an area enclosed necessarily safeguards biodiversity in the demarcated areas in the upper zone of the River Nyando. Local tension combined with national politics and corruption have led to mismanagement of forests' resources and further destruction (e.g. timber logging) in some gazetted areas. To minimize local tension, government authorities and civil society organizations have been supporting local community groups in carrying out conservation activities. For example, they have given technical and financial services to some Community Forest Associations (CFAs) to carry out afforestation activities. Some CFAs have successfully restored previously degraded areas through re-planting of indigenous trees. The biggest challenge for such institutional arrangements is that many CFAs are not yet sustainable because they do not have reliable financial resources to sustain conservation activities. In addition, it has been difficult to measure precisely what proportion of previously lost biodiversity of plants and animals has been fully restored following re-afforestation activities.

5.3.3.2 *Riparian Land Protection*

Another notable regulatory measure used for conserving biodiversity in Kenya is the legal requirement that the riparian land along rivers should be set aside and free from intensive cultivation (National Environment Management Authority-NEMA 2011). Similar measures are also introduced in other countries (see discussions on the Chapter on protection of Brazilian riparian areas). Since much of the River Bank of the Nyando has been degraded (e.g. Swallow et al., 2003), setting aside its riparian land is a perquisite for protecting its biodiversity. But many riparian land owners have been hesitant to comply with this regulation due to the high opportunity cost involved (Agol, 2010). For example, faced with limited livelihood choices and climate change uncertainties (e.g. prolonged drought), many riparian land owners prefer to cultivate right up to the River bank where conditions can be favourable (cooler temperatures and moist soils) for farming.

Quantifying and qualifying the positive impacts of riparian land protection with regard to biodiversity improvement in the RNB is not an easy task. Indicators such as compliance level (e.g. proportion of riparian land owners who have set aside land), amount of area set aside and percentage of vegetation cover are commonly used to estimate the level of biodiversity conservation. These indicators are commonly measured by direct observations and surveys. But in the past, some riparian land owners who agreed to set aside changed their minds and continued to cultivate right up to the River Bank. Such changes need to be monitored and noted otherwise the level of compliance may be misrepresented. In addition, only small sections along the

River Bank have been physically protected (e.g. fenced off) and it has been difficult to estimate their basin wide impacts on biodiversity improvement across the RNB.

From early 1990s numerous projects have been implemented across the river Nyando basin to promote agro-forestry and increase biodiversity through soil and water conservation activities. Since then many local groups have established tree nurseries and hundreds of exotic and indigenous trees have been planted across the River Nyando Basin. Out puts such as number of trees planted, survival rates (expressed as percentages) and total area covered in acres, are commonly used indicators for measuring biodiversity levels. Quite often these indicators are estimated by use of surveys, for example, during monitoring and evaluation exercises. However, these indicators cannot reveal indirect benefits such as improved soils and water quality which are aspects of biodiversity.

5.3.3.3 Livelihood Diversification

Unsustainable practices such as intensive cultivation and overgrazing by livestock cause threats to biodiversity in the RNB. To minimize their impacts, several capacity building initiatives have been implemented to promote alternative livelihood activities. Since the last two decades, many local groups have been carrying out alternative livelihoods activities such as keeping modern livestock breeds (e.g. dairy goats and cows), planting of high value trees that can generate cash (e.g. mangos, avocados), bee-keeping, and small businesses to improve household income. For example, planting fruit trees offers multiple ecosystem services including provision of food, timber and fodder for livestock and restoration of soil nutrients through nitrogen-fixation and generation of organic matter. Indicators such as the activity type, number of households or groups involved, level of acceptance, amount of income/ cash generated tend to be used to evaluate livelihood diversification. However, the direct contribution of livelihood diversification for improved biodiversity conservation remains debatable. For example, if a charcoal trader chooses not to clear a portion of his/her land because they have an alternative livelihood source, it may increase biodiversity during that period. However, the farmer may decide to cut the tree to burn charcoal if the income from the alternative livelihood is not enough. Besides, building evidence that biodiversity has been improved requires sufficient technical and financial resources, which unfortunately is a great challenge for most actors operating in the River Nyando Basin.

5.4 Fostering Multi-Stakeholder Participation

Interest in multi-stakeholder participation in river basin management continues to grow (Nhantumbo et al., 2003) and much of its attraction lies in its promise to promote co-management, democratic decision making and subsequent good governance (Blair, 2000; Mohan and Stokke, 2000; Antunes et al., 2009). Among the key strategies used to promote multi-stakeholder participation in the River Nyando Basin are decentralization and public participation.

5.4.1 Decentralization Through Institutional Arrangements

Increasingly, managing river basins is a shared responsibility of governments, the civil society, and private businesses. This reality has led to widespread promotion of decentralization which has seen many governments worldwide, devolving power to stakeholders at different levels and scales to become active partners and co-managers of river basins (Fraser et al., 2006; Arias-Hidalgo et al., 2013; Benson et al., 2014; Gallego-Ayala and Juízo, 2014). The rationale for decentralized system of river governance is to enhance democratic decision-making involving a mix of stakeholders with different socio-economic, cultural, political and professional backgrounds (e.g. Inguane et al., 2014).

In the River Nyando Basin, the quest for decentralization is reflected in the formal institutional arrangements at catchment and sub-catchment scales. Formal institutions such as the Water Resources Management Authority (WRMA) and the Water Resources Users Associations (WRUAs) are legally responsible for the management of water resources at catchment and sub-catchment level respectively (GoK, 2002). While the WRMA is responsible for enforcing water laws and policies, the WRUAs are tasked with fostering cooperation and conflict resolution (GoK, 2002). The work of these formal institutions is complemented by non-governmental (NGOs) and community-based organizations (CBOs). Many CBOs are basically local community groups which are informal in nature. These groups comprise members who collectively participate in a range of activities from afforestation, flood control, spring and river bank protection to alternative livelihoods such as dairy farming, poultry and bee-keeping.

Power devolution is a key perquisite of decentralization and in recognition, efforts have been made in the RNB to devolve power to local communities through the WRUAs and CBOs so that they can take full responsibility in catchment protection and conservation. However, power devolution tends to be inhibited by misalignment of roles and responsibilities of the different institutions operating in the RNB. Notably and in the past, misunderstandings have risen between the WRMA and the WRUAs in performing certain tasks such as monitoring water use and resolving conflicts at the local level. Although the WRUAs are legally mandated to resolve local conflicts, the WRMA have carried out similar tasks in some parts of the RNB in the past. The WRUAs are still largely seen as lacking capacity and confidence and in many cases, they remain powerless spectators especially where tension and conflicts between local water users erupts. Although it is often claimed that power has been devolved to the WRUAs, this is fairly theoretical as the WRMA still makes important decisions at the lower levels.

Since the mid- 1990s the numbers of CBOs have more than doubled in the RNB as a part of an initiative to promote decentralized systems of river basin governance. Through external funding from the donor community[3], some notable efforts continue to be made to empower local community groups through capacity building initiatives. For example, governmental and non-governmental organizations continue to facilitate formal training workshops and farmer field visits for group representatives. Such efforts aim to help local people gain practical skills in catchment protection and livelihood diversification.

The numbers of CBOs, their goals and objectives and magnitude of operations are commonly used indicators to evaluate the extent of decentralization in the RNB. However, empirical evidence has shown that these indicators alone do not necessarily echo power and capacities required to carry out respective roles and responsibilities (Agol, 2010). Many CBOs in the RNB continue to be challenged with limited financing, infrastructure, personnel and technical knowledge. Over-reliance on external funding sources (e.g. international donors and private businesses), causes lack of independence and self-sufficiency for many local community groups and this is critical challenge for sustainability. Community empowerment is largely perceived as an indicator of devolution and decentralization but one which is difficult to measure. This is because many community groups are still not able to make independent and informed decisions on river basin management and livelihoods matters.

5.4.2 Public Participation And Consultation

Across the RNB there have been efforts to engage the public in different processes such as policy formulation, research, monitoring and evaluation. Success in fostering actual public participation has varied significantly depending on the context. Over the last two decades, non-governmental and governmental organizations have been mobilizing the public in the RNB to find practical solutions to catchment degradation through a number of projects such as the Trans-VIC, and Lake Victoria Environment Management Project (LVEMP). These projects have provided important platforms through which local groups (e.g. comprising farmers, pastoralists, businessmen) have participated in flood control activities (e.g. food-for-work initiatives), afforestation and water storage (e.g. construction of water pans). Since the 1990s, thousands of local farmers have participated directly in soil and water conservation activities such as agro-forestry and building terraces within their farms, with the aim to restore the various ecosystem services of the RNB. Farmers' participation tends to be evaluated

3 Financial support from donors such as the European Union, SIDA, CIDA, JICA facilitates formation of CBOs by counting how many of them participate as well as estimations of the amount of land restored. But lack of continuous monitoring has often led to much uncertainty in some areas, with regard to how many farmers have sustained soil and water conservation practices over the years. For example, it is not clear how many farmers continue to practise agro-forestry in their farmlands after the projects through which they were supported exited. It is necessary to evaluate exit strategies within these projects as a measure of sustainability in a river basin such as the Nyando.

The importance of public participation and consultation in river basin management is widely upheld (Jingling et al., 2010) with the assertion that the process promotes democratic decision-making (Lemos and de Oliveira, 2004). The quest for democratic decision-making has led to the endorsement of public participation and consultation across the River Nyando Basin. Many governmental and non-governmental organizations tend to facilitate stakeholders' dialogues (e.g. meetings, workshops and conferences) which are seen to present the public in river basin management. In the recent years, stakeholders' dialogues have become increasingly attractive in the RNB because they are perceived as important platforms through which river basin issues can be resolved. In the RNB many of the local representatives of stakeholder fora are community group leaders or local elites who are expected to consult the public in river basin matters and activities. Among the activities include strategic planning and implementation of new projects, participatory monitoring and evaluation, formulation of policies and plans (e.g. catchment and sub-catchment management plans). But measuring public consultation in policy-making processes, research, monitoring and evaluation has proved difficult in the RNB due to limited resources.

Public representatives can be selfish individuals who are out to fulfil their own interests. In the RNB, some local ellites who act as community representatives tend to seize the various opportunities provided by governmental and non-governmental organizations to enhance their own livelihood activities. For example, some expert farmers or businessmen/women who are already successful have had disproportionate advantage of being supported through river basin projects, for example, when they are offered practical training on livelihood diversification. Although seen as local champions, some individuals have remained the sole beneficiaries of development initiatives in the RNB. It is common practice to provide a daily sitting allowance (DSA) to participants at stakeholders' fora by giving money to those who have attended meetings and workshops. The level of attendance can be quite high in meetings which are facilitated by non-governmental organizations because they pay more money to cover DSA. It has been observed that this kind of incentive tends to attract individuals who neither have interest on the issues of the River Nyando nor the goodwill to negotiate on behalf of the public.

Hence assumptions should not be made that those who attend stakeholder's dialogues as community/public representatives always act on behalf of public interest. On the other hand, some individuals, although seen as community representatives, neither have power nor interest to mediate the interests of the public in the managememt of the River Nyando. For example, some women representatives have been recruited in community group committees, yet levels of their participation in important activities such as formulation of sub-catchment plans are quite often very low. Empirical studies from Honduras and Bolivia have shown that where female representation on local initiatives was not mandated, women participated poorly in decision making processes (Blair, 2000).

Public participation in the RNB tends to be measured and evaluated by use of meagre numbers (e.g. number of workshop attendees) without considering issues of commitment and dedication. Empirical studies have shown that multi-stakeholders' participation tends to be evaluated by counting numbers of attendees without paying much attention to the quality of their participation, discussions and associated outcomes (Manzungu, 2002; Marimbe and Manzungu, 2003; Chikozho, 2005). This is a major drawback for sustainable river basin management.

5.5 Lessons Learned And Conclusion

Sustaining adequate water supplies in river basins to meet present and future socio-economic and environmental objectives is a perquisite for sustainable development. In river basins where there are multiple water sources and users, it is important to gather baseline information on their productivity, types and level of use in order to determine their sustainability. However, due to limited capacities of responsible authorities and actors,, many river basin managers such as in the River Nyando are challenged with gathering reliable and sufficient information on water resources availability and use. In measuring water quantity and quality, attention tends to be paid to bio-physical parameters (e.g. volumes and chemical composition) without putting much consideration into socio-economic dimensions such as human population density and rate of growth in a given area, poverty levels, agricultural and industrial activities, levels of sanitation services and extent of use. Yet these parameters are important for understanding water quality dynamics and patterns of use.

Sustainable management of river basins requires an integrated approach involving a myriad of actors with different roles and responsibilities. In measuring sustainability, it is therefore necessary to know which stakeholders have the most/least influence on river basin sustenance. This requires an understanding of their activities and priorities which in many cases tend to be diverse, as well as their attitudes. Success in bringing positive changes is dependent on the technical and financial capacities of river basin actors as well as their efficiency and effectiveness in carrying out their different roles. Both technical and financial capacities of river basin actors can be measured directly and/or indirectly by use of qualitive and quantitative methods. It important to apply multiple methods in measuring capacities and this requires a considerable investment in time and resources.

In promoting sustainable river basin management, the idea is to devolve power to lower level entities (e.g. Water Resources Users Associations) in order to promote a decentralized river basin governance. The role and validity of decentralization of river governance is highly debatable since it is a common practice to measure such systems by counting the number of lower level institutions (e.g. community-based organizations). Even though some government authorities operating in the Nyando have shown a genuine commitment to devolve power to lower level entities such as the WRUAs, these locally based institutions lack much power and subsequent capacities to carry out their roles and responsibilities. Measuring level of power as an indicator of institutional capacity can be challenging since in many cases, power is seldom exercised due to politics and bureaucracy.

At the centre of public participation in river basin management is the issue of representation. In the RNB, it has proved difficult to mediate the interests, priorities and concerns of all the stakeholders especially in pursuit of democratic decision-making. To ensure sustainability, it is vital to consider issues such as dedication and commitment of representatives at the multi-stakeholder platforms.

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6 Sustaining Local Livelihoods Through Coastal Fisheries In Kenya

George N. Morara, Farida Hassan and Melckzedeck K. Osore

6.1 Introduction

As the human population is rapidly growing along the Kenyan coast, demand for food security and local livelihoods for the coastal inhabitants will continue increasing, thus accelerate the pressure on coastal and marine resources including the fisheries. This situation is relevant to debate on sustainable development which is the present concern across the globe (WCED, 1987; Pezzey, 1989). However, context specific choices have to be made on what would constitute indicators of sustainability, as this subject is still argued differently across the disciplines such as, ecology, economics, sociology, development and political studies (De Wit and Blignaut, 2000). In this chapter, attempt is made to highlight some of the approaches currently used in Kenya to sustain local livelihoods through coastal fisheries. Some of the key indicators which have been used to monitor sustainability in fisheries management and livelihood development along the coast are discussed. The perspective of capital theory approach to sustainable development is reviewed and found useful for consideration in future selection of sustainability indicators.

6.1.1 Overview Of Global Fisheries Status

In the ancient years of global fisheries development, traditional fishing processes were somewhat limited in technology, geographic expansion and target species. In that situation, coupled with relatively low human populations, it was possible to find large proportions of naturally protected fish populations with the majority distributed outside the targeted fishing areas. However, with industrialization of fishing processes and increased efficiency in capturing target species, a steady growth in global fisheries production was observed between 1950 and the mid-1990s, but slowed in the later years of the 90s according to the records with Food and Agriculture Organization (FAO, 2012). This fact may be attributed to enhanced depletion effect on the natural fisheries resources. On the other hand, high food demands of the world's population, which is currently estimated at about one billion people (FAO, 2012) has necessitated intensified effort to increase fish production from aquaculture systems.

The recent changes in world fish production and utilization trends presented in Fig. 1 and 2 respectively may imply the pressing demand for food and nutrition needs of the fast growing world human population, against finite natural resources,

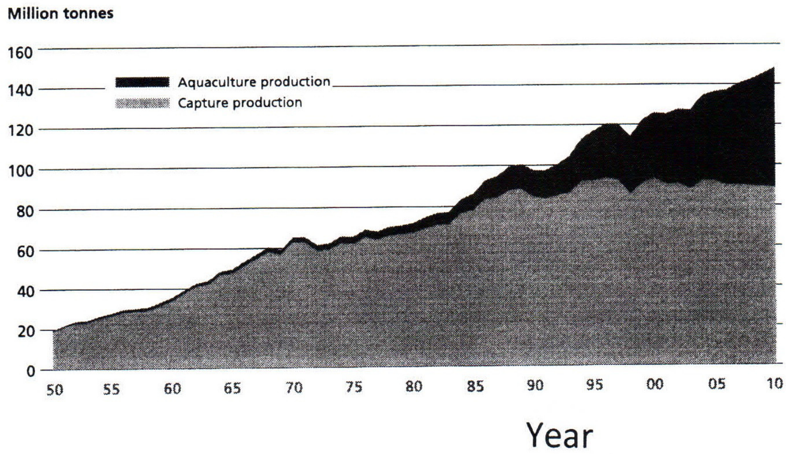

Figure 1: World capture fisheries and aquaculture production (FAO, 2012).

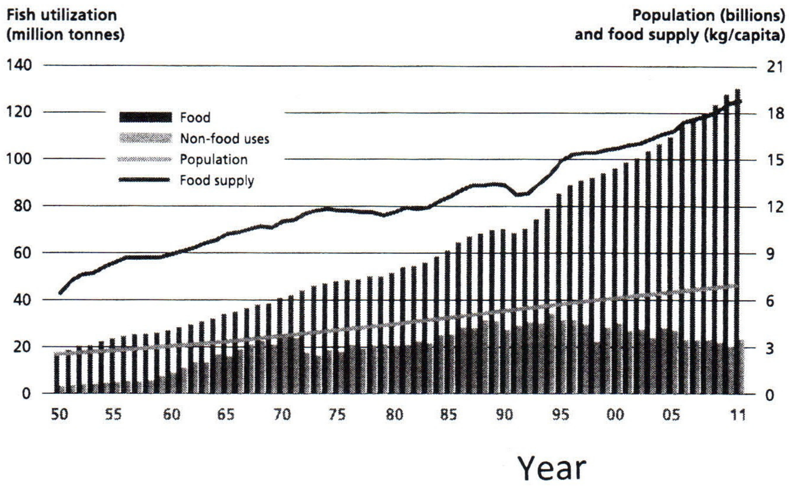

Figure 2: World fish utilization and supply (FAO, 2012).

need concerted efforts by the global community in addressing the sustainability of fisheries from the perspective of both production and local livelihoods. Today, sustainable fisheries management has become a common discourse in many fisheries governance systems. Theoretical fisheries models have been constructed and applied in different contexts to help in designing fisheries management programmes for both underexploited and overexploited fisheries. A classic example of single-species fish population models which has been widely used as an indicator of sustainability in fisheries science and management is the *maximum sustainable yield (MSY)*, described

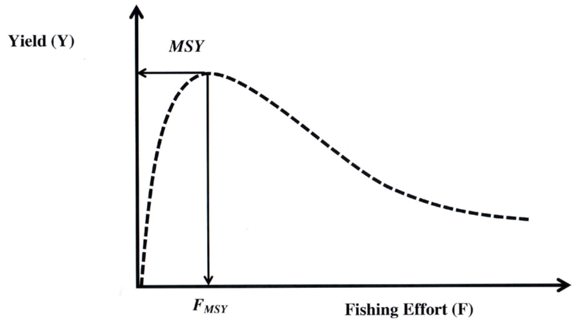

Figure 3: Illustration of a typical MSY model for fish stock assessment and fisheries management (Adapted from Sparre and Venema 1992).

by Pauly et al. (2002) and Mace (2001). Fundamentally, this is a *surplus production model* which underpins the notion of sustainable harvesting and whose objective is to encourage managers to maintain a fish population size at the point of maximum growth rate. The model gives a perspective of the entire fish stock, fishing effort and the total yield obtained during a specified fishing period. Thus, it assumes that a fishery may be maintained at maximum growth rate by harvesting fish individuals recruited into the population while allowing indefinite reproduction. The fishing effort at which the maximum sustainable yield is achieved is called *optimal fishing effort* (F_{MSY})

Fig. 3 illustrates a typical *MSY* and optimal fishing effort (F_{MSY}) concept as applied in fish stock assessment and fisheries management approaches (Sparre and Venema, 1992).

Calculation of *MSY* takes into consideration the fishing pressure or fishing rate expressed as fishing mortality rate (*F*) and the total catch or yield which is expressed as (Y) within a fishing period. Hence the input data for *MSY* are:

1. F (i) = (effort in year i, i = 1, 2,……………,n)
2. Y/f = yield (catch in weight) per unit of effort in year i.

Sparres and Venema (1992) discuss that the Y/f may be derived from the yield of a fishery, say Y(i) of year i and the corresponding effort, f(i). The MSY is thus derived from a linear model suggested by Schaefer (1954) in which the yield per unit of effort (Y/f) is expressed as a function of the fishing effort (f) as below:

$$Y(i)/f(i) = a + b*f(i) \text{ if } f(i) \leq -a/b$$

In the above equation, **a** and **b** are parameters determined from a regression of the yield per unit of effort (Y/f) against the corresponding effort. Thus, **b** represents the slope of the equation while **a** is the y-intercept of the slope. Implicitly, the values of

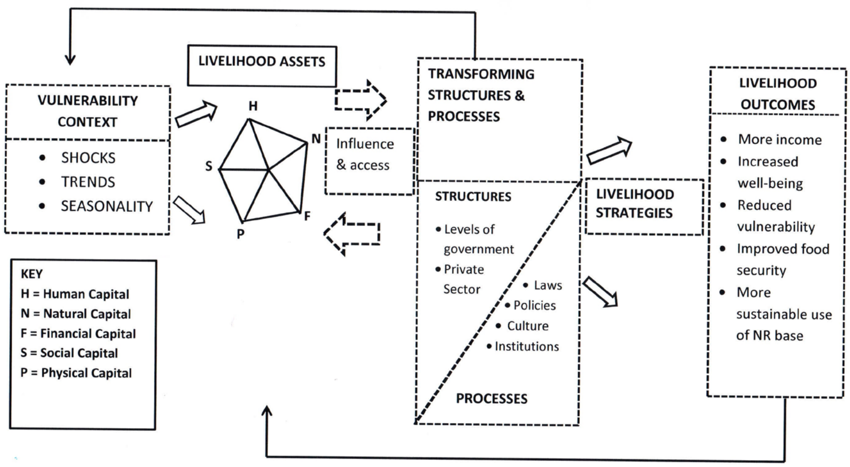

Figure 4: The Sustainability Rural Livelihood Framework (Adapted from Carney, 1998).

these parameters can be calculated when the amount of fishing effort in a fishery and the amount of yield at various levels of fishing efforts are known. Apart from the Schaefer model, other empirical formulas have been developed to provide a rough estimate of *MSY* in various instances where challenges of obtaining fisheries data exist. Some example of such formulas include the Gulland's formula (Gulland, 1971); Cadima's formula (Troadec, 1977); and other models by Garcia et al (1989).

Essentially, estimating the *MSY* values for exploited fisheries requires substantial amount of human capacity and time to ensure collection of accurate data. Therefore many government agencies mandated to manage fisheries resources are often trapped in rigorous assignments of collecting data and analysing the *MSY* values without adequate regard of the need to monitor and evaluate the relevance of such data or other external factors which often undermine fisheries restoration and sustainability (Mace, 2001). Despite the existing wide knowledge on threats to the global fisheries, and the effort made towards restoration of fishery resources e.g. the Earth Summit of Johannesburg, overfishing still persists in many part of the world's fishing areas (Rosenberg, 2003). In fact, this may explain the basis on which the *MSY* concept has been criticized for being less robust in fostering a holistic fisheries management approach and blamed for drastic collapse of fisheries in many regions across the world (Larkin, 1977; Walters and Maguire, 1996). It is largely theoretical and ignores in its computation many other factors that influence fisheries. For example, factors such as environmental degradation, age and size of the species in question as well as the effect of by-catch tend to be discounted. As a result, many fisheries governance systems are somewhat sceptical about reliability of using a computerized *MSY* as a sustainability indicator.

6.1.2 Paradigm Shift In Fisheries Management And Sustainability Indicators

Since the early 1990s, new paradigms have evolved focusing on Ecosystem Based Management (EBM) systems for fisheries and forestry and other natural resources (Slocombe, 1993; Slocombe, 1998). The concept of ecosystem-based management takes into consideration an array of the possible interactions within an ecosystem. It has the feasibility of integrating both anthropogenic and ecological factors into a management framework rather than focussing on single species or ecosystem services in isolation (Levin and Lubchenco, 2008). For instance, on the global scale, it is undisputed that the major threats to most of the coastal and marine fisheries are not only rooted in overfishing, but also the disruption of their marine ecosystems due to chemical pollution, nutrient enrichment and climate change among other factors. Therefore, under such circumstances, a realistic indicator of sustainable fisheries should strive to interrogate both ecological and anthropogenic roles in fisheries dynamics. This is consistent with the recommendations of Pauly et al (2002) that since single species assessment models have not served fisheries managers well, they should be complemented with elements drawn from the species ecology and lessons learnt from efforts of limiting fish mortality.

Notably, some advances have been made towards the implementation of EBM. In the context of fisheries management, the concept of ecosystem-based fisheries management (EBFM) has been discussed widely and recommended for adoption (Allison and Ellis, 2001; Christie et al., 2007; Metcalf et al., 2009). Under the EBM framework, it is acknowledged that all fishing activities have inherent and immense impacts on fish abundance, trophic structure of aquatic ecosystems, biodiversity status and most importantly, the long-standing human interactions (Christie et al., 2007). Hence, sustaining fisheries is given equal importance as the local livelihoods which depend on them. This is because both are linked to food security, employment opportunities and economic development at local, regional and national levels.

Contemporary fisheries management embraces EBFM by integrating fisheries with livelihood and sustainability issues. While such efforts are more advanced in the developed countries, they are still in their early stages in third world and developing countries. For example, in Africa today, several donor supported projects are developed with the aim of striking a balance between local livelihoods and fisheries conservation strategies. This reflects a departure from the outdated and top-bottom enforcement of legislative measures which largely disregard the interests of people who are often affected directly or indirectly by the same measures. Involvement of 'local actors' or multiple stakeholders in fisheries management can create a win-win situation and inculcate democratic decision making processes in the fisheries governance systems. An example from West Africa is the case of the Sustainable Fisheries Livelihood Programme in which 25 countries in the region were involved in implementing the concept of Sustainable Livelihood Approach (SLA) to fisheries management (Allison and Ellis, 2001; Allison and Horemans, 2006). The concept was successful in aligning fisheries policies with poverty reduction initiatives, and to a large extent helped in identifying ways of reducing fishing pressure on fully exploited or over-exploited fisheries.

Lessons and experience from the SLA may help to understand the vulnerability of small-scale fisheries and outstanding threats to local livelihoods of artisanal fishers. Vulnerability in this context refers to the high degree of exposure to a devastating shock, risk, stress or food insecurity that may threaten life (Chambers, 1989; Davies, 1996; Allison and Ellis 2001). The factors that may expose livelihoods to such risks include, but are not limited to, environmental stress such as climate change, pollution and habitat degradation, over-exploitation of a resource, and inappropriate legislations. Communities who become resilient to such circumstances are those whose livelihoods have been diversified rather than depending solely on natural resources (Allison and Ellis, 2001).

Knutsson (2006) argues in favour of the SLA applications as appropriate and trans-disciplinary on the basis that this approach is produced, disseminated and applied across the borders of research, policy and practice in resource management. Furthermore, this being a newly emerging field with a significant focus on the human ecology, the approach would be more practical and useful if tools for identification and evaluation of sustainability indicators were provided. Knutsson (2006) thus endeavours to provide this missing knowledge by assessing a set of criteria for integrative approaches to sustainable development problems in the context of SLA.

Other approaches similar to SLA and widely discussed in literature include the Sustainable Rural Livelihood (SRL) a framework which was originally provided in Carney (1998). The SRL was later elaborated in other context specific articles that have attested the linkages between natural resources and livelihoods (Scoones, 1998; Carney et al. 1999). The work of Carney (1998) provides a framework for researchers and managers to interrogate the many complex options of livelihoods development and their interaction with the environmental, economic and political processes. The framework identifies five typical assets of livelihood which may be influenced to trigger a situation of livelihood vulnerability or resilience with positive outcomes. The five assets are: the human capital; natural capital; financial capital; social capital; and physical capital (Fig.4).

From literature review, it can be inferred that these assets are implicitly embodied in the concept Capital Theory Approach (CTA) to sustainable development, which is discussed extensively by Stern (1997) and De Wit and Blignaut (2000). The literature shows that, the application of CTA still requires policy makers to have clear understanding of what constitutes the man-made and natural capital stocks in the context of sustainable development. However, emerging from the discourse objectives and indicators of sustainable development can be anchored on either the environmental or ecological point of view. In the case of environmental approach, substitutability between man-made and natural capital is favoured with the assumption that the overall capital stock will be maintained over a period of time. This approach is also described as the *weak sustainability* (Stern, 1997; De Wit and Blignaut, 2000). The contrast to this is the ecological approach or *strong sustainability,* which argues for complementarity of the man-made and natural capital with assumptions that specific capital stocks will be maintained intact over time (Blignaut and De Wit, 1999).

De Wit and Blignaut (2000) contend that the main point of argument with regard to the CTA is on the vagueness of the concept and its inadequacy in accounting for the main elements of sustainable development. However, while both the environmental economic and ecological economic approaches can be applied to maintain the capital stock over time, there is concurrence in literature that these two approaches depart with respect to the degree of capital stock substitutability for each other (Costanza, 1991; Toman, 1994; Daly, 1996; Stern, 1997). It is in this context that Stern, 1997 elaborates the major subcategories of “capital” to include aspects such as natural, manufactured, human, moral, ethical, cultural and institutional elements.

6.1.3 Sustainability In Fisheries And Livelihoods Contexts

From the previous sections, we have used the terms ‘fisheries’, ‘livelihood’ and ‘sustainability’ as the subject matter of discussion. In the following sections we attempt to contextualize the meaning of these terms and elaborate their perspectives in the coast region of Kenya, East Africa.

The term ´fisheries´ is widely applied in literature to refer to the consumptive harvesting activity of aquatic organisms from either an artificial or their natural water systems for commercial, or subsistence purposes. According to Fletcher et al. (2002), this refers to the entities engaged in raising or harvesting fish which is determined by some authority to be a fishery. Therefore, we can simplify the definition of fisheries as the union between aquatic organisms and humans in which the inherent features are the aquatic environment in a geographical region, fish populations living in that environment, human interaction with the fish population and the legal rights to engage in utilizing fish resources in specified waters or regions. In FAO (1995), fisheries management is defined as the integrated process of information gathering, analysis, planning, decision making, allocation of resources and formulation and enforcement of fishery regulations by which the fisheries management authority controls the present and future behaviours of the interested parties in the fishery, in order to ensure the continued productivity of the living resources.

In this discourse we use the term ‘livelihood’ to imply a form of support required for living or survival. From the Oxford Dictionary of English (2010), it refers to a set of activities, involving securing necessities of life (water, food, fodder, medicine, shelter, clothing) and the capacity to acquire these necessities working either individually or as a group by using endowments (both human and material) for meeting the requirements of the person and his/her household on a sustainable basis with dignity. Such set of activities are usually carried out repeatedly. According to the International Federation of Red Cross and Red Cross Crescents Societies (IFRC), a livelihood comprises people's capabilities, assets and activities required for generating income and securing means of living (IFRC, 2010). In this regards, considerable focus is given to the people's endowment and interaction with the available resources or opportunities such as agriculture, fisheries, forestry, mining, tourism among others. A similar definition is discussed by Ellis (2000) and Allison and Ellis (2001) who describe a livelihood as comprising three aspects namely: *assets* (natural, physical, human, financial and social), *activities*, and *access* to assets that is mediated by institutions and social relations.

The term sustainability is commonly used in the context of development where socio-economic and environmental objectives are highly considered. The World Commission on Environment and Development (WCED) defines sustainable development as "*development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs*" (WCED 1987: pp 15). In 1988 FAO Council defined sustainability in a broader perspective to include the management and conservation of the natural resource base, and orientation of technological and institutional change to ensure the attainment of continued satisfaction of human needs for both the present and coming generations. Such sustainable development in key sectors such as agriculture, forestry and fisheries, conserves (land) water, plants and (animal) genetic resources, besides being environmentally non-degrading, technologically appropriate, economically viable and socially acceptable (FAO, 1989). The latter definition implies that considerations are given to the extent of welfare optimization from finite natural resource base with minimal resource degradation and regulated exploitation regime over time. However, it is worthwhile to note that sustainability elements can also be applied in other sectors without exploitable natural resource methods of industrial production, programmatic ideas or even governance structures.

Chambers and Conway (1991, pp:6) have elaborated the concept of sustainable rural livelihoods, that: "*A livelihood comprises the capabilities, assets (stores, resources, claims and access) and activities required for a means of living: a livelihood is sustainable which can cope with and recover from stress and shocks, maintain or enhance its capabilities and assets, and provide sustainable livelihood opportunities for the next generation; and which contributes net benefits to other livelihoods at the local and global levels and in the short and long term.*"

This chapter highlights some of the approaches that have been applied for sustainable management of coastal fisheries and livelihoods in Kenya. It focuses on both state and community led initiatives, especially the marine protected areas (MPAs), community conserved areas (CCAs) and co-management approach through beach management units (MBUs).

6.2 Kenya's Marine Ecosystems And Resource Dependency

Kenya's coastline is about 650 km long covering an area of about 83,603 km^2. The coastal area is endowed with unique ecosystems with rich natural resources including marine fish, coral reefs, seagrass beds, mangrove forest and diverse cultural heritage. Almost the entire part of the coastline is covered by a fringing reef. In these areas, there are abundant populations of herbivorous fish species which maintain ecosystem balance by grazing on algae, a function which enables the corals to flourish. These ecosystems, especially the seagrass beds and mangrove forests are particularly vulnerable to overexploitation, destructive use, and the impacts associated with climate change. The high species diversity and richness, including over 250 fish species, make the marine ecosystems in Kenya areas of high protection interest. Currently, there are six designated marine protected areas (MPA) along the coast which have been designated as national parks.

The coastal region of Kenya has about 3.3 million human inhabitants. The economy of these coastal communities depends mainly on artisanal fishing, small-scale farming, livestock husbandry, subsistence forestry and small-scale businesses. Although the coastal and marine resources provide many opportunities for economic growth and reduction of poverty, their unsustainable management has contributed to degradation of the resource base as a result of high human population pressure.

6.2.1 Coastal Fisheries And Livelihoods In Kenya

Kenya's fishery sector generally contributes about 4.7 % of the country's Gross Domestic Product (GDP). This explains the motivation of the Government to strengthen the sector through promotion of sustainable fisheries and aquaculture for improved food security and livelihood of the dependent local communities. The endowment of the coastal region with a rich fisheries resource presents myriad opportunities for economic and social transformation of the local people. Apart from this, fish provides an important source of protein for the coastal human populations.

However, the Kenya coastal fishery sector is largely at artisan level and limited to the reef habitats which are currently under fishing pressure (Munga et al., 2013). The richest inshore marine fishing grounds are mostly located in Lamu Archipelago, the Ungwana Bay, North Kenya Bank, and Malindi Bank (Fig 5). The estuarine systems of the two major Kenyan rivers (Rivers Tana and Sabaki) which form the Malindi – Ungwana Bay fishing grounds, are equally very productive and support local livelihoods in the region. In the latter, commercial prawn trawling has been carried out since the 1970's. Recent surveys by Kenya Marine and Fisheries Research Institute (KMFRI) show that although the fishery resource in the Malindi-Ungwana Bay is under exploited, the whole fishery system is associated with high conflicts due to destructive effect of trawlers on the traditional fishing gears; competition for common

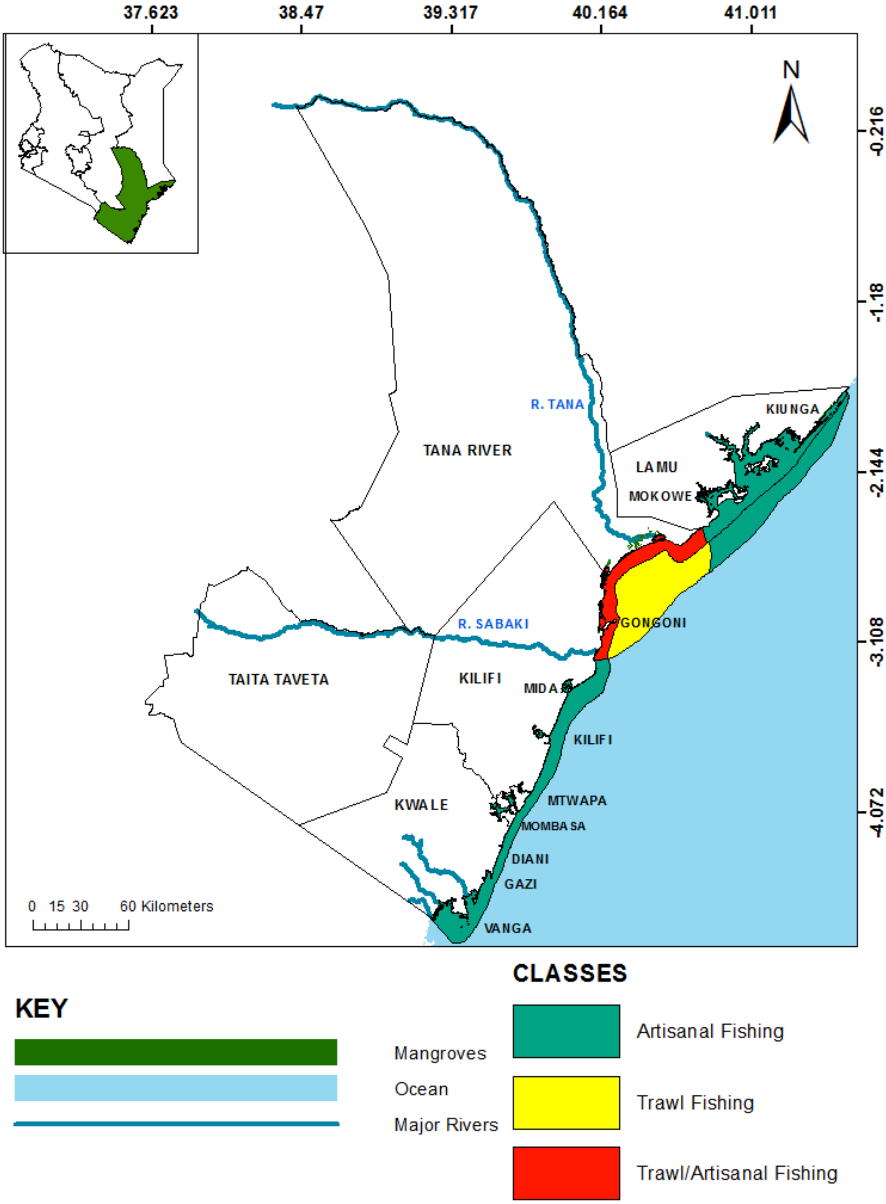

Figure 5: A map of the Kenya coastline showing the area of artisanal and trawl fisheries (Drawn by Noah Ng'isiange, KMFRI).

resources; by-catch wastage and mistrust amongst the fisher groups (Fulanda, 2003; Munga et al., 2013).

Generally, Lamu Archipelago has the most productive coastal fishing area in Kenya with abundant fish populations. Its remoteness and the proximity to insecurity zone near Somalia present many logistical challenging issues, thus provide temporary protection of fish populations in the area. On the other hand a marine reserve exists in Kiunga where co-management approach has been established and works well for fishery.

Statistics from the State Department of Fisheries (a Government agency) indicate that the coastal and marine fisheries sector accounts for about 10,000 tonnes (10%) of fish of the total annual fish landings and employed approximately 13,700 fishermen by 2012 (Government of Kenya, 2012). The fishery thus supports about 60,000 people, who live near the key fish landing beaches, for their income generation and food security.

While Kenya's coastal and marine fisheries have the potential for high offshore fisheries production, the present fisheries is largely based on a small number of demersal fish species caught by artisanal fishers who mostly operate between the shoreline and the reef. A study conducted on the species composition in landings of this artisanal fishery by Wakwabi et al. (2003) revealed domination by demersal species in the catches and trailed by echinoderms, which constitute 42% and 4% respectively (Table 1). The study observes that the most common fish species in the landings are: the rabbit fish (*Siganus sutor*), variegated emperor (*Lethrinus variegatus*), dash-dot goat fish (*Parupeneus barberinus*), parrot fish (*Sergeant majors*), sweetlips, scavenger, red snapper (*Lutjanus argentimaculatus*), rock cod (*Plectropomus aneolatus*), thumbprint emperor (*Lethrinus harak*), yellow goat fish (*Parupeneus barberinus*), peacock rock cod (*Cephalopholis argus*), pick handle barracuda (*Sphyraena jello*), sailfish and black tip kingfish. Although not all fish species in the coastal strip of Kenya have been identified, significant efforts have been made by FAO and the KMFRI staff and other fisheries scientists, to document most of the species exploited for commercial, recreational or subsistence uses. Part of this work has been documented by various authors (e.g. Glaesel, 1997; Mohammed, 2002; Anam and Mostarda, 2012).

The aerial densities of fishermen in the Kenya's coastal reef have been estimated to range from 7-13 fishers per square kilometre (McClanahan and Kaunda-Arara, 1996). These statistics may be outdated and no recent studies have been conducted in this direction. Hence it is highly likely that the density may have increased as a result of the rapid population growth in the coastal region which has inadvertently mounted pressure on the reef fishery. It is also apparent that the current coastal fishing activities are rather chaotic and indiscriminate in species capture (Fondo, 2004). A random assessment of the fisheries from two fishing areas along the coast of Kenya conducted in 14 designated sites, 11 in Lamu (North coast) and 3 in Vanga (South coast) indicate a wide range of fishing methods employed by fishermen (Table 2). Gill nets, shark nets and beach seines are the most frequently used types of fishing gear.

Table 1: Composition of coastal artisanal fishery in Kenya.

Fishery category	% Composition
Demersal species	42
Pelagic species	18
Crustaceans	12
Sharks, Rays & similar species	18
Molluscs and Echinoderms	4
Deep sea and game fish species	6

Table 2: Method of fishing commonly used in artisanal coastal fishery in Kenya.

Fishery category	% Occurrence
Beach seines	16
Diving and fishing guns	3
Gill nets	26
Hook and line	13
Ring nets	7
Shark nets	26
Traps	6
Other traditional methods	3

Commercial bottom trawl fishery is also done in 5-200 nautical miles (nM) waters as opposed to the 0-5 nM for artisanal fishery. The various gears being used have their specific merits and demerits for sustainability of the fishery. This largely depends on the sizes of the gear, the mode of operation and the specific grounds of fishing. Ring nets, for instance have been discouraged for their indiscriminate fishing effect on the fishery besides causing destruction of fishing habitats. Trawling on the other hand has serious fisheries consequences due to the high proportion of by-catch composition in the fish caught (Fulanda, 2003; Munga et al. 2013). Therefore, it is not surprising that a trawl ban was imposed in 2006 on the Malindi–Ungwana Bay due to declining fish catches and resource use conflicts that threatened livelihoods of the dependent fisher communities.

Plate 1: Artisanal coastal fishing activity using the beach seining method (Photo: Stephen Mwakiti, KMFRI).

6.3 Approaches To Sustainable Livelihoods And Coastal Fisheries In Kenya

The foregoing sections emphasize the coastal fisheries as an important resource yet it is either under-utilized, as in the case of offshore fisheries or overexploited in current situation of inshore fishery in Kenya. It is apparent that attempts to implement a sustainable livelihood approach and sustainable rural livelihood methods have not been effective in the coastal region of Kenya. Interest in participatory approaches is growing with the objectives of empowering the local communities to own and manage their fisheries. In this section, we present and discuss three practical approaches that are being used in sustaining coastal fisheries in Kenya in the light of their strength and weaknesses.

6.3.1 Establishment of Marine Protected Areas

Marine protected areas (MPAs) are highly recommended across the globe for their effectiveness in conservation and management of coastal and marine resources. In the tropical coral reef ecosystems, MPAs have served as an effective tool for maintaining biological diversity and species abundance as well as fisheries management (Kelleher, 1999; Gell and Roberts, 2003). For over four decades, Kenya has used MPAs as a conservation tool, thus established nine national and marine parks and

Table 3: Marine Protected Areas in Kenya (Nyawira et al, 2003).

Site name	Designation	Size (Km^2)	Year established
Kiunga	MNaR, UNESCO Biosphere Reserve	600	1980
Malindi	MNP	6.3	1968
Malindi-Watamu	UNESCO Biosphere Reserve	177	1968
Watamu	MNP	32	1968
Mombasa	MNP	10	1986
Mombasa	MNaR	200	1986
Diani	MNaR	75	1993
Kisite	MNP	28	1978
Mpunguti	MNaR	11	1978

MNP: Marine National Park
MNaR: Marine National Reserve

reserves under the management of Kenya Wildlife Service (Table 3). According to Nyawira et al (2003, pp: 3), the mission of MPAs in Kenya is ‘to protect and conserve the marine and coastal biodiversity and the related eco-tones for posterity in order to enhance regeneration and ecological balance of coral reef, seagrass beds, sand beaches, to promote sustainable development, and to promote scientific research, education, recreation, and any other resource utilization’. Three main goals are also highlighted in the Kenyan context as: (1) preservation and conservation of marine biodiversity for posterity, (2) provision of ecologically sustainable use of the marine resources for cultural and economic benefits and (3) promotion of applied research for educational awareness programmes, for community participation, and for capacity building. Therefore the MPA approach has great potential of increasing biodiversity, promotion of underwater tourism, protection of the coastline besides improving livelihoods through subsistence (marine food consumption) and commercial reef fisheries. In addition, MPAs can enhance social capital for local communities adjacent to the MPA areas. The merits of MPA approach is that it helps to control human activities, protects fish breeding areas and improves ecosystem services in the protected areas. Studies by Watson el al. (1996) and McClanahan et al. (2007) have shown that MPAs are effective in restoring degraded coral reef and fish abundance on the Kenya coast.

The Success of MPA widely depends on the extent of stakeholders’ consultations and engagement. However, traditionally, government led protection of marine parks and reserves is often characterised with restrictive access to resources that is mainly due to the top-down policy orientation. In practice, this approach has some challenges associated with minimal community involvement. McClanahan et al. (2005) observes that most of the MPAs were created after pressure to the government from the tourism industry. Despite the ecological and economic benefits from such MPAs (Francis et al., 2002), these areas remain disputed especially for artisanal fisher communities who feel excluded from their management. It is obvious that conflicts on resource use and some sort of resistance will continue to emerge from adjacent communities dependent on the resources contained in the protected areas (McClanahan et al., 2005; Munga et al., 2010). According to Cinner et al. (2010) the failure of MPA to recognize the multiple and complex social and economic conditions at the planning and implementation processes may be attributable to their opposition by the fisher community.

Therefore, the MPA approach somewhat undermines the principles of sustainable livelihood in different ways. First its top-down nature demands high annual budgets. For example, it requires adequate human resource and equipment to conduct monitoring control and surveillance (MCS) in MPAs for policy enforcement and compliance. Besides, the MPA system is time consuming since regular reports are required on the state of such protected area. The second demerit of the system is its tendency to increase levels of illegal activities, especially where such areas are not properly zoned and lack clear boundaries. Restriction of access to a resource often bears a negative connotation as it deprives the dependants of their perceived rights and intrinsic livelihood asset thus trigger situations of vulnerability, locally considered as marginalization.

Sustainability indicators for consideration in MPAs should include among other aspects; the extent of biodiversity richness; levels of awareness and illegal activities in the area; income generation from tourism activities and extent of the ecosystem services and accruing benefits to the adjacent communities.

6.3.2 Establishment of Community Conserved Areas (CCA)

Over the last two decades, serious threats have been posed to the reef biodiversity and livelihoods of coastal people. These are mainly due to over fishing and use of unsustainable methods. Increasing coastal human population and limited economic opportunities have exacerbated the situation. In response to the challenges facing the top-down government-led MPAs with regard to policy enforcement and compliance, a hybrid management approach has evolved in the form of community based marine protected areas. This approach is participatory in design and bestows the controls to community leadership while the responsible state agency provides oversight roles.

Community established MPAs, also referred as community conserved areas (CCAs), are somehow new in Kenya, having evolved over the last one decade (Maina et al., 2011). This concept is similar to the Locally Marine Managed Areas (LMMAs) which has its origin from the Pacific in Fiji where it has existed since the 1990s (Govan et al., 2008). For the CCAs to be effective, the most important requirement is the consistency in target community engagement for optimum sharing of information and learning. In Kenya, the roles of communities in the conservation objectives are supposed to be clarified and where appropriate enshrined in national policies or legislative framework. Examples of these include the Beach Management Regulation of 2007 for

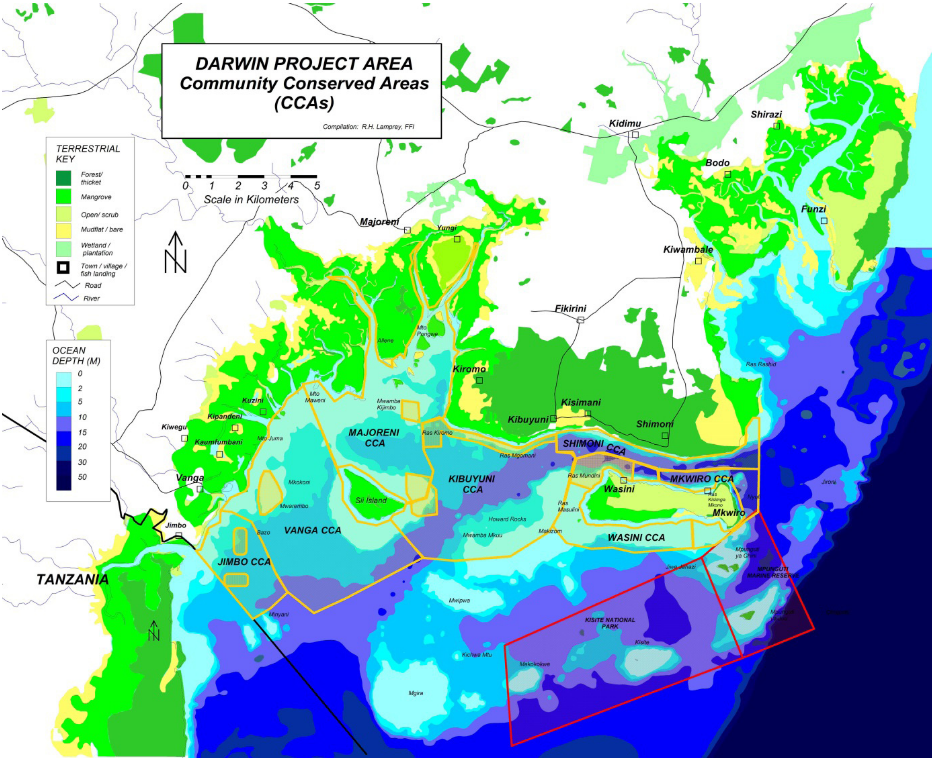

Figure 6: South Coast Kenya Project Area Bathymetry and Community Conserved Area (Source: Flora and Fauna International – FFI).

fisheries management; the Wildlife Conservation and Management Act of 2009 and the Forestry Act of 2005 for mangrove areas protection. The merits of this system include sharing of the operational cost, high level of community learning, guaranteed access to the resources, and social systems development. Tens of community conserved areas (CCA) have been identified and the process supported for community participation along the coast region. An established reference case is the CCA between Shimoni and Vanga of Kenya's south coast which involves about 7 community groups made up of the artisanal fishers and fish traders. It is estimated that 12,400 hectares of the marine area is under community management as illustrated in Fig. 6 (Lamprey and Mushage, 2011; Brett, 2011).

Although the CCA approach to conservation of coastal and marine fisheries is still new along the Kenya coastal region, positive impacts have been made by some community based organizations such as Kuruwitu Conservation Welfare Association (KCWA) in the north coast, where the first community conserved marine area in Kenya was initiated in 2005 by a fisher community concerned about the declining fish catches and coral cover in their inshore fishing grounds (*Pers Obs*). The Community,

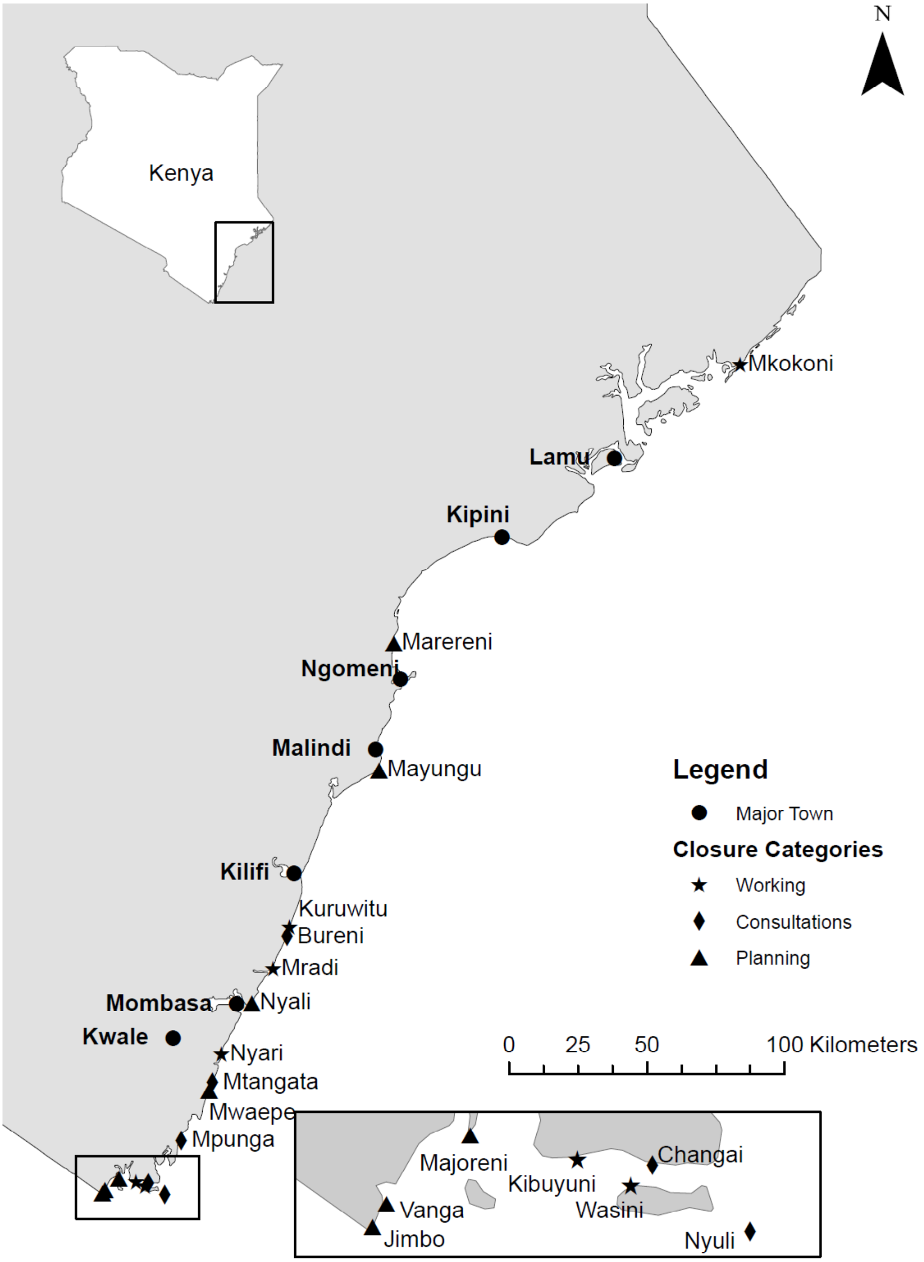

Figure 7: Distribution of Community Conserved Areas locally known as '*Tengefu*' and their status of establishment along the Kenya coast (Source: Wildlife Conservation Society - WCS).

supported by the East African Wildlife Society (EAWS), established a small area of “no fishing zone” of about 2 Km^2 around the fish landing site. Interestingly, this was regularly guarded and monitored by the community members themselves. The aim of the support was to help improve the governance system by devolving the control and powers to the local communities for them to own all regulatory processes, thus enhancing their commitment to the sustainable use of the fisheries.

The approach also provided opportunities for forming socially cohesive leadership structures which permit learning and practicing together. Research has shown that this initiative of community designated small “no fishing zones”, locally known as *tengefu* in Swahili language (meaning spared or set aside), has positive impact in sustaining both marine conservation and livelihood strategies. Particularly, *tengefus* are critical in making fisheries management institutions more flexible and adoptive (McClanahan and Cinner, 2012). Fig. 7 shows the current distribution of *tengefu* initiatives along the Kenya coast and their status.

The process of establishing the *tengefu* involves three key steps. First, is the series of consultative meetings between members of a fisher community, conservation groups (usually non-governmental organizations) and the State Department of Fisheries (SDF). The need and objectives of conserving the identified site are discussed and agreed upon by these stakeholders. This is followed by feedback to the larger group of stakeholders including the wider fisher community, boat operators, fish traders and local residential hoteliers. This step is especially critical for all the stakeholders to understand the conservation objectives desired and attain consensus. Finally, the process results in the formation of a community management plan with mechanisms of its participatory monitoring and evaluation for successes. Therefore it can be inferred here that some of the key sustainability indicators of CCAs are the establishment of strong institutional framework for resource management, increased property rights, enhanced law enforcement and improved social capital for coastal resources dependent communities.

6.3.3 Establishment of Co-management through Beach Management Units

The top down management of fisheries resource in Kenya has evolved over the last decade to become a shared responsibility of the Government and Fishers communities. In 2006, the Government of Kenya legislated into law the formation of community led fisheries management systems that would see the formation of beach management units (BMUs) for sustainable fisheries (Kundu et al, 2010). BMUs work more or less like the CCAs but with their scope and mandates rather limited to fisheries. The aim of BMUs is to enhance management effectiveness of fisheries resource through a co-management approach. In the coast region, over 140 BMUs have been established and work very closely with the SDF in regulating fishing activities (Fisheries Department, 2009). The BMUs have democratically elected leadership structure and operate under a defined area. The underlying principle of the BMUs, as in the case with community conserved areas, is the wider stakeholders' empowerment and participation in conservation and wise use of fisheries resources. BMUs membership comprises the fishers, fish traders, transporters and to some extent, small-scale fish processors. Therefore they provide a common platform for fisheries resource users to establish their own code of conduct and make revisions whenever necessary. This includes the possibility for their management to impose charges on non-members for accessing fisheries resource within the BMU's areas of jurisdiction. Furthermore, the BMU leadership may also ratify recommendations from their meeting requiring members to reduce their fishing efforts, submit regular data for accuracy of fisheries statistics among other regulatory decisions that may be made from time to time. In addition, BMUs are encouraged to develop their own project ideas and mobilize resources for management of their areas of operation. Prior to their establishment, BMUs were provided with a series of training by the SDF with the objective of building their capacity in aspects of conservation, financial management, democratic leadership and governance of common property.

Although BMUs are still at their infancy stage in the coast region, they have the potential of strengthening management of coastal fisheries and marine resources. The fact they have legal status and their own by-laws stipulates their mandates and terms of operation. They are well suited to promote linkages and networks with other agencies including government departments, Community Based Organizations (CBOs) and civil societies for concerted conservation efforts. In addition BMUs have been successful in establishing cooperative and savings societies for collective selling of fish, negotiating better market prices and facilitating personal savings. Examples of this are the BMUs in Faza, Kizingitini and Kiunga in north coast, Lamu (Pers Obs). However, the BMUs will have to struggle harder to establish themselves as sustainable co-management institutions. The main sustainability challenges facing them include lack of adequate resources and technical skills for their effective management.

6.3.4 Livelihood Diversification

Livelihood diversification is today an important subject of discussion as a strategy for building resilience in rural households in Africa (Scoones, 1998; Ellis, 1999; Woodhouse, 2002). There are important lessons that can be learnt from livelihood diversification strategies: These can lead to increased household income as well as equitable distribution of income among household members. According to Ellis (1998) and Woodhouse (2002) diversification can be either at household level where the household has more than one income earner (earner diversification) or at individual level where the head of the household has income from more than one activity (i.e. activity diversification). In this regard the role of women, especially in household diversification is critical.

In Kenya, the fisheries sector is dominated by artisanal fishing methods and cultural set ups that have marginalized the dependent communities for a long time. The sector is characterized by uncertainties, low literacy levels and unawareness of sustainable livelihood options. Furthermore, as in the case for many parts of the world (Horemans and Jallow, 1997; Williams et al., 2002), the role of women in fisheries had been ignored for a long time, although the trend has changed in recent years.

Significant efforts have been put by both state and non-state agencies to increase awareness amongst the coastal communities with regard to existing opportunities for their livelihood diversification and sustainable use of coastal and marine resources. In this case, the key focus is building new capacities and strengthening existing ones so that local coastal communities can engage in other forms of income generation from the coastal and marine resources. For instance, coastal communities have been sensitized, trained and encouraged to venture into mariculture and aquaculture practices as new ways of increasing fisheries production and household income. These strategies involve the farming of aquatic organisms either in marine blackish waters (mariculture) or fresh water (aquaculture). These new livelihood strategies are being supported by the SDF in partnership with KMFRI, through various projects such the Kenya Coastal Development Project (KCDP). Silvoculture, which involves raising and replanting mangrove seedlings has proved effective in rehabilitating degraded coastal mangrove ecosystems which have been degraded by impacts from human activities. The rehabilitation of degraded mangrove ecosystems helps to improve the biodiversity and enhance breeding grounds for marine fish species.

McClanahan and Cinner (2012) observed that capacity building is a critical part of operationalizing adaptive management such as the CCA and BMUs co-management. They argue that developing skills and access to capital flow allows the marginalized communities to diversify their livelihoods. In the practical sense, capacity building provides opportunities for the fisher communities to assess and compare their investment options and enhance their adaptive capacities to respond to external factors such as seasonality especially during hard times.

There are many other aspects of capacity building strategies for the fisher community in coastal Kenya. Apiculture and silvofisheries are advocated for their potential to provide options of diversifying food production. These reduce risk in case of climate change impacts apart from providing some income opportunities for the local communities and reducing pressure in the fishery sector. Silvoculture is the practice of mangroves conservation for improved fisheries production. Apiculture, which involves beekeeping in forests such as mangroves ecosystems, has particularly been successful in areas such as Majoreni and Kibuyuni in south coast Kenya where communities are able to link their livelihood benefits with the conservation objectives. Training in value addition for optimization of income from the fishery and apiculture is critical. Examples of these are the training on solar dried and smoked fish provided by KMFRI, and honey packaging techniques by Kwetu Training Centre, a CBO working with local communities in the coast region. Other areas of capacity building include the issues of access to credit for capital asset development; awareness creation on HIV and AIDs and Other socio-sanitation related diseases which may increase vulnerability and threaten livelihoods of the fisher community.

To a large extent, capacity building is supported by both state agencies and non-state institutions that have developed programs or projects targeting artisanal fishers and rural communities. A typical example of a state initiated capacity building is the KCDP grants and scholarships for coastal communities to enhance natural resource management and community services in coast region. Some non-state agencies actively involved in capacity building along the coast region include: The East African Wildlife Society (EAWS); World Wide Fund (WWF); Coastal Oceans Research and Development in Indian Ocean (CORDIO); among many other organizations.

It can be argued that capacity building programmes in themselves may not always guarantee livelihood sustainability unless they are able to engage the local resource users and communities to effectively deal will changes in their livelihood options. IMM (2008) observes that while livelihood enhancement and diversifications are appreciated by both conservationists and development practitioners as mechanisms to promote livelihood development and discourage harmful exploitation or degradation of natural resources, the majority of efforts to this support are so far supply-driven and focused on single "blueprint" solutions. "*Such solutions are not built on an understanding of the underlying factors helping or inhibiting livelihood diversification, and often fail to appreciate the obstacles faced by the poor in trying to enhance and diversify their livelihoods."*(IMM, 2008, pp:6).

Thus, livelihood diversification ought to take into consideration realistic options of reducing over-dependence on resources within conserved areas, while compensating the dependent communities whose access to the resources is restricted. Here we suggest that the list of sustainability indicators for livelihood diversification may not be exhaustively presented, but it includes: the capacity and capability of community groups to develop opportunities for positive livelihood change through initiation of new livelihood strategies; diversity of income sources and enhanced marketing strategies for community or household members; institutionalization of community governance structures for policies, legal framework and participatory resource monitoring.

6.4 Sustainability Indicators

Economic growth in coastal Kenya has eroded cultural ways of fishing, leading to overexploitation of many reef fishes. These practices together with the increasing pressure from coastal human population will increasingly degrade the less resilient marine ecosystems. In turn, this will have serious ramifications for the livelihoods of the coastal communities. Therefore, it is important to understand the tools and/or methods of measuring and reporting progress towards sustainability of the coastal fisheries and livelihoods. In essence, indicators or indices of sustainability should be determined upfront and periodically evaluated to confirm the progress achieved.

Table 4: Sustainability Indicators (SI) of fisheries and livelihood in coastal Kenya.

SI Category	Aspect checked	Examples of SI measured
Ecological	Ecosystem health	Mangrove, seagrass and coral cover Biodiversity richness Fish abundance Pollution levels
Biological	Productivity and reproduction	Primary production Population growth rate Population mortality rates
Socio-cultural	Social and cultural security	Human demography Food security Public health and safety Equity Capacity (Education and training) Awareness levels Voluntary participation Gender roles Vulnerability and resilience Conflicts Crimes rates
Economic	Macroeconomics and Microeconomics	Gross Domestic Production Per capita income Household in come Employment Transport Market opportunities Savings and credits
Institutional and Organizational	Policies and Governance framework	Mandates Interests Values, Norms and Beliefs Groups or Associations Collaborations and partnerships Area of operation Behavioural changes
Political	National and local political dynamics	Leadership regimes Democratic decisions Level of consensus

Indicators can either be quantitative or qualitative depending on the purpose, but the quantitative ones are often more convincing and widely preferred (Gallopin, 1997).

In Kenya, despite the wide discussions with regard to the sustainability of coastal fisheries and livelihoods, the level of actual operationalization of the sustainability indicators to inform management decisions is still quite low. This may explain the reasons why success in coastal and marine areas conservation is rather limited in either context or geographical areas. Where intervention mechanisms have been enforced, the indicators of success tend to be pegged on observation of easy targets such as, changes in fish catch, species diversity, size and number of conserved areas. However, as noted in the previous sections, sustainability indicators should transcend beyond merely observing single factors in isolation, such as environmental, social, economic or political conditions. Rather, to the largest extent possible, the indicator should encompass the dimensions of ecological, biological, social, economic, institutional and political conditions surrounding a fishery in question. Indeed these six dimensions tend to influence the trajectory sustainable development the fact which has been reverberated by Hediger (1997).

By aggregating sets of indicators identified from the above six dimensions, measurable indices relevant to the context in discourse can be built and monitored for effectiveness in sustainable management of the coastal fisheries. In table 4, an attempt is made to elaborate the six sustainability indicators (SI) which are commonly used to monitor the performance of coastal fisheries and local livelihoods in Kenya. However, the extent to which the data and information on these indicators are adequate and relevant to the sustainable fisheries and livelihoods management is a subject of debate among the key agencies and stakeholders in the fisheries sector of Kenya.

It should be considered that sustainability of coastal fisheries and the local livelihood cannot be discussed in isolation of the entire ecosystem. According to Carney (1998) five critical assets should be taken into consideration with regard to livelihood sustainability. These are the human capital; natural capital; financial capital; social capital; and physical capital. Since fisheries resources, environment and society are tightly interrelated pillars, holistic assessment of their sustainability indicators should be practised to pinpoint success and weaknesses of any management strategy put in place. For instance, assessment of fisheries and livelihood sustainability in protected areas should interrogate: 1) Why specific management approaches are preferred for implementation; 2) Who was or is being affected or benefitting from the implemented approach and 3) How the approach affects the resource and people in general including political aspects. This set of questions assist to establish measurable targets that can be monitored and evaluated to ascertain whether fisheries conservation objectives and local livelihood needs are being sustained. Reasoning in the line of capital theory approach to sustainable development, as introduced previously, the fisheries governance systems in Kenya should be re-evaluated on the basis of their alignment to either weak or strong sustainability.

6.5 Experience And Lessons Learned

Some significant progress has been made towards enhancing sustainable coastal fisheries and livelihoods in Kenya. A critical part of the approaches being implemented is the participatory process. Experience has shown that the involvement of local communities in planning and implementing specific resource management strategies is a better option that not only inculcates process ownership but also provides opportunities for flexible governance. Such participatory approaches enable the relevant authorities to make an entry point into the community and in a manner that facilitates social leaning and establishment of own rules. In coastal fisheries management the BMU, CCA and capacity building approaches have positive influence on the five components of sustainability.

However, levels of success and impacts of whatever participatory approach designed cannot be assumed. It is necessary to measure and monitor the process using reliable sustainability indicators. The indicators should be integrative and help to provide answers to the *why*, *who* and *how* questions discussed earlier. Answers to the questions should also be bundled and linked to the three pillars of sustainable development (environmental, economic, social and governance). In this perspective, it is worthwhile to note that the above sustainability indicators may switch to either positive or negative sides due to external drivers such as climate change, increased due to coastal population growth, among other drivers. Further studies are recommended in this line of thought so as to ascertain whether the current coastal fisheries management and livelihood strategies are indeed sustainable in the long term.

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7 Peninsular Pronghorn Conservation: Too Many Paradigms, Too Few Indicators

Alejandro de las Heras and Marina Islas-Espinoza

7.1 Introduction

Pronghorn (*Antilocapra americana*) is the only member of the Antilocapridae family, and differs from bovids, cervids and other ruminants. It is found in North American deserts and grasslands (Fig. 1). Pronghorn is the second fastest land animal but can run for much longer than cheetah. Antilocapridae evolved in North America and were a successful family thanks to digestive and temperature regulation evolutions. These evolutions probably were a response to climate becoming highly seasonal about 34 million years ago, with glaciations alternating every 41-100 thousand years with temperatures slightly warmer than today, and millennial cycles of 2^{o}C local cooling (Maslin, 2009). Lacking equivalent temperature regulation *equus* including horses came close to extinction (Mitchell and Lust, 2008; Kulemzina et al., 2014).

During the 19th century the pronghorn population plummeted from 35 million to 20 thousand. From 1924 however, the population increased to 700,000, most of them in the US and less than 2500 in Mexico, where population is seemingly declining (Hoffmann et al., 2008), despite hunting prohibition everywhere in Mexico since 1922 (INE, 2000). All Mexican pronghorn are protected under the Convention on International Trade in Endangered Species (CITES) Appendix I (CITES and UNEP, 2009). In the US, subspecies *A. americana sonoriensis* is protected under the Endangered Species Act, ESA (US FWS, 1967). Another subspecies, also protected in Mexico is *A. americana peninsularis (peninsularis* henceforth), mostly present in El Vizcaino Biosphere Reserve in the Baja Peninsula (Fig. 2) and struggling to return to its 500-head 1925 population (INE, 2000).

Disturbingly, flagship protected areas in North America, such as Banff, Yellowstone and El Vizcaino have failed to provide a thriving environment for pronghorn and other ESA big game species (Berger et al., 2008; Hebblewhite et al., 2009). The aim of this study was to explore the drivers of such failures and derive a minimal set of indicators to assess state and threats, conservation malpractices, transparency and accountability.

To understand the failure of *peninsularis* to grow demographically despite two decades of intensive protection, this chapter firstly stacked the practice and theory of conservation against each other. Secondly, a knowledge network experiment identified short-term solutions to management issues. Knowledge systems are networks of actors and organizations that link knowledge and knowhow with action (McCullough and

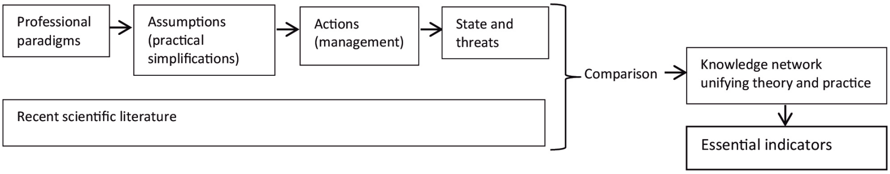
Matson, 2011). Finally, the foregoing elements (state, threats, management practices, assumptions and paradigms), representative of the first decade of implementation of El Vizcaino management plan (published as INE, 2000), as well as theoretical and practical recommendations, were used to derive a minimal set of essential indicators for use in improved management practices, and to inform long-term stakeholder participation (Scheme 1).

The state of captive, and free-roaming but fed, *peninsularis* was ascertained based on fitness and health data. Immediate threats to the populations were identified using breeding records and digital geographic information. These were then linked to a list of ongoing management practices in El Vizcaino. The underlying assumptions of practices were teased out of the lingo used in work conversations during nine continuous months with the personnel, leaders and non-governmental organizations (NGO) representatives in charge of the intensive management of *peninsularis*. Assumptions are working simplifications of paradigms in conservation disciplines; they are inherited from contact with external advisors or with other wildlife management projects. Such paradigms were identified in explicit documents (zoo association guidelines and park management plans), derived from long-term interactions with professionals (veterinaries), or resulted from research (on ranching in El Vizcaino, de las Heras et al., 2014). Others were deduced from work conversations with hunting guides (in the bighorn sheep project in El Vizcaino).

7.2 *Peninsularis* State And Threats

Peninsularis is probably below the expected recovery target of 500 head by 2010: 200 are in the wild (AZA Antelope and Giraffe Advisory Group, 2008) and 268 under management (daily head counts in La Choya peninsula, Fig. 2), of which 165 roam freely and 103 are captive (64 males, 33 females and 6 juveniles, separated in 3 enclosures). Mortality in the enclosed population during the 9-month observation period amounted to 2 males and 1 female (8, 1 and 1 years-old), all killed by conspecifics. From February 2007, fawns were raised in captivity, the youngest being 2 days old at capture. In 2008, 30 fawns were born but 19 had died from clostridiosis by March or had been euthanized; survival was 33% at weaning. No births were recorded in 2009-2010. Fawn survival at weaning in Yellowstone was 5-15% and 26-44% (in absence and presence of wolves, respectively; Berger et al., 2008) and 51-100% in three reintroductions in Mexico from Wyoming (Cancino Hernández, 2006). The dead 1-year-old female was the only 2007-2010 recorded recruit but she never got impregnated. With 2 surviving fawns among those born in 2007-2010, survival at maturity was 4-15%.

Moderate *Eimeria* parasitosis (mild in enclosed males and severe in females) was observed (laboratory analysis by A. Barbabosa). Painful hooves were also present due to lack of wear in the sandy enclosure terrain, which added to probable arthritic and skeletal issues related to copper (Cu) deficiency (Fig. 1). Cu:Zn and other nutritional



Scheme 1: Method.

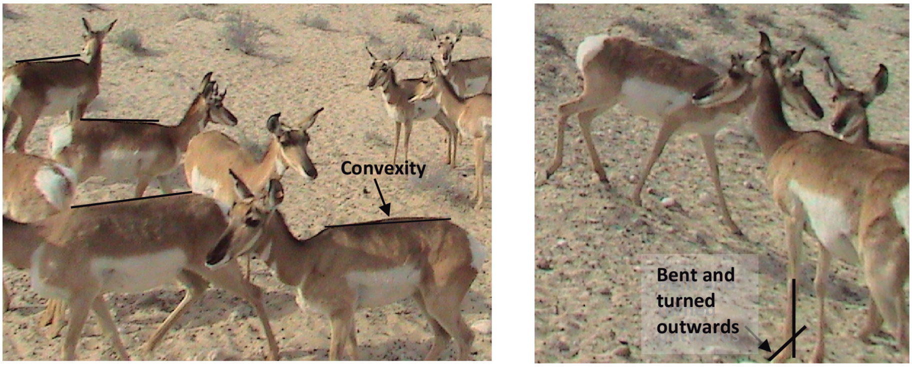


Figure 1: Clinical signs of Cu deficit. Convex backs and feet problems (bent, moving sideways while walking, turned inwards or outwards, difficult gait), hair problems (discolored around the eyes, ill-delimited hair colors, stiff hair) were observed in videos and pictures of free-roaming individuals in La Choya. Left: Convex back as opposed to flat backs. Right: Animal with straight back and good hair (even, intense color, good separation between colors in flank and hind quarters) but bent distal segment of front leg compared to the individual walking down (M. Huerta personal communication) (©Photos by A. de las Heras).

imbalances could lead to low sperm and ovule quality, embryo losses and low resistance to clostridiosis (M. Huerta, R. Rangel personal communication). Emotional-behavioral health issues were stereotypies (repetitive behavior of some individuals near the fence) and the deadly episodes referred to above.

The foregoing are signs of probable nutritional imbalance in the alfalfa (*Medicago sativa*) diet fed by humans. As to desert vegetation patchiness, it had reproductive implications: flushing or extra food consumed prior to breeding was only likely to happen if *peninsularis* roamed a wide range after rainfall (Fig. 2, left). Contrariwise, flushing was limited in a captive environment (Fig. 2, right); high protein alfalfa inputs did not compensate for nutrients available in the wild.

Access to nutrients was also restricted in the multiply fragmented habitat: by 1973 the trans-peninsular road cut off the western hyperarid Vizcaino portion of desert from the slightly wetter Angelino-Loretano portion (Peinado et al., 2005) to the east. The core protected area dedicated to *peninsularis* protection in El Vizcaino lies in the hyperarid part (Fig. 3).

Figure 2: Seasonal microphillic brush and grass bloom after rainfall. Left: 2-8 inches tall vegetation after 7mm January-February accumulated rainfall. Red and green vegetation only appear after rainfall. Grey vegetation is perennial. Right: Inside the fence, grazed seasonal microphyllic plants during the flush period closely resemble the usual scant vegetation cover. The difference with ungrazed vegetation outside the fence is patent. (©Photos by A. de las Heras).

Water salinity in well water, possibly due to seawater intrusion in the study area, likely increased *peninsularis* water requirements. Nutrient and water imbalances in the flushing, mating and breeding season were likely to affect sperm and ovule quality as well as embryo and newborn survival (R. Rangel personal communication). Water requirements varied considerably between seasons (Fig. 4) and so strongly contradicted the prevalent assumption of *peninsularis* reliance on the sole yearlong Pacific Ocean fog condensation on vegetation.

The foregoing issues were indicative of vulnerability to demographic and environmental stochasticities (risks affecting small biological populations). Genetic and catastrophic stochasticities could also be lurking (Table 1) since a metapopulation (set of spatially separated populations) survives if local extinction of a population is compensated for by migration from another population. This holds when sufficient density of local patches exists. In the case of *peninsularis*, wild and managed populations were cut off from each other; population growth in the latter was stalled. Seasonal migration between Vizcaino and Angelino-Loretano seasonal vegetation covers was halted. This impeded metapopulation dynamics and entailed enhanced extinction risk for wild and managed populations. Because metapopulation processes (reproduction and migration) were weak, strong population measures were warranted in the short term (such as assisted reproduction), as well as longer-term ecosystem measures (to restore gene flows and access to nutrients and free-standing freshwater).

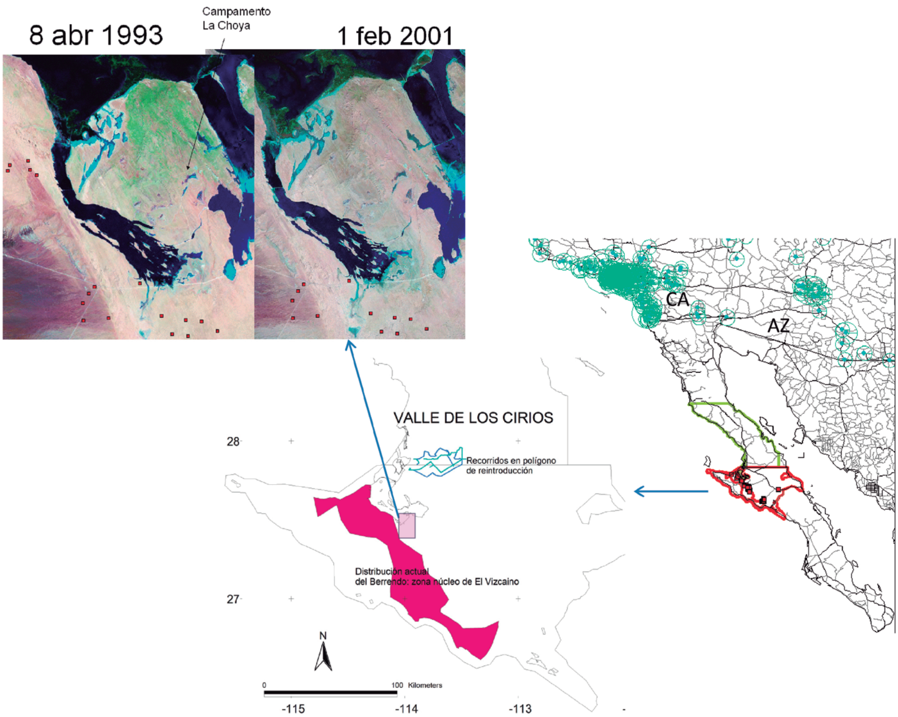


Figure 3: The multiply fragmented *peninsularis* habitat. Right: Animal movements out of the Baja peninsula are impeded by the US border, highways and metropolitan areas on either side of the border (CA: California; AZ: Arizona). Urban places are represented by circles proportional to their populations. The dry Vizcaino section of Cirios Protected Area (yellow outline) and El Vizcaino Biosphere Reserve (red outline, with red dots indicating sightings of *peninsularis* in the wild) are the conventionally accepted natural habitat of *peninsularis* (INE, 2000). Center: disturbances in the core area of the Biosphere Reserve dedicated to *peninsularis* (purple outline) include road traffic, reserve-authorized off-road racing, and extensive cattle ranching. La Choya's experiment was to be expanded to a 40-km-perimeter hunting enclosure (GPS tracks in blue) inside Valle de los Cirios protected area. Left: Typical of deserts, moisture is patchy in La Choya peninsula, recently cut out by saltworks (saltpans and a canal in the south) protecting the captive population from poaching but further fragmenting habitat and gene flows with wild populations.

7.3 Conservation In Practice: Assumptions And Paradigms

A list of observed *peninsularis* conservation activities was divided into 4 main groups and each was related to professional paradigms which encourage these activities (Table 2). The zoo paradigm was inherited by the *peninsularis* conservation project via contact with several US institutions, mostly the Cabeza Prieta National Wildlife Refuge in Arizona and its captive reproduction facility -under the auspices of the Binational Committee for the recovery of *sonoriensis* pronghorn-, and the Los Angeles

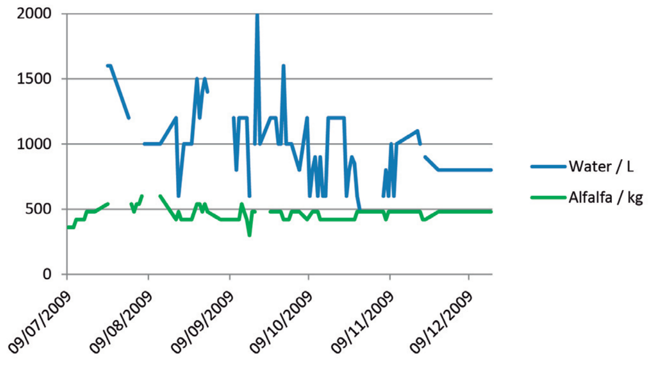


Figure 4: Water and alfalfa inputs to enclosed and free-roaming *peninsularis*. Ad libitum water supply was a reasonable estimate of water requirement. Water: feed 7-day and 14-day moving averages (not shown) showed a reduction from summer levels (3:1) to winter levels (1.5:1). Water was supplied by the saltworks as part of their cooperation with the Biosphere Reserve but was often too saline.

Table 1: Potential threats and associated stochasticities.

Threats	**Stochasticities**			
	Demographic	**Genetic**	**Environmental**	**Catastrophes**
Isolated populations	Reproductive failure, accidental mortality	Inbreeding, local extinction	Fragmented, incomplete ecosystem. Restricted access to very patchy vegetation. Climate variation impact strongest in driest remnant wild habitat	Extreme draughts under climate change
Zoonoses			Contamination by livestock and ranchers handling *peninsularis*	Epizootics. Nutritional deficiencies diminish resistance
Poaching	Increases mortality, decreases reproduction	Affects the 'best' phenotype, not diseased individuals		
Domestication			Habituation to human presence. Use of facilities as shield against predators	Human breeding: decreased immunity, increased infections
Privatization	Lack of transparency on recovery trend. Scarcity management secures funds	Purity of subspecies preferred over genetic diversity. No assisted reproduction	Fencing, stereotypies, excess density, injuries and death	

Zoo. Upgrading of the *peninsularis* conservation project in the American Zoo and Aquarium Association (AZA) survey (Shurter and Fischer, 2006; AZA Antelope and Giraffe Advisory Group, 2008) probably meant conformity to AZA *ex situ* captivity, demographic, public appeal indicators, criteria and prescriptions.

The vet paradigm in its simplified form applied to livestock was also applied to *peninsularis*; it aims at production (individual biomass increase) and reproduction. Therapeutics is secondary and pathos (suffering) overlooked. The ranch, zoo and vet paradigms focus on demography, captivity and feed largely overlapped. The amount of feed and water were principal and food quality secondary. Mexican alfalfa exports to the US meant low-rate feed was bought. A project to produce alfalfa for *peninsularis* in Cirios would likely mean more compliance with the ranch paradigm.

In addition, *peninsularis* management obeyed Mexican wildlife bylaws (Reglamento General de Vida Silvestre) regulating 'wildlife management and utilization units' (unidades de manejo y aprovechamiento de la vida silvestre, UMA). The intended project emulating the successful bighorn sheep UMA in El Vizcaino was based on the hunt paradigm and corporate mitigation funds. Both bighorn and pronghorn are two of the five ESA big game species (the Grand Slam in hunting parlance). Hunting UMAs assume that replacement by man of senescent dominant males improves the genetic pool, that man as keystone species can supplant all predators, and that human protection is better than ecosystem balance. Poaching, in addition, is commonplace in El Vizcaino (INE, 2000).

The park paradigm dominant in the US conservation model has spread worldwide and in particular inspires the core areas of Biosphere Reserves. This paradigm relies on removing humans from wilderness, but it makes large allowance for disturbance from tourism. The park paradigm could be traced in all managing activities (Table 2). The Biosphere authorities however were largely sidelined by the *peninsularis* high-profile project, funded by Ford Motor Co initially with a 400 thousand US dollar grant in 1997 (INE, 2000) and operated by an NGO which also managed the World Bank Global Environmental Facility funds for 27 Mexican protected areas. Still El Vizcaino contributed most of the workers of the project. The NGO paradigm was patent in the resource management tier of the project (Table 2). The NGO paradigm however seemed to face a commitment dilemma: On the one hand, caring for the animals; on the other hand managing (i.e. maintaining) scarcity of the subspecies, to keep funds flowing to this and other conservation programs. The NGO paradigm upheld the importance of the subspecies over free specimens. Sensitivity to reputational risk was noticeable in the reluctance to carry out censuses which could show failure to achieve recovery targets. Autonomy of the *peninsularis* project was also noticeable in its absence from the agenda of meetings of El Vizcaino overseeing committee.

Table 2: Summary of conservation activities. Square brackets: underlying paradigms. Parentheses: Activities not carried out.

Managing populations [Zoo, Vet, Park]	**Medicating [Vet]**	**Feeding and watering [Vet, Ranch]**	**Managing resources [NGO, Hunt]**
Fencing and building facilities	During breeding. Keeping records	Transporting alfalfa over 1000 km	Lease from landholders
Marking individuals. Separating sexes and juveniles	(Assisted reproduction)	Negotiating water truck pipes with the saltworks corporation	Negotiate funds with corporations, NGO and individuals
Chute-capture. Capture of fawns. Transport to captive management facilities		Feeding and watering daily. Occasional cleaning of enclosures	Negotiate easement with NGOs and landholders
Surveys. Head counts. Counting newborn fawns. Animal and vegetation studies. (Census)		Providing minerals and pellets	Negotiate survey, autonomy, manpower and truck from Reserve
Fielding coyotes. Surveillance		Keeping records	Negotiate with local alfalfa producers
Managing scarcity: defending the existence, and purity, of a subspecies			Public relations: hosting tourists and press

Table 3: Overlapping assumptions and paradigms. Domestication (D) and privatization (P) as consequences.

PRACTICE	**ASSUMPTION**	**PARADIGM**
[D,P]Captivity	Umwelt and telos irrelevant. Habitat area irrelevant. Protection against coyotes, poaching	Zoo, ranch
[D,P]Breeding	Fawns born in captivity are UMA property	UMA
[D]Selection by man	Phenotype (outer aspect) preferred over heterozygosis (genetic diversity), selection by man not by natural evolution	Vet, ranch
[D]Feed and water	Protein first in animal production. Salinity irrelevant	Vet, ranch
[D]Tourism	Disturbance is negligible	Zoo, park, hunt
[P]Autonomy	Insiders/outsiders dichotomy	NGO
Single-species management	Pronghorn as desert gardener. Valid umbrella species. Hyper-arid plains as original habitat. Carrying capacity referred to one single species. Net herbivory effect neglected	Park

Umwelt is the particular worldview of an animal species. Telos is what characterizes an animal species (its role in the environment and evolved fitness abilities).

Overlapping assumptions and paradigms (Table 3) seemingly revealed domestication and privatization trends. Domestication use of baby bottle and formula, enclosure, overreliance on feed and human-supplied water, pronghorn use of human facilities as shield from coyotes, (see Berger, 2007) and privatization (ownership of UMA-born fawns, and lucrative hunting) were emergent properties potentially negating fitness in the wild. Whether domestication and privatization were intentional was unclear.

7.4 Knowledge Network Short-Term Recommendations

Peninsularis conservation had eluded catastrophic stochasticities, such as ungulate epizootics in adults, possibly thanks to isolation in La Choya. But in projected isolates north of the 28th parallel, epizootics could be more probable in enclosures previously exposed to livestock. Isolation of *peninsularis* from workers who are also ranchers or live in ranching communities, or from exposed visitors, seemed difficult. Human breeding might worsen catastrophic stochasticities via isolation from mother colostrum and nutrient deficiency in feed.

To allay demographic and genetic stochasticities, until that time when sustainability is achieved through restoration of complete ecosystems, reproduction and recruitment (survival to sexual maturity) ought to be monitored and lapses prevented by recourse to assisted reproduction (which includes consanguinity and paternity analyses and artificial insemination). Human selection based on phenotype first and then on heterozygosis may allay the threat of genetic depression. It does not guarantee however the genotype most immune to diseases. Microsatellite analyses (Carling et al., 2003) would help determine allelic polymorphism (J.C. Vazquez personal communication). Tagging and microchip follow-up ought to help avoid consanguinity. Laparoscopic insemination and anesthesia could then be carried out by external practitioners committed to long-term cooperation (R. Rangel pers. comm.). Other future needs include sperm banks, mineral micronutrient analyses in feed, wild vegetation and blood (e.g. Cu, Zn, Se, Mo), as well as stress hormone analyses (cortisol and epinephrine).

7.5 Longer-Term Need For Complete Ecosystems

The importance of a complete environment for *peninsularis* was exemplified by a female and fawn after winter showers, when they were most averse to alfalfa feed, and occupied an Adam tree (*Fouquieria diguetii*) patch with abundant hare defecations to hide and mask smell from coyotes. This suggested awareness and use of a complete ecos, with intertwined telos and ethos (Fox, 2005), i.e. fulfillment of physical, behavioral and psychological requirements and roles in the ecosystem.

Although absent from the zoo-vet-ranch paradigms, habitat is the most salient factor affecting the viability of small populations (Hebblewhite et al., 2009; Hoffmann et al., 2008; Lee and Jetz, 2011; Rodrigues et al., 2005). The assumption that *peninsularis* are gardeners of the desert interestingly points to seed dispersal and nutrient recycling via feces. Ungulate effects on plants however run the gamut from dispersal to alteration and mostly entail a net decrease in plant biomass (Cadenasso et al., 2002; Maron and Kauffman, 2006). Although ungulate impact seems proportional to nutrient content (Asner et al., 2009), impact of *peninsularis* on palatable shrubs (*Atriplex canescens*, *Encelia farinose*, *Frankenia palmeri*, *Fouquieria diguetii*) might also depend on soil water retention capacity and salt content. None of these shrubs is Baja- or Vizcaino-endemic (Peinado et al., 2005); this suggests a wider natural ecosystem for *peninsularis* than hyperarid and sometimes hypersaline Vizcaino.

Rather than gardeners -an anthropomorphic notion- ungulates are strong interactors (Soulé et al., 2003; Donlan et al., 2006) with direct and indirect (cascade) effects on the habitat and resource availability of other herbivores (insects, lizards, lagomorphs and rodents; Gibbens et al., 1993; Maron and Kauffman, 2006; Pringle et al., 2007), microbiota (soil biological crusts; Manier and Hobbs, 2006, and fungal plant symbionts; Clay et al., 2005), pollinators and dispersers, competing ungulates, and predators.

Although threat lists (IUCN – International Union for the Conservation of Nature – Red List, CITES and ESA) and the *peninsularis* project alike focus on organismal biology, the definition and unstable application of species concepts, the lack of knowledge of most species and the focus on population size are now strongly debated. Population and habitat are not sufficient criteria and ecological targets are pressingly needed; *peninsularis* as umbrella species is not sufficient to establish biodiversity conservation targets (Possingham et al., 2002; Reed et al., 2002). Ecosystem-level and multitrophic-level studies are essential in understanding extinction dynamics in endangered species (Hebblewhite et al., 2009).

As regards the park paradigm, naturalness criteria are defeated in practice by ubiquitous human presence; they are being supplanted by historical fidelity, autonomy of nature, ecological integrity, and resilience targets (Hobbs et al., 2010). As fitness in the wild may require larger gene flows than usually reckoned by population viability analyses (Reed et al., 2002), connectivity between patch populations is essential. Design of buffer zones, biological corridors and stepping stones in fragmented park habitats now requires experimental evidence in particular relating to wildlife movement, and a focus on animal complementarity in the face of cyclical and anthropogenic climate changes (Soulé et al., 2003; Chazdon et al., 2009; Woodruff, 2010). A radical response, rewilding or restoration of complete Pleistocene ecosystems, aims at devolving evolutionary capability (Donlan et al., 2006; Woodruff, 2010). Complete ecosystems include top-predators (keystone species, Soulé et al., 2003) preying on coyotes; in their absence, pronghorn fawn survival plummets (Berger et al., 2008). Reintroduction of top predators is critical for simplified ecosystems (Pringle et al., 2007). Rewilding of degraded rangelands more than rest alone requires restoring and sustaining natural processes (Curtin, 2002).

At a narrow timescale, pronghorn populations vary in relation to drought and winter severity; and water needs vary in relation to succulence of vegetation (Hoffmann et al., 2008). At a wider timescale, the drought trend (reviewed in de las Heras et al., 2014) indicates an increasingly dry Sonoran desert. The anatomic adaptations of pronghorn may not be sufficient when higher temperature and less humidity coincide with restricted movement in an ecosystem devoid of free-standing water, succulent vegetation, and keystone predators (wolves and pumas) to exert top-down control on pronghorn herbivory. Past monsoon Holocene switch-offs (ibid.) are cautionary tales for the North American monsoon that brings summer moisture to eastern Baja. The coincidence of stochasticities and systematic threats such as climate change bears resemblance with climatic and anthropogenic factors conspiring 50-10 thousand years ago in megafaunal and antilocaprid mortality (Koch and Barnosky, 2006).

7.6 Discussion

7.6.1 Privatization And Information

While the debate on ecosystem restoration has coalesced around rewilding and coupled human and natural systems, current human relationships with nature are driven by privatization. Privatization critiques relate firstly to the lack of information being issued by private initiatives, the need for independent reporting, oversight and accountability for the consequence of wilderness management (Alcorn et al., 2005; Igoe, 2007). Secondly, funds become a dominant logic of elite NGOs monopolizing donors while at the same time reducing participation of and coordination with other stakeholders. This overreliance on donations makes projects vulnerable to inherently unstable financial markets, donor interests, policies and fearful to offend or lose donors (Alcorn et al., 2005; More, 2005). Thirdly, self-interest leads to a defective view of our obligations to the future (Pezzey, 1989), and a focus on managing extinction which sidelines restoration and evolutionary processes (Donlan et al., 2006). Fourthly, restricted access is the mechanism for privatization. It assumes either the form of exclusion of visitors unwilling to pay when management is for profit, as in Yellowstone, Yosemite or in private easements in Patagonia (More, 2005; Carey, 2009), or the form of access restriction for local residents as in Canada, the US and South African game farms and protected areas (West et al., 2006; Healy, 2007). Fifthly, in contexts devoid of self-imposed rules and enforcement, “privatization is the worst possible fate” (Bowles et al., 1998; Terborgh, 2000).

7.6.2 Information And Stakeholders In Knowledge Networks

In El Vizcaino Biosphere Reserve inhabitants are “marginalized by the very conservation process meant to engage them as key actors in promoting natural-resource protection” (Young, 1999). This runs counter the need in endangered species’ recovery projects for interdisciplinary frameworks and teams, exchange of knowledge and resources to build research, management, assessment, and policy partnerships (Reed et al., 2002; Chazdon et al., 2009). Interim science and management reports must be shared among stakeholders and information shared on all issues in regular meetings and interactions (Hebblewhite et al., 2009). Participatory science can engage local residents in monitoring, reporting and discussion activities as ways of promoting awareness and action. This can deliver much more fine-grained information than large organizations relying on expert judgment and facing difficulties in meeting their pledges on open-access data and grass-root participation (Rodrigues et al., 2005). This however implies a shift in values from aversion to reputational risk (inherited from large donors) towards transparency, international and local accountability, and involvement of interested third parties. This is equivalent to a shift from the privatization trend to participatory science and practice.

Another way of participating is via conservation easements, whereby landholders have been seeking to protect their land rights from corporate mining interests in the allocated *peninsularis* habitat. Unused community land has also been relinquished by landholders in favor of the Biosphere Reserve (Harris, 2008). In such common asset trusts, users can make their own rules – often managing more successfully than private owners and legal parks – and produce freely available information and technologies enhancing and protecting public goods (Gibson et al., 2002; Hayes, 2006; Beddoe et al., 2009). This pooling of large land tracts and perpetuity easements is consistent with the IUCN definition of protected areas, and could in theory host Pleistocene rewilding or bioneering experiences – interventionist ecological management – and include the knowledge of local traditional communities, in a timeframe allowing for evolution to cope with climate change and other large-scale fluctuation factors (Donlan et al., 2006; Dudley, 2008; Woodruff, 2010).

Parks may not be the optimal governance structure for local conservation (Hayes, 2006) especially when power is lost to NGOs, prompting lack of coordination. The park and NGO paradigms also make too many concessions to resource extraction and to disturbances associated with tourism: in El Vizcaino’s *peninsularis* core protected corridor, off-road tourism and races degrade the land (United Nations Environment Programme and World Conservation Monitoring Centre, 2008) but permits continue to be issued on account of good relations with the residents of the Reserve. The sensitivity of *peninsularis* to disturbances can be inferred by reference to death of two juveniles induced in 2009 by the presence of a photographer in the enclosures, or by *peninsularis* being relegated to the least hospitable habitat to avoid human presence.

If however, international and local participation in rewilding make inroads and subvert the current park-and-NGO governance of wildlife, apparent contradictions could disappear between the necessity for very large undisturbed land tracts and local involvement. This is dependent on a set of indicators agreed upon in knowledge systems (Box 1). These indicators are intended to guide the assessment of conservation practices, ascertaining consistency with or justified departure from current literature consensuses, as well as facilitating knowledge networks. Specific indicators should be defined by stakeholder participation (see however the Appendix for a proposal).

BOX 1. Minimal set of indicators for network involvement and accountability

TIER1
State, long- and short-term threats,
and response indicators (stochasticities and systematic threats)

TIER 2
Ecos, telos and ethos indicators (disturbances),
short-term (recruitment, mortality and biometrics)
and longer-term (complete ecosystem restoration) indicators

TIER3
Knowledge network, malpractice, transparency and accountability indicators

7.6.3 Indicators In Practice

In theory, sustainability is indicated by the indefinitely continued existence of a living population, or more generally a natural stock (de las Heras, 2014). In practice, tallying, analyzing indicators, guiding collaboration and correcting decisions is a sequence of activities both time- and energy-consuming. Cost often makes it difficult to dedicate personnel to ensuring data quality, scope and depth. And so it is often felt that only a narrow set of indicators is needed. This attitude runs counter the comprehensive view of the natural and social environment which sustainability implies. This is further complicated by the need to quantitatively assess decisions against evolving, qualitative, scientific knowledge.

The way out of this conundrum is to rely on a larger set of skills and to standardize the loop of information-communication-collaboration implicit in a knowledge network. The latter will identify the indicators that are relevant and practical to collect, assess conservation efforts, and implement corrective and preemptive measures (see Appendix for a data collection instrument). This loop would likely steer a conservation program away from management and into decision-making with identifiable tradeoffs. Strategic tradeoffs may involve funding or public relations conflicts. Field tradeoffs arise when indicators are too difficult to obtain and hamper timely field decisions, in which cases proxies are required.

Finally caution is warranted in a series of common situations. Firstly, tradeoffs are sometimes solved by forcing data to comply with goals. This happens when pressure mounts and goals are over-optimistic. Secondly, when problems are ill-specified, indicators are of little help. This occurs, as exposed here, when standard theories and responses bias analyses or when managers supply all the indicators, also producing bias. Finally hiring specialists as the third party responsible for indicators poses two diverging problems: outsiders have to adjust their skills and knowledge to the situations at hand or else, conservation actions have to adjust to the specialists' framework. Again, a workaround is to establish long-term links with specialists within a knowledge network.

7.6.4 Indicators, Models, Metrics Or Unified Theory?

NGO *peninsularis* management lacked indicators. More generally El Vizcaino lacked organized datasets of permit-holders, temporary work, funding, wildlife and spatial information. We took a detour and instead of stating directly what relevant indicators might be, we devised a method for collectively delineating short-term measures and indicators, and outlining long-term solutions from a comparison with available literature.

This approach to participatory science relying on indicators differed from statistical models aiming at unraveling the covariates of extinction (Lee and Jetz, 2011) which require, but do not always use, good quality data. Other model issues are that they rely on implicit assumptions and software and so yield different results (Reed et al., 2002; CONABIO, 2007). Deep issues are also involved in expert ranking of the importance of biological taxa and nationwide top-down prioritization, disregarding peninsular biogeography.

Metrics, or sets of interrelated indicators, although desirable require good indicator characteristics (unequivocal interpretation, assumption-free estimation, easy replication and easy validation). Indicators and metrics have an important practical role to play towards the acutely needed unified theory in conservation.

7.7 Conclusions

Demographic growth indicators called for a reconsideration of practices in *Antilocapra americana peninsularis* recovery. Qualitative indicators evidenced underlying assumptions and paradigms lagging behind recent (molecular and ecological) science. In particular interactions with zoo professionals had led to an incipient domestication trend. The mainstream NGO paradigm seemed conducive to a privatization trend, in which recovery was delayed on account of continuation of donations. Recent conservation literature recommends caution against privatization and a shift in focus from population to a complete ecosystem where strong interactors like *peninsularis* are in balance with plants, the other herbivores and predators. A simplified view has confused the allotted *peninsularis* area in the Biosphere Reserve (the hyperarid Vizcaino area) with its natural habitat. The eastern summer and western winter rainfalls, and the extremely developed adaptation of *peninsularis* for migration, suggest a much larger and richer ecosystem. Restoration of a complete ecosystem calls for participation of landholders in perpetuity easements. Short-term solutions to stalled demographic growth include assisted reproduction and balanced nutrition. Population growth and easements are not independent and cooperation between actors calls for a durable knowledge network. In the latter, exchange of key indicators supports participatory decisions.

The relative ease with which these indicators can be obtained and validated suggests their use in many recovery projects. A windfall for scientists and concerned citizens, if project documentation becomes widespread would be the possibility to assess the global response to the current extinction crisis, and especially the efficacy of Biosphere Reserves in species and ecosystem recovery.

Conservation practice and theory should systematically be compared. Accountability, transparency and stakeholder involvement in knowledge systems should circumvent ingrained paradigms. They require an appropriate set of indicators.

At this point recommendations are usually in order. However, in line with best knowledge network and participatory practices, recommendations should be made after the stakeholders have identified the problems to be solved. This chapter is best considered a ‘conciliatory participation’ typical of intermediary roles in knowledge networks. These intermediary roles have often facilitated communication among stakeholders, just as indicators have been a requisite for objective problem identification and problem-solving.

Appendix: Pronghorn Management Questionnaire (Essential Indicators)

Questionnaire to be completed twice yearly
by an independent third-party
and validated and analyzed by all stakeholders
(PLEASE ATTACH EVIDENCE FOR EACH SECTION)

Neutral third party completing this questionnaire_________________
Date __/____/___
Date of last questionnaire__/____/___
Is this questionnaire completed twice annually? Yes□ No□
Questionnaire validated and analyzed by

- _________________________________
- _________________________________
- _________________________________

State
Metapopulation growth rate _______% (managed population ________%).
Mortality rate______%. Fertility rate______%. Permanent migration (immigration-emigration) rate_______%. Biometric data (attach). Parasitology data (attach).

Threats
Isolated populations (including managed populations)
Number of isolates__________Locations________________________
Total head in isolates_______________. Seasonal migration rate between isolates' habitats_______%. Is there risk of reproductive failure? Yes□ No□ Fawns born in the last breeding season____. Not enough males? Yes□ No□. Not enough females? Yes□ No□ Is inbreeding occurring? Yes□ No□ Is there risk of extinction? Yes□ No□ Is there accidental mortality? Yes□ No□ When? ___________________. Main cause_______________________. Secondary cause_____________. Is migration hindered by habitat fragmentation? Yes□ No□
Causes of fragmentation____
Isolates habitat: Locations of extreme aridity____________________________
Locations of extreme salinity____
Zoonoses
Possible source of domestic epizootics__________________________________
How can the etiological agent travel?___________________________________
Poaching
Are there news of poaching? Yes□ No□ Evidence of poaching? Yes□ No□
Locations____, suspected parties___________________, modus operandi________________________
Catastrophes
Extreme draught: date of last event__/__/____. Duration of last event____________. Duration of current draught___________. Is there desertization (change in vegetation)? Yes□ No□ Location_________________
Is there a risk of drowning? Yes□ No□ Where? __________________________

Management actions (MA)
Science-based (attach list and justify)
Non-science-based (attach list and justify)
Food provision (specify diet including macro and micronutrients, amount, and frequency of supply)
Water provision (specify salinity, amount and frequency of supply)
Number of food providers___________. Number of water providers______
Number of food supplement providers______. Number of medicine providers_________. Number of funding sources_______.
Specify leases and their duration____
Specify (perpetuity) easements____

Assessment of MA
Expected demographic and genetic impact of MA (attach).
Injuries: incidence________, prevalence___________.

Expected domestication effects? Yes□ No□ Habituation to human presence? Yes□ No□ Use of facilities as shield against predators? Yes□ No□ Human breeding? Yes□ No□ Assisted reproduction? Yes□ No□ Stereotypies? Yes□ No□ Density in enclosures_______________ Are the populations dependent on man? Yes□ No□

Expected privatization effects? Yes□ No□ Transparency on recovery trend? Yes□ No□ Benefits stemming from scarcity____________________________________

Corrective and preemptive MA
Deviations (attach)____
Corrections (attach)____
Anticipated deviations (attach)____
Preemptive actions (attach)____

Existence of knowledge network
Stakeholders (attach list). Involved stakeholders (attach list). Frequency of meetings__________. Frequency of informal dialogues with each stakeholder group
____

Are decisions made on a single species? Yes□ No□ Specify net primary productivity of the habitat landscapes (attach) Specify carrying capacity for the species____________; for all ungulates__________; for predators________.

Specify the trophic network of the protected species, including meso and apex predators (attach). Specify the functional diversity of herbivores in the habitat (attach). Are decisions made based on information about a complete ecosystem? Yes□ No□ Specify the net herbivory effect (attach).
Specify invasive research methods _______________________________________

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8 Restoration Success Of Tropical Forests: The Search For Indicators

Jerônimo Boelsums Barreto Sansevero and Mário Luís Garbin

8.1 Introduction

Human activities are viewed as a main cause for the planet's current environmental crisis (Millennium Ecosystem Assessment – MA, 2005; Ellis et al., 2010). Deforestation, forest fragmentation, agricultural activities, pollution and expansion of urban areas have changed about two thirds of earth's ecosystems (MA, 2005) and, as a consequence, have led to biodiversity loss, species' extinctions and reduction of the resilience capacity of many ecosystems (Peterson et al., 1998; Silva and Tabarelli, 2000; Laliberté et al., 2010). Indirectly, such transformations affect the provision of many ecosystem services such as soil fertility, water quality, pollination, and recreation (MA, 2005; Ditt et al., 2010). Based on this condition, ecosystem restoration is crucial not only to conserve biodiversity, but also to provide ecosystem services (Rey Benayas et al., 2009; Palmer and Filoso, 2009). This is particularly problematic for tropical forests due to their high diversity and habitat loss (Laurance, 1999; Myers et al., 2000; Chazdon 2008). Currently, there are some ambitious restoration goals to revert this scenario of environmental degradation (Calmon et al., 2011). The Convention on Biological Diversity aimed to restore 15% of world's degraded areas until 2020, whereas the Bonn Challenge plans to restore 150 million hectares around the world (GPFLR, 2014). This scenario provides a moment of great challenges and opportunities to implement restoration efforts and to revert the current trend of degradation of earth's ecosystems. Edward O. Wilson stated: *"The next century will, I believe, be the era of restoration in ecology"* (Wilson, 1992).

Despite the increase in the number of restoration efforts in the last decade in tropical forests (Chazdon, 2008), there are some fundamental questions linked to this science that remain unanswered. Amongst them, two are worth highlighting: how to measure the success of ecological restoration? What are the variables to be used as indicators of the success? Answering these questions will shape our understanding of the restoration process, as well as the selection of indicators used to measure success. However, these questions were not conclusively answered due to two main issues. First, restoration ecology is quite a young discipline and, as such, it still presents conceptual issues (Hobbs and Norton 1996; Miller and Hobbs 2007). Conceptual limitation was also attributed to the science of ecology as a whole (Peters, 1991; Shrader-Frechette and McCoy, 1993), even though ecology is currently viewed as robust enough to provide strong patterns and even laws (Dodds, 2009).

Nevertheless, such limitation is one factor that contributes to the gap between scientific knowledge and decision making (Barbosa et al., 2004). Secondly, there is a small number of long term monitoring programs for restoration projects (Bash and Ryan, 2002; Suding, 2011). This state of affairs presents a great challenge for the development and validation of restoration efforts because it coincides with a time of great opportunities for the implementation of restoration projects (e.g. Cabin, 2007). Therefore, there is a need for tools to make restoration ecology a more robust scientific enterprise. The objectives of this chapter are threefold: (1) to present the main ideas for the evaluation of restoration success and the indicators used; (2) to discuss the main advantages and drawbacks of the main strategies of restoration - active and passive; and (3) to emphasize the need for a more widespread use of functional approaches to evaluate success in restoring tropical forests.

8.2 Restoration Ecology: Definitions, Indicators And Strategies

The Society for Ecological Restoration defines restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (SER, 2004: pp 2). They highlight the final objective of a restoration project as the formation of self-supporting ecosystems that are resilient to perturbations without human assistance. However, as discussed by Ruiz-Jaen and Aide (2005), the main issue is to know how we reach this objective. The Primer of Restoration Ecology (SER, 2004) suggests a list of nine attributes to be considered when evaluating the restoration success of ecosystems: (1) similar diversity and community structure in comparison with reference sites; (2) presence of indigenous species; (3) presence of functional groups necessary for long-term stability; (4) capacity of the physical environment to sustain reproducing populations; (5) normal functioning; (6) integration with the landscape; (7) elimination of potential threats; (8) resilience to natural disturbances; and (9) self-sustainability.

Restoration success can be summarized by three general indicators (see Ruiz-Jaen and Aide, 2005a; Wortley et al., 2013): (1) diversity and abundance; (2) vegetation structure; and (3) ecological processes (see Box 1). A measure of success can be obtained by comparing one, two or these three indicators with reference ecosystems (White and Walker, 1997; SER, 2004; Ruiz-Jaen and Aide, 2005b). A reference ecosystem is considered as the target of restoration and should represent the community structure, species composition, and ecosystem functioning prior to a disturbance (White and Walker, 1997). This perspective, known as the recovery paradigm (see Suding, 2011), assumes that restoring communities present a directional trajectory towards a stable state (e.g. old-growth forest). Due to the generality of these three indicators, there is a need to specify what will be in fact measured in the field and the biological meaning of these measures to restoration success.

Diversity indicators are those related with species (or any other operational taxonomic unit) richness and abundance at different trophic levels. Vegetation structure can be characterized by measuring its cover biomass, basal area, leaf area index, and dominance of different ecological groups (e.g. pioneer and shade tolerant species), at different layers (herbs, shrubs, trees). Ecological processes are those directly related with ecosystem functioning (Ruiz-Jaen and Aide, 2005a) such as: nutrient cycling, carbon flux, seed dispersal, pollination, herbivory, etc. Thus, the measurement of this set of indicators provides a detailed diagnosis of the restored community (Elmqvist et al., 2003; Ruiz-Jaen and Aide, 2005b), and allows checking for the need of new interventions. However, because ecosystems are dynamic, a single measurement in time is of limited power to evaluate the success of restoration (Parker, 1997). Long term monitoring is crucial to understand the successional trajectory of a community (e.g. Parker, 1997; Zedler and Callaway, 1999; see also chapter 3) and to develop predictive models (Anand and Desrochers, 2004; Tucker and Anand, 2004; Peng et al., 2010) (Fig. 1). Severely degraded ecosystems can arrest ecological succession and lead to alternative stable states (Suding et al., 2004) (Fig. 1). When stable states that differ from the reference system are reached, the actions required to return the system towards the planned trajectory can be more complicated when compared to that required for the initial degraded state. Thus, understanding how these indicators vary in time is of fundamental importance in order to: 1) choose appropriate restoration strategies (e.g. passive or active restoration); 2) select species to be used; 3) characterize the successional trajectory and ecosystem resilience (Suding et al., 2004).

Box. 1 - Measuring restoration success

An important aspect in measuring restoration success is the definition of the indicators (diversity, vegetation structure and ecological processes) to be quantified. There are two review studies that analyzed how restoration success has been measured in terrestrial ecosystems (Ruiz-Jaen and Aide, 2005a; Wortley et al., 2013). The results showed that only 15% of published data measured restoration success (Ruiz-Jaen and Aide, 2005a). Most of these studies were done in North America (53%), whereas the continents with higher extents of tropical forests (South America, Africa and Asia) encompassed only 12% of the studies (Ruiz-Jaen and Aide, 2005a). These results emphasize the need to increase the research characterizing the success of ecological restoration, especially for tropical forests. Moreover, it is noteworthy that the time interval used to evaluate success is about 1-15 years in 71% of the studies (Wortley et al., 2013). This short period can be a limiting factor to observe changes in diversity, vegetation structure and ecological processes, especially in heavily degraded sites (Guariguata and Ostertag, 2001; Holl, 2007; Dias et al., 2012). The most used indicator was diversity (29%; Wortley et al., 2013; Fig. 2). Only in 11% of the studies the three indicators were evaluated altogether (Fig. 2). This is critical because it shows that the overall picture about measuring restoration success around the world is incomplete, and it is not clear whether we are measuring success in a reliable and efficient way for most of the restoration initiatives.

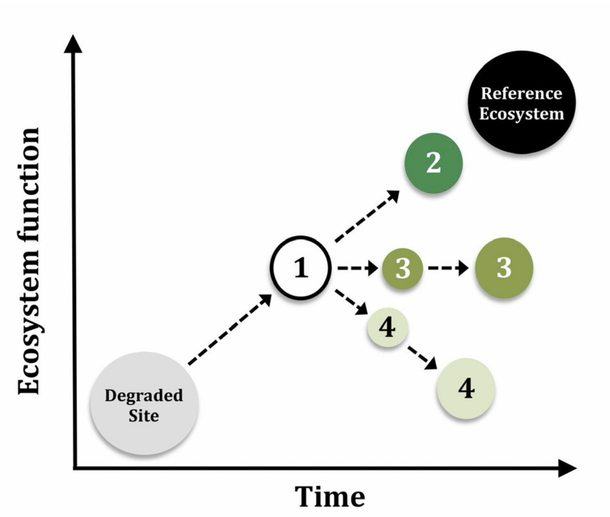

Figure 1: Hypothetical successional trajectories in degraded ecosystems. 1 – Early successional stages with the reestablishment of key aspects of ecosystem function (e.g. biomass increase, biodiversity and nutrient cycling). This stage represents the intermediary stages of a successional trajectory towards the Reference Ecosystem; 2 – The progressive successional trend is maintained and it is expected that ecosystem function will be similar to the Reference in the next years. 3 – This case represents the establishment of an alternative stable state (see Suding et al., 2004). The main drivers of this process are disturbance rate, presence of invasive species, and drastic changes in abiotic conditions (e.g. soil conditions and land use changes). 4 – Retrogressive succession leading the ecosystem to a degraded state. In general, these events are related with perturbation (fire, soil erosion, hurricane, etc) and competitive exclusion driven by invasive species. All these factors act as important barriers for natural regeneration.

Restoration strategies can be classified into two groups: passive and active (Holl and Aide, 2011). Active restoration is made by using interventionist techniques, such as tree plantations, seeding, topsoil application and artificial patches (Parrotta and Knowles, 2001; Camargo et al., 2002; Zanini and Ganade, 2005; Rodrigues et al., 2009; Dias et al., 2012). Plantation is the most common approach of active restoration in tropical areas (Rodrigues el al., 2009) whereas passive restoration occurs through natural regeneration (Holl and Aide, 2011). On one hand, the increased forest cover in some tropical areas (see Aide et al., 2012), also known as forest transitions, is a demonstration that passive restoration has great potential due to low implementation costs (Holl and Aide, 2011). Forest transitions in the tropics result from rural-urban migration, and consequent abandonment of agricultural lands (Aide and Grau, 2004). On the other hand, in some areas where the degradation processes were more intense, active restoration techniques may become necessary. Therefore, land use history, surrounding matrix, types of disturbances, and the presence of natural

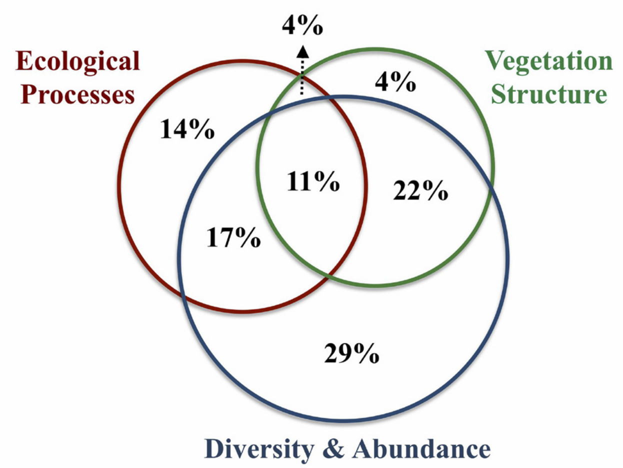

Figure 2: Venn diagram showing the three indicators (vegetation structure, diversity and abundance, and ecological processes) used to measure restoration success in terrestrial ecosystems and their relative proportions (modified from Wortley et al., 2013). Diagram was calculated in R (R Core Team, 2013) using the venneuler package (Wilkinson, 2011).

regenerating native species should be considered when deciding which will be the best restoration strategy in a given site, whether active or passive (Holl and Aide, 2011). The understanding of the regeneration process in time is crucial to predict the restoration success, regardless of the chosen strategy.

The factors affecting natural regeneration, i.e. passive restoration, can be summarized in: intensity and frequency of disturbance (Pickett and White, 1985), land use history (Guariguata and Ostertag, 2001; Holl, 2007), and dispersal limitation (Tabarelli and Peres, 2002; Pereira et al., 2013). Moreover, the interaction among these factors has been pointed as the main cause of the large observed variation in species richness and abundance, basal area, and nutrient cycling in tropical secondary forests (Brown and Lugo, 1990; Guariguata and Ostertag, 2001). Secondary forests originated from abandoned pastures can take 20 to 60 years to reach similar values of species richness and biomass from that found in old growth forests (Guariguata et al., 1997; Finegan and Delgado, 2000; Letcher and Chazdon, 2009). However, the recovery of species composition or groups of species (endemic and non-pioneer species) can take more than 100 years (Finegan, 1996; Liebsch et al., 2008). For example, abandoned pastures affected by fire in Brazilian Atlantic forest show low species richness, high dominance and the presence of invasive grass species, even after 20 years since the last fire event (Fig. 3c). Thus, a precise diagnostic analysis in the field is of fundamental importance to allow proper decision making about the effectiveness of passive restoration initiatives.

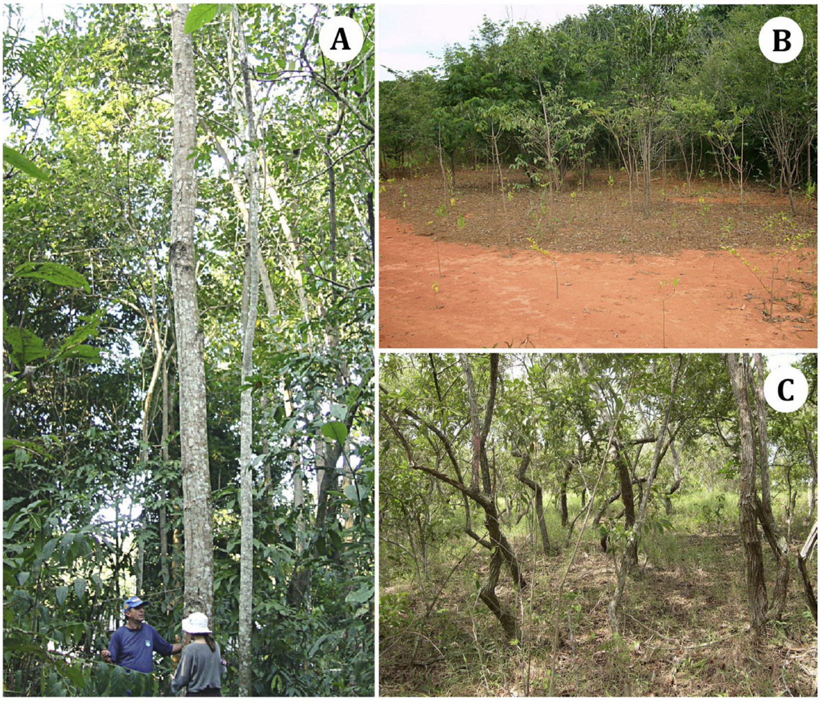

Figure 3: Restoration projects in tropical forests using different strategies (passive and active restoration) in contrasting environmental conditions. (A) – Plantations of native-tree species in Brazilian Atlantic Forest recovered vegetation structure in a short term (11 years), and presented increased species richness. These results were mainly associated with the presence of zoochorous species in the overstory and fruit availability for frugivores (see Sansevero et al., 2011). (B) – Restoration of an Amazonian flood-prone forest (Igapó forest) affected by deposition of bauxite tailings (see Dias et al., 2012. The use of nucleation techniques, as litter addition, promoted plant growth, seedling abundance and species richness. (C) – Abandoned pastures subjected to fire events in Brazilian Atlantic Forest. Even 40 years of abandonment and 15 years since the last fire, these communities demonstrated a low species richness (12 species / 0.18 ha), high dominance (90% of all trees) of fire resistant tree species and presence of invasive grasses in the understory (Sansevero, 2008). In this example, passive restoration alone was ineffective due to community resilience loss. (Photos: J.B.B. Sansevero).

Active restoration (e.g. tree plantation) is an efficient strategy for areas subjected to drastic abiotic changes (Rodrigues et al., 2009; Holl and Aide, 2011). Plantations of tree species can catalyze forest succession, increasing species richness, improving soil fertility and restoring ecological interactions (Parrotta et al., 1997; Harrington, 1999; Ruiz-Jaen and Aide, 2005b; Sansevero et al., 2011; Suganuma et al., 2014). Abundance of tree/shrubs species in 11 years old forest plantations can reach similar values to that of old-growth forests (Sansevero et al., 2011 – Fig. 3a). Natural regeneration can contribute in more than 40% to total basal area in 11 years old forest plantations

(Sansevero et al., 2011), and with 69% of tree species abundance in 28 years old plantations (Pulitano and Durigan, 2004). Therefore, even in planted forests, natural regeneration has a key role in the recovering of vegetation structure. Nevertheless, the effects of the number of planted species in the restoration success are debatable due to the large variation in responses showed by different studies (Lugo, 1997; Aronson et al., 2011). Despite the importance of biodiversity in ecosystem functioning and stability (Naeem, 1998), forest plantations with low diversity (1-10 tree species) can lead to a rapid recovery of species richness and abundance (Silva-Junior et al., 1995; Lugo, 1997; Ruiz-Jaén and Aide, 2005b). This is mainly due to the fact that restoring communities are affected by external factors as the distance to forest fragments, differences in soil attributes, and climate conditions. Overall, the interaction among these different factors and the chosen strategy, whether passive or active, can produce very different responses leading to uncertainty about successional trajectories and the success of ecological restoration.

The recovery of vegetation structure is faster when compared to the recovery of species diversity and ecological processes for both, passive and active strategies (Guariguata et al., 1997; Suganuma et al., 2014). However, it is noteworthy that the high spatial and temporal variation in species diversity and composition of tropical forests (Leigh et al., 2004) is a complicating factor in determining the reference ecosystem (e.g. Suganuma et al., 2013), because of the increased analytical complexity required. Consequently, there will be no reliable indicators to evaluate the success of the restoration efforts due to such variation. In order to solve this limitation, two solutions have been proposed. The first one is based on using a range of values measured in several reference ecosystems, for a given indicator, instead of single measures. However, such approach is unsatisfactory for indicators with high variation (e.g. percentage of zoochorous plants – 47% – Suganuma et al., 2013). The second possibility is to use integrative measures of vegetation structure, diversity and ecological processes as proposed by functional approaches and that have seldom been used in restoration ecology (Aerts and Honnay, 2011; Cadotte et al., 2011).

8.3 How Functional Ecology Can Contribute With Restoration Ecology?

The functional approach is represented by the set of species' traits in a given community and the relationship between these traits and environmental conditions (Solbrig, 1992; Weiher et al., 1999; McGill et al., 2006). These traits can be morphological, physiological or architectural and can be classified into response or effect traits (Lavorel and Garnier, 2002; Cornelissen et al., 2003). Response traits are those related to dispersal capacity, establishment and persistence (see Weiher et al., 1999). Therefore, functional traits are likely to affect the performance of a species (usually abundance) in response to changes in environmental conditions (e.g. climate, soil, landscape and disturbance) and biotic interactions (e.g. competition, facilitation) (Díaz and Cabido, 2001; Moran and Catterall, 2009; Garbin et al., 2014). Effect traits are those related with ecosystem productivity, nutrient cycling, carbon storage and resource availability (see Finegan et al., 2014). The distribution, abundance and diversity of functional traits (functional diversity) directly affect ecosystem functioning (Díaz and Cabido, 2001). Thus, restoration ecology can greatly benefit from a functional approach (Gondard et al., 2003; Cadotte et al., 2011). It can be used to select species for restoration projects, and as a tool for measuring restoration success. This can be done by comparing the functional structure and diversity of the restoring community with that from reference systems.

Table 1 presents examples of functional traits with their responses and effects in plant communities. Response traits provide an important tool when restoring plant communities because they allow understanding plant survival and development under field conditions, especially for seedling planting (Grossnickle, 2012). The selection of traits associated with higher survival and optimal seedling development can help choose adequate plant species and increase the probability of success of the restoration project. For example, a seeding experiment in Amazon showed that species with higher seed mass were positively associated with germination and seedling survivorship (Camargo et al., 2002). The same functional approach can be used to maximize desirable effects in the communities being restored. For example, the provision of ecosystem services is positively associated with plant functional diversity (see Díaz et al., 2007). Moreover, ecosystem resilience and stability could also be analyzed through this same conceptual model.

The examples shown in Table 1 are a small sample of functional traits that can be used to assess functional responses or effects in ecosystems (see Pérez-Harguindeguy et al., 2013). Considering the large number of possibilities, trait selection is not trivial (Violle et al., 2007), but it should consider practical aspects as easiness of measurement (soft and hard traits – Lavorel and Garnier, 2002), and the responses and effects of the traits in face of environmental variations (Violle et al., 2007; Pillar et al., 2009). However, trait variation alone is not sufficient to understand ecological patterns. What makes a trait functional is how it impacts fitness through its effects on growth, survival and reproduction (Violle, 2007), and how such variation in both trait and performance relates to environmental gradients (e.g. Pillar et al., 2009). Nevertheless, the importance of a given trait can vary in response not only to environmental conditions, but also in response to interaction patterns with other species (Garbin et al., 2014). Thus, the next step in order to strengthen the use of functional approaches in restoration ecology is to consolidate a list of functional traits capable to explain key processes for restoration ecology (e.g. Funk et al., 2008), such as: seedling survival, growth, disturbance tolerance, plant-plant (e.g. competition, facilitation) and fauna (pollination, dispersal and herbivory) interactions, nutrient cycling and carbon storage.

Table 1: Functional traits relevant for restoration ecology with their respective responses and effects in ecosystems.

Functional traits	Response and effects in ecosystems
Dispersal syndrome; and Seed mass	Dispersal ability (a,b); Spatial distribution patterns (c); Resource availability for frugivores (d); Seed mass – positively associated with seedling establishment (e)
Leaf dry matter content (LDMC)	Correlate negatively with potential growth rate (a); Resistance to physical hazards (e.g. herbivory, wind, hail) (a); Nutrient cycling - leaves with high LDMC also tends to decompose more slowly than that from leaves with low LDMC
Stem-specific density	Stem density is related to the growth-survival tradeoff; a low stem density leads to a fast growth whereas a high stem density leads to resistance against pathogens or physical damage. Carbon gain is influenced by species stem-specific density (f);
Bark thickness (and bark quality)	Thick bark provides protection of vital tissues against attack by pathogens, herbivores, frost or drought. In general, this trait has special relevance in trees or large shrubs in environments subjected to fire (a)
Root-system morphology	Root system (depth, diameter, lateral extent, and root biomass) is related with capacity of acquire resources and competitive ability (a)

(a) Pérez-Harguindeguy et al., 2013; (b) Howe & Smallwood, 1982; (c) Seidler & Plotkin, 2006; (d) Hasui et al., 2007; (e) Moles & Westoby, 2004; (f) Shimamoto et al., 2014;

Traits can also be used to classify plants into functional groups, or types (Díaz and Cabido, 2001; Pillar and Sosinski, 2003). Plant functional types can be defined as groups of plant species sharing similar functioning, similar responses to environmental factors and/or similar roles in (or effects on) communities or ecosystems (Pillar and Sosinski, 2003). In high diversity systems, the classification of species into functional groups is a strategy to "simplify" communities in the search for more robust patterns. Using this tool, it is possible to compare communities sharing a small number of species (Díaz et al., 2004), analyze the response to disturbances (Muller et al., 2007), assess the resilience of communities (Laliberté et al., 2010), and predict successional trajectories (Chazdon et al., 2009). The prediction of successional trajectories from species composition has been considered a hard task in tropical forests due to their high variation in species diversity and composition (Finegan, 1996; Chazdon et al., 2007). The application of predictive models to the functional structure of plant communities in restoration allows the estimation of the time needed to restore a given area as well as the trajectory of the system; both are important aspects to properly manage communities in restoration, and to redirect successional trajectories. Moreover, as ecosystem services are the basis of the earth's life supporting systems, effective restoration has crucial implications for human wellbeing.

8.4 Conclusions

Two contrasting features can describe the current scenario for tropical restoration ecology. First, there is a challenge imposed by the drastic changes and degradation of ecosystems. Secondly, there are also opportunities created by several restoration initiatives at local and global scales. Given the difficulties associated with current indicators of restoration success based on species diversity, vegetation structure and ecological processes, it is extremely timely to consider that functional approaches play an important role in providing reliable and simplified indicators for restoration success. The use of such indicators can catalyze more restoration initiatives, because they offer insurance that such efforts will in fact accomplish their initial goals, as to provide ecosystem services, contribute for biodiversity conservation and increase ecosystem resilience in response to climate change.

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9 Sustainability Indicators In Brazilian Cattle Ranching

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9.1 Introduction

9.1.1 Brazil And Global Food Production

Brazil is one of the largest agricultural producers worldwide and agriculture is one of the backbones of the country's economy representing, in 2013, 22.5% of GDP (Cepea, 2014). Historically known for its sugar cane and coffee plantations, Brazil is nowadays a major producer and exporter of a range of agricultural products: fruits, cereals and meat. Brazil is the world's largest net exporter and its share in the global food market is 9.7% in volume and 5.8% of the value of the global food market (WTO, 2014). The country also owns the largest commercial cattle herd with 211 million heads, responsible for about a quarter of the total volume of meat transacted in foreign trade supply. In 2012, the value of production of meat for the domestic market was estimated at around US$ 25.85 billion or 1.12% of the country GDP. According to the Brazilian Institute of Geography and Statistics (IBGE - Instituto Brasileiro de Geografia e Estatística, in Portuguese), despite the world financial crisis in 2008, Brazil registered a record agricultural production, with an unprecedented production of grains that has reached 145 million tons. In 2012, Brazil was ranked as fourth in the global production of coffee and meat (dried) and fifth of sugar cane production (FAOstat, 2014). Furthermore, Brazil is the country with the largest forecasted increases in output over the next four decades (FAO, 2006).

In 2009, agricultural land in Brazil (arable land, under permanent crops and under permanent pastures) occupied 36% of the total land use of the country (calculated as Total Area – Native Vegetation, data from IBAMA, 2012). Arable land and area under permanent crops occupy approximately 9% of the land use of Brazil, which corresponds to 65 million hectares (of which the majority, 57 million hectares is arable land). In terms of occupied area, soybean is the main crop (25 million hectares in Brazil). Although arable land represents a significant proportion of the land, pasturelands occupy approximately 75% of the agricultural area of Brazil (159.8 million hectares, which some 57.6 are of natural pasturelands; IBGE, 2009).

9.1.2 Increasing Future Demands

Much has already been discussed regarding global population increase, concomitant demand increase entailing competition for land and the pressure that the finite global resources experience (Smith et al., 2010; Lambin and Meyfroidt, 2011; Alexandratos and Bruinsma, 2012). The Food and Agriculture Organization estimates that 70% increase in food production is needed by 2050 to feed a global population projected to reach 9.1 billion people, whose increasing per capita income will shift the diets towards meat based (Smith et al., 2010). Many action-oriented initiatives worldwide are currently devoted to tackling these multiple challenges with an increasing manifestation of urgency, given that the global population has recently hit seven billion. Developing countries are bound to face increasing demand for meat driven by adoption of livestock-based diets, a trend that has already been observed in China and Brazil. The latter is predicted to have the biggest increase in meat production driven by both increasing domestic demand and for exports (Alexandratos and Bruinsma, 2012).

9.1.3 Unsustainability Of Brazilian Cattle Ranching

Opposite to western-style intensive agriculture that is often associated with biodiversity loss and environmental pollution, in Brazil extensive low productivity agriculture often leads to environmental degradation. Similarly, Brazilian pasturelands are characterized by low stocking rates and this low efficiency has historically led to deforestation[4] and to other adverse effects on environment such as soil erosion. Indeed, Martha Jr. et al. (2012) reported that the growth of the Brazilian beef production between 1950 and 1975 was primarily explained by the expansion of extensive pastures (86%) with cattle ranching productivity explaining only 14%. Since 1996, total pasture area in Brazil negatively contributed to beef production growth, while productivity gains (increased stocking rates and improved animal performance) accounted for beef production growth. However, pasture expansion continued to account for 5.6% of beef production growth in the North region between 1996 and 2006 and current productivity levels in Brazil are still below its sustainable potential. For example, Strassburg et al. (2014) shows that the current productivity of Brazilian cultivated pasturelands is between 32 and 34% of its potential.

The double strength of Brazil - environmental and agricultural - is mainly due to land abundance rather than a decoupling of intensive use of natural resources and production. Conflicts between production and nature are especially evident in

4 within so called ‘slash-and-burn’ process in which cattle is removed from degraded pasture to the area of recently cut and burned forest

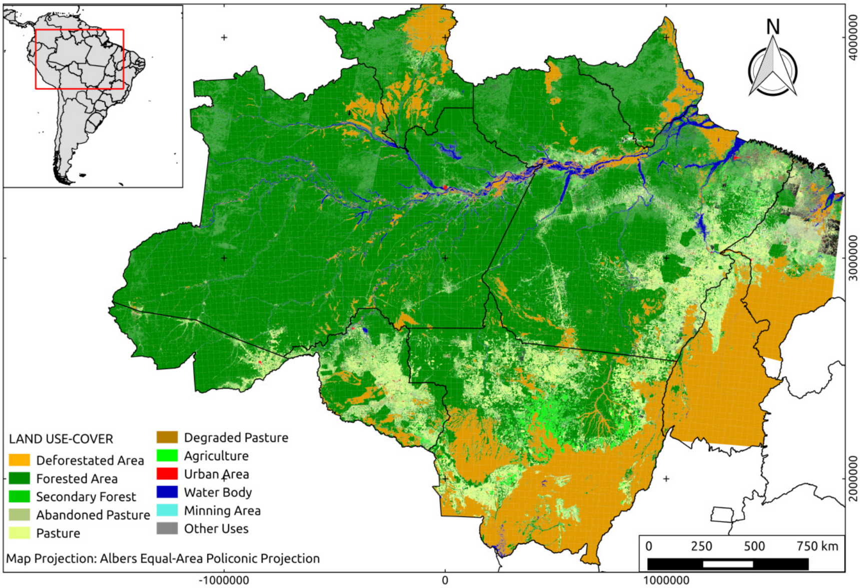

Figure 1: Land use in the Amazon region with visible arc of deforestation.

areas where agricultural border is expanding, such as in the Arc of Deforestation in the Amazon Forest (Fig. 1). Brazil has increased the average agricultural productivity over the last two decades, accompanied by a drastic drop in the rate of Amazon deforestation (INPE, 2014). Yet, Brazil is still the second in the world rank of absolute deforestation, only behind Russia (Hansen et al., 2013). Also in other biomes, such as in Cerrado (Brazilian savannah), the persistence of extensive cattle ranching acts as the main factor of land use and land-cover change (Verburg et al., 2014).

9.1.4 Brazil´s Environment And Recent Initiatives To Protect It

In addition to being one of the largest agricultural producers, Brazil is also the world's largest holder of tropical ecosystems. The most recent Global Forest Resources Assessment (FAO, 2015) shows that the country occupies 6% of global territory but shares 11% of the world's remaining natural vegetation. Brazil is also the most biodiverse country on the planet (56,000 known plant species, versus 29,375 in Indonesia) (UNEP-WCMC, 2010). However, Brazil is the world leader in deforestation (55 million hectares over 1990-2010). In order to protect natural vegetation, the New Forest Code (Brazilian National Law No. 12.651 from May 25th, 2012) has recently been implemented.

According to the new legislation, landowners are obliged to maintain 80% and 35% of forest of their total land area in the Amazon and the Cerrado region, respectively (so called Legal Reserve, *Reserva Legal (RL)*, in Portuguese). Moreover, the natural vegetation surrounding water bodies and other special areas such as mountaintops must be retained (Permanent Protection Areas; *Áreas de Preservação Permanente (APP)*, in Portuguese). The country is also currently developing a National Restoration Strategy aiming to restore around 15 million hectares of native vegetation. In addition, there are other mechanisms to incentivize more sustainable land management, such as various proposed credit lines. For instance, the Brazilian government introduced a Low Carbon Agriculture program ("Programa ABC", in Portuguese), which provides subsidized credit for implementation of climate-smart agriculture techniques, such as no-tillage, crop-livestock systems and the recuperation of degraded pasturelands (BMA, 2010).

The initial uptake of this program was low, but it has increased 50% from 2011 to 2012 (Angelo, 2012). Further, it was not very well distributed: 69% of the funding available between 2012 and 2013 was used by ranchers in the south of the country (Observatório do Plano ABC, 2013). This is partially because many small and medium producers from the Amazon region have limited knowledge regarding application process, such as requirements of documents. As a consequence, these farmers often do not apply for a loan, and if they do, they are unlikely to be awarded one (Cohn et al., 2011). There are currently several interventions being developed in the Amazon region in order to help producers with the access to the ABC program. The government also considers improving old and introducing new credit lines to promote sustainable cattle ranching, by supporting the implementation of good agricultural practices that require machinery and other inputs.

9.1.5 Sustainable Intensification Of Brazilian Cattle-Ranching Systems

Pasturelands, on account of their relative extent both globally (2.8 billion hectares versus 1.5 billion hectares of croplands, Goldewijk and Ramankutty et al., 2004) and in Brazil (159.8 million versus 65 million hectares of arable land), have received increasing attention in the context of protecting nature (Tilman et al., 2002; Bowman et al., 2012; Barretto et al., 2013). Recent studies show that pasturelands productivity is below its sustainable potential, and that increasing productivity in the areas under current pasturelands in Brazil is key to reconcile future increased food production and conservation (Bowman et al., 2012; Strassburg et al., 2014). Strassburg et al. (2014) shows that through sustainable intensification of cattle ranching it is possible to triple the productivity of existing pastures and thus meet demand for meat until 2040 (including for exports). Sustainable intensification, in essence, means increasing food production in existing farmlands without increasing pressure over the environment, and not undermining capacity to continue producing food in the future (Royal Society of London, 2009; Foresight, 2011).

Environmental, social and economic factors associated with technology access are determinant of both the dynamics and productivity levels of cattle production systems in different regions of Brazil. Indeed, the level of intensification of the system varies according to the environmental conditions in a biome, as well as depending on the size of farm (small-holders, medium and large). It is particularly important that sustainable intensification of smallholder livestock systems favors their social welfare, and it should be developed in regions with higher production potential and where the environmental conditions favor intensification. Further, for intensification to be accompanied by reduced deforestation, other initiatives such as legislation and capacity building must be put in place (see also section 2.2.7).

Sustainable intensification is already being observed in some regions in Brazil. For instance, Valentim and Andrade (2009) reported that between 1975 and 2006 the proportion of the total area of cultivated pastures increased from 24% to 56% with concurrent increase of 83% in pasture stocking rates. In the same period, Brazilian cattle herd increased by 102%. The wide adoption of improved grass cultivars of the genus Brachiaria and Panicum developed by EMBRAPA (Brazilian Corporation of Agricultural Research), mainly in the Cerrado and Amazon biomes, was one of the main factors accounting for increasing pasture-based cattle productivity in Brazil (Valentim and Andrade, 2009).

9.1.6 Selected Indicators For Sustainable Cattle Ranching In Brazil

9.1.6.1 Permanent Preservation Areas (APP) And Legal Reserves (RL)

Deforestation alters the water cycle, both by reducing evapotranspiration processes and by changing the infiltration of the water through soil profile (elevates surface runoff). This process changes the water balance - the balance between the water leaving the system in liquid form and in gaseous form. Among the areas that are considered Permanent Preservation Areas, the most impacted by livestock activity are the riparian zones (Kalif, 2007). According to the forest code, these areas, must be kept intact in order to preserve water resources. Although, APP width defined by law does not automatically guarantees provisions of all ecosystem services of riparian zones, the existence of APP, respecting the dimensions and features of the legislation, guarantees the preservation of a significant portion of the riparian forests.

9.1.6.2 Water Quality

Pollution level of water resources with nitrogen and other macronutrients is one of the sustainability indicators of cattle-ranching systems. Water quality is both influenced by remaining natural vegetation that functions as a natural filter as well as by appropriate management of the farm that prevent soil erosion (see also subsection 2.2.3). The water quality of streams, reservoirs, lakes, ponds, springs may be affected in several ways by agricultural practices. For instance, the extensive livestock farming usually suppresses the riparian forests to provide the animals' access to water. As a result, it facilitates runoff of pesticides and fertilizers used in agricultural fields and pastures. Nutrient excess may result in eutrophication and in the loss of water quality.

9.1.6.3 Soil Erosion

The level of soil erosion depends on land management and can result in either lack or excess of nutrients, loss of soil texture and loss of organic matter. According to Nascimento et al. (1994), the degree of soil erosion is directly related to its stability, the decrease in productivity and consequent pasture degradation. According to Müller et al. (2004), degraded pastures are one of the major problems of land use in the Amazon, forcing ranchers to clear forested areas.

Soil physical properties can also be influenced by inappropriate land use, which in turn may lead to soil compaction and consolidation. Soil compaction can be caused by cattle trampling and may influence nutrient uptake and lead to low water retention and accessibility. Finally, it also facilitates runoff of rainwater, raising the level of streams and rivers. However, it can be accompanied by an increase in the total amount of sediments deposited in the water, changing immediately the water turbidity (an indicator of the amount of particulate matter). Pesticides are also carted in excess which can cause toxicity in aquatic organisms within these particles, compromising water quality.

9.1.6.4 Landscape Connectivity

The farmer's decision on where to allocate remaining vegetation within the Legal Reserve is of utmost importance for landscape planning at larger scale. Its prioritization can contribute both to biodiversity conservation and to avoid environmental disasters in susceptible areas. This could be achieved by the creation of ecological corridors and the reestablishment of environmental services.

Studies about ecosystems fragmentation effects come from the "Island Biogeography Theory" (MacArthur and Wilson, 1967), which focuses on the effects of size and distance of oceanic islands as a direct influence on the probability of success or failure for species dispersion, influencing their diversity in a given island. Conservation biologists apply this theory to continental forest areas, considering its shape, spatial characteristics, presence of corridors and the structure and composition of the surrounding environment. Thus, landscape connectivity may also represent an indirect measurement of ecosystem integrity and its benefits to biodiversity and other ecosystem services.

One concept for connecting fragments is through ecological corridors, which may be a continuous strip of forest that connects one fragmented area to another allowing an organism to "travel" from one fragment to another (Loney and Hobbs, 1991; Hobbs, 1992; Simberloff et al., 1992; Bennett, 1997; Puth and Wilson, 2001). The most common examples of ecological corridors are historically focused on mammals and birds. However, more recently, corridors have been developed for plants and invertebrates (Benninger-Truax et al., 1992; Macedo, 1993; Micheli and Peterson, 1999; Kageyama and Gandara, 2001). The Permanent Preservation Areas (APP) could also fulfill the role of ecological corridors.

9.1.6.5 Landscape Flammability

The flammability of an environment is its propensity to fires. The flammability of the landscape consists of a mosaic of environments including anthropogenic ones that can facilitate the occurrence of large-scale fires. This propensity of flammability in tropical humid landscapes that have suffered anthropization depends on the existence of fuels (flammable material) and on drought events. The increase of deforestation in Amazon, coupled with increased logging, in addition to climatic events of drought and careless use of fire have created highly flammable environments in the Amazon landscape. Indeed, fire density in the Amazon region (Fig. 2) is directly related to land cover (Fig. 1) wherein the areas of high fire intensities (red color) are located in deforested area as well as in areas occupied by pastures.

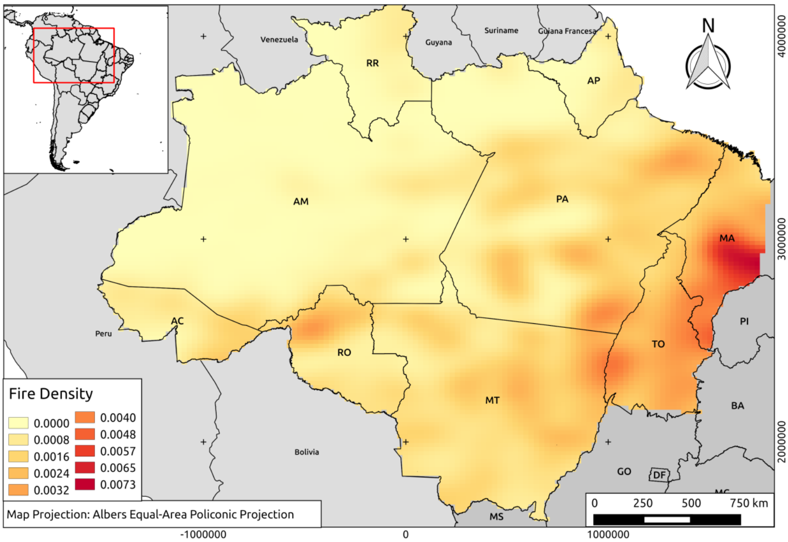

Figure 2: Density of fire occurrences between 2014 and 2010 (fire foci per m²) in the Amazon region.

9.1.6.6 Intensification Level (Productivity)

The level of intensification (measured, for instance, as animals or animal units per hectare) is a particularly relevant indicator when considering extensive pasturelands in Brazil. As discussed above, in Brazil, extensive pastures are often (although not always) associated with environmental degradation and socio-economic unsustainability. Sustainable intensification can be, therefore, perceived as a solution for increasing production (via productivity increase in the area already occupied by agricultural production), avoiding expansion into new areas and sparing native vegetation in Brazil (Latawiec et al., 2014). It should be noted that in the Brazilian context intensification to semi-intensive cattle-ranching system is used, rather than truly intensive (confined) as historically open space, land-based cattle ranching has been used.

If land sparing is performed in a spatially intelligent way that leads to improved landscape connectivity as discussed in section 2.2.4., it may lead to sustainability at a higher than farm level and contribute to regional sustainability. However, some authors point out that pressures over the environment are not necessarily mirrored by the level of farm inputs (Gaudino et al., 2014; see also chapter 10). They conclude that the real pressure on the environment from a particular farm should be evaluated on the basis of the measured impacts it inserts on the environment rather than extrapolated from the level of inputs. In fact, intensive or semi-intensive cattle ranching can lead to negative impacts on the environment. An example within discussions on sustainable intensification is the risk of the so called 'rebound effect'. Rebound effect is a classic economic phenomenon where increased production does not lead to diminished demand for a resource (in this case land) but rather opposite, and results in higher demand (Lambin and Meyfroidt, 2011). In that sense, instead of desired land sparing for nature, production intensification may lead to deforestation.

Another factor that should be taken into account when evaluating sustainability of a cattle-ranching system is whether any action attempting to increase sustainability in one farm does not lead to unsustainability elsewhere (in a form of 'leakage'[5]). So, if actions towards sustainability are to result in best positive outcomes, a range of conditions must be provided. For instance, appropriate legislation, increased governance at state and local levels, complementary measures to foster land sparing and avoid rebound effect, use of geotechnologies and geographic information systems for monitoring (see also section 2.2) should be put in place. These initiatives should also be accompanied by credit access, which would assure efficiency of production and improve ability to provide good quality beef all year round within sustainable production systems. In addition, clear tenure arrangements (avoiding land speculation), and agricultural zoning may aid better land-use planning and

5 Leakage of agricultural or logging activities means displacement of these activities to other areas where relevant monitoring and law enforcement is not in place.

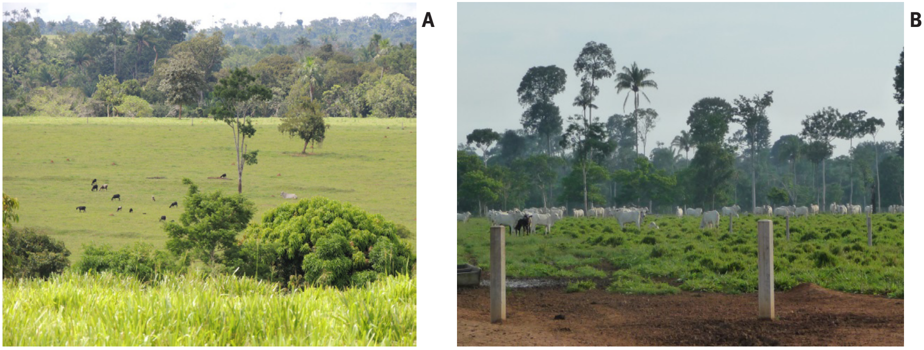

Figure 3: Levels of intensification of Brazilian cattle-ranching system. Panel A shows an extensive (and degraded – visible piles from termites) cattle ranching (photo by Bernardo Strassburg); panel B shows a farm undergoing a process of sustainable intensification (photo by Agnieszka Latawiec).

help maximize the positive impacts of intensification. Finally, the adoption of Good Agricultural Practices (Boas Práticas Agropecuárias, in Portuguese) that may catalyze to sustainable intensification should be promoted. Extension and capacity building should be provided and appropriate management and compliance with regulation regarding the use of agrochemicals, incentives to diminish use of fertilizers and to use them adequately, monitoring and control of farm inputs and outputs should be controlled. It is essential that farmers obtain long-term sustainable agriculture and it is fundamental to engage them within innovation outreach and research and development activities (MacMillan and Benton, 2014).

9.1.6.7 Job And Income Generation

Job and income generation are crucial socio-economic indicators considered within Brazilian cattle-ranching systems. As discussed above, extensive cattle ranching makes little sense from the economic point of view, yet has been in place for decades. There is some contradictory evidence whether sustainable intensification of cattle ranching is leading to job loss or gain. It is possible that due to increased complexity of farm management, the producers will need additional (often skilled) new workforce on theirs farms. On the other hand, intensification is often associated with mechanization of production and higher specialization, which is often associated with lower demand for workforce (Latawiec et al., 2014). However, although on farm the demand may be lower, there might be increased overall employment in the agricultural sector (including for example machines production and urban employment). From the sustainability perspective, job creation is only sustainable if it lasts, even if only associated with capacity building. Therefore, technical know-how must be provided (rural extension), new workforce must be trained or re-trained (capacity building) and adapted to changing socio-environmental landscape. Technical extension, development and dissemination of technologies, training of the farmers and the personnel responsible for technical assistance and knowledge transfer are therefore crucial aspects related to evaluating sustainability of cattle-ranching systems.

9.1.6.8 Animal Wellbeing

Animal welfare is a crucial sustainability indicator of cattle-ranching systems. It means free access to water, comfort, freedom, disease protection and shade, among others. Although the word “intensification” has widely been discussed with respect to animal ethics and welfare, yet the discussion is often complicated by the subjective perception of what actually welfare means. For example, for those more influenced by the so called ´Romantic/Agrarian´ (Fraser, 2008a) world-view, intensification (in the sense of confinement) has inherently negative consequences because animals' natural behavior in natural environments is limited or prevented. On the other hand, others with a ´Rational/Industrial´ (Fraser, 2008a) world-view may tend to overlook these issues and point instead that indoor systems can lead to better health through protection from harsh weather, predators, pathogens and provide the ability to treat animals easily for diseases and parasites. Further, although their movement is limited they may be protected from physical injuries caused by other animals. Even though the research may be heavily influenced by dichotomy of these approaches, there is substantial evidence that some intensive-production strategies, such as highly selective breeding for extreme levels of production may produce congenitally harmed animals or influence wellbeing in other ways (Fraser, 2008b, Dawkins, 2012). For instance, emergence and spread of *Escherichia coli* in Europe is associated with high density of animals, which tends to facilitate the circulation of these and other pathogens (FAO, 2013).

As a part of evaluating sustainability of cattle ranching, it is important to take into account training not only with respect to appropriate handling of the animals but also strong ethical foundations and farmer's understanding that animals experience pain, stress and discomfort. Appropriate governance should be put in place to prevent animal mistreatment, while antibiotics and pesticide use should be controlled. Campaigns to inform wider public of recognizing products where farmers comply with animal and environmental best practices and traceability of products should form part of a broader evaluation of sustainability of cattle ranching at national level.

9.1.6.9 Greenhouse Gases Emissions

Agriculture is a major contributor to greenhouse gases emissions and curbing these emissions has been the focus of a range of initiatives (e.g. above discussed ABC plan; section 1.3). In general, due to rational use of fertilizers and to the shortened age of animals ready to slaughter, semi-intensive well-managed cattle-ranching systems are considered to have reduced emissions as compared with extensive systems (and intensive, beyond the sustainable carrying capacity). Almeida (2010) shows that an intensive system emitted 3.6 kg CO_2 eq / Kg of live weight whereas an extensive system emitted 3.8 kg CO_2 eq / Kg, if only methane emissions were considered (reduced CH_4 emissions might be due to shortened lifespan of cattle). Nevertheless, if other gases are included, the intensive system was responsible for almost as twice as the emissions from the extensive system (Almeida, 2010). Well managed grasses are also known for carbon sequestration when compared to other uses such as degraded systems and primary forests. Segnini et al. (2012) demonstrated that carbon storage in degraded areas, forest areas and managed pasture were of: 102, 118 and 144 Mg ha^{-1}. As discussed in section 2.2.7., if intensification is introduced narrowly it may cause leakage by increasing emissions in other areas.

9.2 Discussion

9.2.1 Sustainability In The Context Of Brazilian Cattle Ranching

The quest for sustainability promotes actions and relations between people and nature, a process whose understanding has become the focus of research with the advent of the concept of Sustainable Development (WCED, 1987). There are many ways to discuss this concept and explain its multiple dimensions: it requires the interaction between different areas of knowledge, the definition of space-time scales, and the recognition of structural boundaries of the system that is intended to be built. In the case of the cattle supply chain in Brazil, as in any other production process worldwide, much of the challenge for the internalization of sustainability is the equalization of these dimensions. It defines a theoretical horizon to be pursued and, consequently, the forms of measurement of such internalization. The measurement of sustainability requires the parameterization of aspects linked to the achievement of that theoretical horizon, resulting in what is known as ‘sustainability indicators’.

For the definition of sustainability indicators of the Brazilian livestock, we must recognize, isolate and parameterize the main problems associated with this activity considering a multidisciplinary perspective and different scales. Associated problems are those that generate externalities and/or are barriers to maintaining a productive activity in the long term. Further, these problems should not be the same for all biomes as well as to the states within these biomes. In general, in the case of the Amazon region, production characteristics and land availability differ from other biomes in the country.

For instance, in the Amazon, livestock is characterized by relatively lower productivity when compared to agriculture. There are some plausible hypotheses. Historically, agriculture was developed as a peripheral activity in the Brazilian economy, mainly based on exports of sugar and coffee. Despite its apparent role to produce meat and leather products for domestic market, there were very low investments in cattle ranching. Furthermore, rural extension was almost absent. During decades, livestock increased due to vegetative growth rather than to planning. From the economics point of view, the reproduction of capital was materialized on livestock itself, focused on attending short-term demands. The factor that could limit this system was the land - especially on farms located far from markets, where inventories grew up faster than flow out rates - but land was abundant, and the cattle management required few labour (Furtado, 2007). This very specific dynamics not only can explain the rationality of the extensive system but also its effects on environmental degradation, including the pastures, explored until exhaustion and replaced by new areas, meaning new deforested areas. In addition, in the last decades, the high opportunity costs of lands close to larger markets promoted the specialization and segmentation of cattle ranching, where pastures were more often replaced by highly-capitalized agriculture. Such phenomenon has been an additional vector of the displacement of cattle ranching to further areas, particularly in the Amazon region where there is a vast availability of land. For instance, in 1974 when the agribusiness capitalization boom happened, the region known as Legal Amazon increased its portion of the national herd from 8.9% to 31%.

While there is scarcity of areas for ranching in the Central-South region, the opposite can be observed in frontier regions such as the deforestation arc. In these regions, forested areas of easy access or close to roads are still abundant and therefore cheap. In 2013, the average price of areas in the regions with these characteristics in the states of Pará, Mato Grosso and Rondônia were of US$ 645 ha-1 ($x = 644.43$, $\sigma = 284.91$; $p=0.05$, $n=9$, our results from data prices from IFNP, 2014). These prices are around 70% lower than restoration costs. Further, because of low land prices, the real revenue for the last decade considering buys and sells was of 5.48% per year, more than threefold the total cattle revenue. Finally, logging activity in these regions generated a total revenue of US$ 2.8 billion in present value in the last decade.

Even though the total amount of credit allocated to investments in cattle intensification has increased dramatically in the last years, studies show that the biggest obstacle for producers to intensify their activities is the difficulty in accessing credit, both regarding resource availability and the required assurances. Therefore, it is easier to understand the rationale of the apparent irrational logic of cattle ranching in Brazil and its consequent low mechanization indexes: high land availability, opportunity costs and obstacles for accessing financial resources. Property legal insecurity due to land tenure conflicts and fraud are hard to quantify, but add up to barriers for the development of a more intensified activity. Considering so, risks of land speculation markets may increase, as well as revenues from logging and the obstacles for financing, resulting in an extensive activity with even lower mechanization indexes. Such abundance of land coupled with an undefined land tenure and the lack of governance in the agricultural frontier results in low investment with low productivity, dominated by the economy based on the production in large areas.

In this context, the two main legal forms for environmental conservation within private rural properties in Brazil - the Permanent Preservation Areas (APP) and the Legal Reserve (RL) - have a very low level of compliance in the cattle farms. This means that if a system of indicators considers compliance as one of the minimum criteria for assessing sustainability, a significant portion of rural properties would not be able to join these indicators. However, such restriction should not be treated as an argument to avoid the incorporation of legal aspects in a list of indicators of sustainability of Brazilian livestock.

9.2.2 Indicators Should Be Transboundary And Consider Timeframe

Division of sustainability into social, environmental and economic aspects in practical terms is useful, for example for monitoring, but it is also often challenging to define whether a particular indicator, let us say, job creation in cattle ranching, should be classified as economic or as social indicator. In order to represent selected indicators discussed in this chapter and to capture a range of policies and initiatives that are influencing performance of these indicators we therefore propose a concept that represents environmental and socio-economic indicators divided into three aspects pertinent to sustainability of Brazilian cattle ranching (Fig. 3). We do not argue that the division in social, environmental and economic pillars of sustainability is incorrect, nor we promote this scheme to be applied for a range of circumstances. We do however believe that it may contribute to a broader understanding of sustainability indicators in the context of Brazilian cattle ranching and aid visualization purposes. It should also be noted that the range of indicators proposed by various institutions (governmental, public, private and NGOs) will influence perception, acceptance and adoption of indicators promoted by them (more discussion on this topic can be found in chapter 2).

The issue of scale is a recurring aspect within discussions on sustainability indicators. The question that often arises is “at what scale something is sustainable?”. This is very important in the context of Brazilian cattle ranching, for example, regarding land spared at the farm level: if not done in the right place, it may have little contribution to sustainability at a regional level (no connectivity). Sustainability can therefore be also measured at different scales: farm (Permanent Protected Areas - PPA), regional (landscape connectivity) and country (law enforcement). For example, the Rural Environmental Registry (Cadastro Ambiental Rural – CAR, in Portuguese; created by article 29 Law: nº 12.727 from 2012) was created aiming at environmental planning and monitoring at the property scale. It is, therefore, a strategic tool (Ferreira et al., 2012) for sustainable development in different scales since it builds a juridical framework that restricts some economic activities while regulates and stimulates sustainable land management.

The Rural Environmental Registry is a self-declaratory registration required for all rural properties. It is determined based on its boundaries and aims to identify Permanent Protected Areas, Areas of Restricted Uses and Legal Reserves. Thus, the

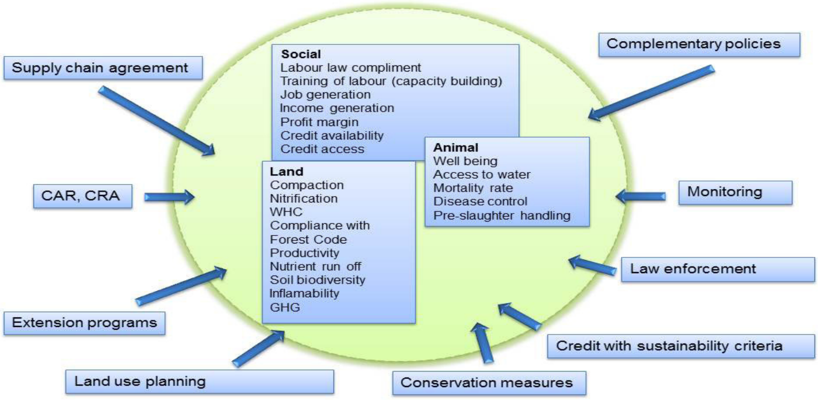

Figure 4: Sustainability indicators framework for Brazilian cattle ranching. Green circle represents indicators at a farm level that are influenced (and can be measured) by initiatives at regional (extension, connectivity) and national level (policy enforcement). This figure presents selected amalgamated indicators discussed in this chapter that are influenced by policies, supply chain agreements, law enforcement etc. Discussion in the text.

owner will be able to assess whether there are any protected areas to be restored, and its location. In case restoration is necessary, it is possible to join an Environmental Regularization Program (Programa de Regularização Ambiental – PRA, in Portuguese) in which mechanisms of the environmental adequacy will be established as well as a sanction to be applied on non-compliance with the agreement signed. In the juridical framework of Legal Reserves (Article 12 of Law nº 12.651), it is possible to have economic uses concomitantly with environmental adequacy provided it does not have negative impacts on native forests. Furthermore, the economic use has to be approved by environmental agencies. By allowing the economic use of Legal Areas by sustainable management, the government may overcome the dichotomy between conservation and productivity, and may therefore increase forested areas.

The same law (Law 12.651) also established the program of Environmental Reserve Quota (Cota de Reserva Ambiental – CRA, in Portuguese) as part of its incentives to environmental preservation and recuperation (Article nº 44). The CRA is a nominal title that corresponds to the area of Legal Reserve that exceeds the area determined by law. This title must be issued by the environmental agency and the property owner must present supporting documentation as a proof of its excess area. It can therefore be used as a trade tool in order to compensate the environmental deficit of other proprieties in the same biome.

The governmental agencies have a critical role in developing supporting programs for the adoption of good practices that conciliate the agricultural productivity with environmental protection. Some examples of policies and programs are Payments for Environmental Services (PES), as well as incentives for restoration with economic benefits. Nevertheless, land owners are only eligible to participate once they have registered in the Rural Environmental Registry. It demonstrates the strategic importance of this Registry together with the need for the development of public policies that foster the application of the Law.

Another pertinent aspect to sustainability is defining what is sustainable for whom and for how long. There is little sustainability in APPs restored in 2014 that will be cut the following year (see also chapter 8). In addition, as discussed above, it is also important to measure pressures on environment rather than levels of inputs. If it is not feasible and impractical, due to high costs, for example, it is important to access additional information that could help us assess sustainability of a system. For example, the mere information on pesticide use is useful; however, the best would be to measure impacts of this pesticide on the environment. If this is not possible it would be important to collect additional information (for example on soil texture to know infiltration rates) and then levels used can be crossed with the soils and pesticide partitioning levels to infer environmental impacts. Table 1 synthesizes positive and negative environmental and socio-economic impacts that should be captured by sustainability indicators for cattle ranching.

9.2.3 Data Availability And Importance Of Monitoring

In order to monitor the change and development of sustainability indicators, data availability is crucial. In the Brazilian context, they are commonly scarce given high costs required for fieldwork. Therefore, the development of Geographical Information Systems (GIS) and Remote Sensing (RS) has been contributing to enable data acquisition for large areas while reducing costs. These tools have been widely used for environmental diagnosis and monitoring. In Brazil, institutions such as The National Institute for Space Research (Instituto Nacional de Pesquisas Espaciais – INPE) and EMBRAPA, among others, have shown different uses of those tools with good reputation (Kintisch, 2007), such as pasture monitoring (Geodegrade[6], MapaStore[7], QualiSolo[8] or GeoRastro[9]).

The GeoDegrade project (from EMBRAPA) aims for the development of geospatial tools that contribute for the identification of degraded pastures. Thus, the main challenges are the assessment and construction of spatially explicit biophysical indicators that allow the identification of different levels of pasture degradation,

6 http://www.geodegrade.cnpm.embrapa.br/web/geodegrade/home
7 http://www.cnpm.embrapa.br/projetos/mapastore/index.html
8 http://www.cnpm.embrapa.br/projetos/qualisolo/
9 http://www.cnpm.embrapa.br/projetos/georastro/

Table 1: Selected indicators of sustainable pasture intensification in Brazil and examples of socioeconomic and environmental impacts that these indicators should capture.

Indicator	Socioeconomic impacts		Environmental impacts	
	Positive	**Negative**	**Positive**	**Negative**
Productivity	Increasing domestic food production and financial resources, improving local economic growth, poverty reduction, improved human welfare, land sparing	Land grabs, loss of alternative land use opportunities, social conflicts, displacement, suppression of traditional way of life, increased inequality, land concentration	Land sparing: more land available for nature conservation, thus biodiversity and ecosystem services provision increase	Rebound effect: increased deforestation, pressure for pasture expansion particularly in areas with less governance, biodiversity and ecosystem services loss
Nutrients	Replacement of nitrogen fertilizer by symbiotic nitrogen fixation by legumes reduces costs and adverse socioeconomic effects from environmental contamination	Health impacts from possible increase in pollution (if fertilizers applied inadequately), increased costs, health risks from intensive agricultural practices	Diminished fertilizer use and risk of water, land air pollution, better efficiency of nitrogen and phosphorus use	Increased environmental pollution
Management of the herd	Improved use of feed-cost efficiency, increased animal productivity, stocking rates improved (increase in pasture carrying capacity)	Investment for insemination or purchase of improved quality bull, may require additional workforce (trained workforce for herd management and supervision while applying vaccines, pest control measures and possible antibiotics), investment in fencing and relevant know-how to manage rotational systems	Less pasture degradation from trampling – less compaction and soil consolidation, less biomass needed, improved nutrient cycle, improved animal welfare, less pollution from pesticides and fertilizers	Movements of machines in the farmland, environmental degradation, human and herd health compromised due to inadequate use of agrochemicals, water, air and soil pollution from concentrated patches of urine, pest control agents, outbreaks of new diseases

Continued **Table 1:** Selected indicators of sustainable pasture intensification in Brazil and examples of socioeconomic and environmental impacts that these indicators should capture.

Indicator	Socioeconomic impacts		Environmental impacts	
	Positive	**Negative**	**Positive**	**Negative**
Jobs and finance	Jobs creation, additional income, development of new skills and expertise, emergence of new businesses, increased income at national level (royalties), empowerment of local community, new social infrastructure	Lack of pasture management monitoring, lack of (trained) workforce, job loss, changes in social balance, may require access to credit or other direct form of subsidy, if pollution/ sedimentation of local water resources occurs it will require payment to remove those adverse effects	Diminished costs from environmental pollution, if farm management performed adequately	Increased environmental costs (costs of remediation), if increased water or land pollution follows
Animal wellbeing	Animal health, good physical and mental conditions due to free access to fodder, clean water and space in natural environment, animals are controlled for diseases and kept free from pests, which in turn diminishes costs	Requires investment in better management, training, equipment, relevant know-how	Semi-confinement may lead to less pollution	Reduced animal welfare (lack of access to free space, if confinement applies), weighting process of animal and selection for further feeding or slaughter may cause animal stress and deteriorate welfare, transportation is a serious issue

Continued **Table 1:** Selected indicators of sustainable pasture intensification in Brazil and examples of socioeconomic and environmental impacts that these indicators should capture.

Indicator	Socioeconomic impacts		Environmental impacts	
	Positive	**Negative**	**Positive**	**Negative**
GHG	Adapting to climate change and mitigate impacts of climate change	If a rebound effect occurs, climate change will impact wellbeing, increased costs and losses for ranching	Increased above and belowground carbon stocks, land sparing leads to mitigation, more intensive systems reduce emissions per unit of beef/ milk produced	Rebound effect can lead to substantial increase in emissions, over-fertilization and mismanagement increases N_2O emissions
Soil properties	Better quality of soils means long-term positive effects to farm, more sustainable and increased productivity, better aeration and nutrient content	Requires know-how and initial investment	Rotational systems (and good soil management) increase soil fertility (nutrient content and availability), reduce soil degradation, increase pasture resilience and pasture persistence, lead to less pollution, more soil organic matter, improved physical properties, higher water holding capacity, bulk density, diminished excess runoff of nutrients and agrochemicals	Soil compaction, erosion, fertility loss, nutrient runoff

combining Remote Sensing data with a validation process that includes field work. The QualiSolo project uses soil parameters as indicators to different land uses (e.g. soybeans, sugar-cane and pasture). By analyzing soil parameters spatially, it is possible to understand the correlation between land management, production capacity and its environmental effects. Finally, in order to foster the sustainable agricultural production, the GeoRastro project proposes mapping all the actors of the supply-chain: the producer has the responsibility of a transparent sustainable management. This way, we assure that the production system is not only environmentally friendly but also sanitary safe. On the other hand, there are some projects that are interested in monitoring and mapping forest degradation, which can be used as a parameter to build a sustainable indicator. The Legal Amazon's Deforestation Rate Program (Programa de Cálculo do Desflorestamento da Amazônia – PRODES, in Portuguese) uses Remote Sensing tools to map and estimate clear-cutting deforestation rate monthly, since 1988. The result is published as spatial data identifying 'forest', 'non-forest' and 'deforestation'. A complementary project is the TerraClass, which uses data from PRODES in order to elaborate a land-use map, allowing the identification of deforestation areas. Finally, the DETER (Sistema de Detecção de Desmatamento em Tempo Real – Live Deforestation Monitoring) alerts and DEGRAD (Degradação Florestal na Amazônia brasileira) projects were developed to support deforestation supervision and control, including the identification of areas in process of deforestation by degradation (not only clear-cutting deforestation). Those projects are of great importance since they are complementary and cover different parameters that allow not only government agencies, but also Non-Governmental Organizations and citizens to elucidate the deforestation process and pressures of land uses over environmental resources.

9.3 Conclusions

This chapter presented a selection of sustainability indicators for evaluating performance of cattle-ranching systems in Brazil. Obviously, it does not discuss all aspects of cattle ranching in Brazil nor is it exhaustive with relation to multiplicity of indicators. We believe however that it contributes to better understanding of complexity of assessing sustainability of Brazilian cattle-ranching systems. The selection and application of sustainability indicators is a subjective decision, often determined by availability of time, financial resources and conviction of what one may consider sustainable. Based on our experience and literature review we draw the following conclusions:

1. It is necessary to understand the context in which to use sustainability indicators in cattle-ranching systems (western-style intensive cattle ranching is often associated with unsustainability, whereas it is extensive cattle ranching in Brazil that contributes to environmental degradation).
2. Indicators should measure pressure - not inputs. Although productivity can be considered a viable indicator of cattle ranching sustainability in Brazil wherein

extensive farming is usually associated with degradation, the adverse effects on environmental, social and economic aspects should be measured on account on impacts rather that inputs (or level of intensification).

3. Spatial and temporal scale matter (sustainable for whom? at the farm, municipality or country level? for how long?) and should be taken into account in any assessment of sustainability of cattle-ranching system.
4. Indicators can be powerful tools to change the system. For example, a farm compliance with animal wellbeing indicators, can change people's perception, preferences as well as impact farm management (see also chapter 2 on how perception influences indicators and vice versa).
5. Sustainability indicators should be used with care and we need various indicators. For example even if we record soil biodiversity at proximate level to an undisturbed system, it may not necessarily reflect vulnerability and resilience of the system. High inflammability, if an incident of fire occurs, may lead to ecosystem collapse, even one with high biodiversity.
6. Indicators are not necessarily good or bad, it is rather how and when they are used. Level of fertilizer is a good indicator if we also have information on soil texture and possible infiltration excess – without this information, the level of fertilizer can give little information on system sustainability.

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10 Sustainability Indicators For Agriculture In The European Union

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10.1 Introduction

10.1.1 Need For Monitoring Of Agriculture Worldwide

Commodities produced in agricultural systems are paramount for the existence of humans. Agriculture is also characterized by multifunctionality, on account of its variety of functions. Multifunctionality of farming can be distinguished into three dimensions including the supply of agricultural commodities, features of rural areas (e.g. landscape management practices, biodiversity values) and management and use of resources (e.g. land, water, capital) (Knickel et al., 2004). Taking into account the demand side, functions of agriculture can be categorized into environmental, economic and social dimensions (Hall and Rosillo-Calle, 1999), which directly correspond to the three sustainability pillars.

Agriculture can have beneficial or harmful effects on the environment. It is crucial to identify opportunities to optimise the linkages between agriculture and the biological and physical properties of the natural environment because of the connection with many vital global environmental issues including biodiversity loss, climate change, desertification, water quality and quantity, and pollution (Van Huylenbroeck et al., 2007). Excessive intensification of agriculture has caused negative effects on the environment and biodiversity (by reducing habitat heterogeneity; Benton et al., 2003), destroyed vast areas of natural habitat and caused an untold loss of ecosystem services. It is also responsible for about 30% of greenhouse-gas emissions (IPCC, 2007; MEA, 2005). Intensification can raise problems not just in relation to landscape and biodiversity but can also affect soil, water and air (COM, 1999). For example, about 1.2 billion hectares (almost 11% of the Earth's vegetated surface) has been degraded by human activity over the past 45 years (Pretty and Koohafkan, 2002). Degradation (particularly through desertification) is a global problem. More than 70% of the world's dry land is affected by degradation caused by overuse or inappropriate use of land (FAO, 2006). In many parts of the EU, agricultural land is under severe threat from alternative land uses and inadequate land use practices. The damaging effects concern: physical degradation (erosion, desertification, waterlogging and compaction), chemical degradation (changes in acidity, salinisation, contamination by pesticides, heavy metals), and biological degradation (changes to micro-organisms and to the humus content of soil) (COM, 1999).

In addition to degradation, intensification can have negative consequences for water. Water efficiency for irrigation is generally very low and there are major concerns regarding depletion of this resource and persistent conflicts over water rights. Where water usage exceeds the rate of replenishment and the water table falls, environmental consequences can be serious such as salinisation by sea water invading the underground supplies, or loss of biodiversity resulting from changes in flow of watercourses. This happened particularly in Mediterranean countries. Irrigation can also result in water pollution because of an increased concentration of pesticides and nutrients in run-off water. Nitrates and phosphates in water which come from farmers' activity may cause eutrophication. Consequences of this are further drinking water contamination which exceed the European Union (EU) norms, elevated levels of nitrates in marine and coastal areas (large areas of the North Sea coast line and parts of the Mediterranean) leading to algal growth and other forms of changes to the ecosystems. This leads to economic losses not only for fisheries and for tourists but also for all citizens – everybody must pay higher prices for drinking water, which has to be purified.

Agriculture is a part of the economy as the cultivation of animals, plants, fungi, and other life forms for food, fiber, biofuel, medicinals and other products are used to sustain and enhance human life (ILO, 1999), and thus it remains a major force in growth of the whole economy. Important determinants of the economic function contain the complexity and maturity of market development and the level of institutional development (Van Huylenbroeck, 2007). Furthemore, farmers activity may contribute to carbon sequestration (West and Marland, 2003), flood control and water conservation (Mitsh and Gosselink, 2000). Social function of agriculture is associated, for instance, with fundamental existence of rural communities. The maintenance and dynamism of rural communities is critical to sustaining agro-ecology and improving the quality of life (and assuring the very survival) of rural residents, particularly of the youth, women, the elderly people. Social viability includes maintenance of the cultural heritage (Van Huylenbroeck, 2007) and might revitalise rural areas (Sharpley and Vass, 2006). In their interesting, case-study based research (N = 24) on agricultural abandonment in mountain areas from different locations in Europe, MacDonald et al. (2000) showed that abandonment is widespread and generally has undesirable effects on the environmental parameters. Moreover, the influence of environmental changes cannot be predicted due to environmental, agricultural and socio-economic contextual factors. The study of socio-economic characteristics reveals that mountain areas are potentially vulnerable to abandonment through high dependence on agricultural employment and small size of their operation, which may reduce viability and the capacity for adaptation. Twenty one out of 24 zones were suffering from some form of abandonment (e.g. reduction of traditional farming practices, generally those associated with livestock practices such as transhumance or hay meadow management).

Environmental, economic and social functions of agriculture are interrelated and in many cases depend on the policy implemented in the local and national level. The outcomes and consequences of intentional human activities and changes in the agriculture sector must be measured and monitored over time to reveal the positive or the negative impact on environment, economy and social issues. Some ideas to improve environmental, economic or social aspects of agriculture may bring short-term disadvantages (such as lower productivity), but long-term benefits (Van Huylenbroeck, 2007). Therefore the role of monitoring and evaluation of the agriculture sector is crucial and requires regular and continuous tracking of inputs, outputs, outcomes, and impacts of development activities against targets. It this way it can be determined whether adequate implementation progress has been made to achieve outcomes, especially towards sustainability goals. Monitoring and evaluating indicators also provide information to improve management and future project implementation (FAO, 2010; see also chapters 3 and 4 for discussion on importance of monitoring).

10.1.2 Different Definitions Of And Approaches To Sustainable Agriculture – What Is Sustainable Agriculture?

In the past, agricultural strategies have been assessed on the basis of a narrow range of criteria, such as profitability or yields. Nowadays agriculture is considered sustainable when current and future food demands can be met without unnecessarily compromising economic, ecological, and social/political needs (AS, 2004). One interpretation of sustainable agriculture focuses on types of technology, especially strategies that reduce reliance on non-renewable or environmentally harmful inputs. These include ecoagriculture, permaculture, organic, ecological, low-input, biodynamic, environmentally-sensitive, community-based, farm-fresh and extensive strategies. There is intense debate, however, about whether agricultural systems using some of these terms actually qualify as 'sustainable' (AS, 2004). It is believed that these strategies may lack adequate scientific knowledge, they cannot be 'scaled up', they are limited in scope and they are incapable of jointly meeting society's demands for food production, livelihood generation, and mitigating environmental degradation (CU, 2014). A second and broader interpretation focuses more on the concept of agricultural sustainability that goes beyond a particular farming system. Sustainability in agricultural systems is viewed in terms of resilience (the capacity of systems to buffer shocks and stresses) and persistence (the capacity of systems to carry on). It implies the capacity to adapt and change as external and internal conditions change. The conceptual parameters have broadened from an initial focus on environmental aspects to include first economic and then wider social and political dimensions (Cernea, 1991; DFID, 2002):

- ecological – the core concerns are to reduce negative environmental and health externalities (in economics, an externality is the cost or benefit that affects a party who did not choose to incur that cost or benefit), to enhance and use local ecosystem resources, and preserve biodiversity. More recent concerns include broader recognition for positive environmental externalities from agriculture. Sustainable agriculture produces not only food and other goods to the market, but it also provides public goods such as: clean water, maintaining biodiversity, carbon sequestration in soils, groundwater recharge and flood protection.
- economic – economic perspectives on agricultural sustainability seek to assign value to ecological assets and also to include a longer time frame in economic analysis. They also highlight subsidies that promote the depletion of resources or unfair competition with other production systems.
- social and political – sustainable agricultural systems may have many positive side effects including helping to build natural capital, strengthen social capital and develop human capacities (Ostrom, 1990; Pretty, 2003). At the local level, for example, agricultural sustainability may be associated with farmer participation, group action and promotion of local institutions, culture and farming communities.

The concept of agricultural sustainability does not mean ruling out any technologies or practices on ideological ideas. If a technology works to improve productivity and does not cause undue harm to the environment, then it may result in a range of sustainability benefits (Pretty, 2008). In that respect, key principles for sustainability according to Pretty (2008, p. 451) are as follows:

- to integrate biological and ecological processes such as nutrient cycling, nitrogen fixation, soil regeneration, allelopathy, competition, predation and parasitism into food production processes,
- to minimize the use of those non-renewable inputs that cause harm to the environment or to the health of farmers and consumers,
- to make productive use of the knowledge and skills of farmers, thus improving their self-reliance and substituting human capital for costly external inputs,
- to make productive use of people's collective capacities to work together to solve common agricultural and natural resource problems, such as for pest, watershed, irrigation, forest and credit management.

10.2 Introduction To The Case Study – EU Agriculture

10.2.1 Origins Of The Concept Of Sustainable Agriculture In Europe

Agricultural policy in the European Union increasingly emphasizes its sustainability. The industrial direction of agricultural development had caused increase in production, yet had affected in a negative way social, economic and environmental aspects, such as decreased biological diversity or increased water contamination with nitrates coming from agricultural sources. This industrial orientation on farming also resulted in additional external costs (e.g. water pollution or land degradation). Therefore, from the beginning of the 90s of the twentieth century changes in the Common Agricultural Policy (CAP) were initiated. The weakness of CAP before 1992 was lack of coherent economic, environmental and social goals. Reorientation and extension of goals was highlighted in three reforms: 'MacSharry reform' (1992) (see more COM, 1991), Agenda 2000 (1999) (see more COM, 1991; BEU, 1997; COM, 1999), and Luxembourg Agreement (2003) (see more EC, 2014a; EC, 2014b).

The MacSharry reform introduced agri-environment programmes. The dual role of farmers has been underlined - firstly food producers and secondly protectors of the environment in the context of rural development (guardians of the countryside). Farmers should be supported as an environmental manager through use of less-intensive techniques and the implementation of environmental-friendly measures. It is worth noting that COM (1991) referred to environmentally sustainable form of agricultural production and food quality. It also relates to 'specific measures on the environment, to be tailored to the situation in individual Member States' (COM, 1991, p. 11).

The overall reason to introduce Agenda 2000 was to prepare Europe's agriculture for the 21st century and enlargement of the EU. The importance of developing, targeting and monitoring agri-environmental indicators has been highlighted. Agri-environmental indicators show developments over time, provide quantitative information and enable understanding complex issues in the field of agriculture and environment. Although Agenda 2000 underlined the objective of food security and the linkage with the safety of the environment, it was still dominated by the instruments of the first pillar – production support.

In 2007 the Commission evaluated the implementation of the CAP reform implemented in 2003 and adjusted it to a rapidly changing environment (see COM, 1999; EC, 2014c). In 2010 after a public debate, the Commission presented a Communication on 'The CAP towards 2020' (COM, 1999). In October 2011 the Commission presented a set of legal proposals designed to make the CAP a more effective policy for a more competitive and sustainable agriculture and vibrant rural areas. Finally, after intensive negotiations between the Commission, the European Parliament and the Council, a political agreement on the reform of the

CAP has been reached on 26 June 2013. The new CAP 2014-2020 focused on the operational objectives of delivering more effective policy instruments, designed to improve the competitiveness of the agricultural sector and its sustainability over the long term. EU agriculture needs to attain higher levels of production of safe and quality food, while preserving the natural resources that agricultural productivity depends upon. In 2011 the European Commission introduced systems to ensure greater environmental protection and management, known as ´greening measures´ (Brouwer, 2006). Under the new European Commission regulations, 7% of farm area will have to be transformed under the protection of biodiversity (see also section 2.3). The EU in CAP 2014-2020 shall endeavour to limit the negative effects of agriculture (water pollution, soil depletion, water shortages and loss of wildlife habitats) and to encourage its positive contributions (climate stability, biodiversity, landscapes and resilience to flooding). The future CAP shall promote energy efficiency, carbon sequestration, biomass and renewable energy production and, more generally, innovation.

10.2.2 What Were The Historical And Recent Trends Regarding Agriculture And Steps Towards Sustainability?

Over the last centuries, agriculture has shaped many European landscapes. After World War II Europe was in the food shortage, so the farming was orientated into intensification of agriculture. High level of financial support favoured intensive agriculture and an increasing use of fertilizers and pesticides, and intensive methods on crop and livestock farms have often led to a loss of biodiversity and increased environmental degradation in many EU countries (Brouwer, 2006). Land, water and air pollution, the destruction of hedge rows, stonewalls, and ditches and the draining of wetlands have contributed to the loss of valuable habitats for many birds, plants and other species. Intensification in certain areas led to an excessive use of water resources and to increased soil erosion (COM, 1999). During the last 25 years, the EU (since 1990) saw the awareness of the crucial role on the sustainable agriculture steadily growing, which is visible in the following CAP reforms.

Currently, European agriculture is characterised by a broad heterogeneity of production systems with wide-ranging geographical features. A general trend includes decline in farm numbers, increased farm size and relatively stable trends on utilised agricultural area (Brouwer, 2006). Moreover, agricultural production is becoming more specialized. Generally two trends dominate: (i) intensification and specialisation in regions with competitive advantages, inducing concentration of production and more homogeneous farming methods, (ii) extensification of production in remote areas with unfavourable economic, social or environmental conditions, leading sometimes to marginalisation and abandonment of production.

Intensification of production is mainly observed in regions where agriculture is most productive. In contrast, marginalisation and abandonment tends to occur in remote areas or on less fertile land where traditional extensive agriculture is threatened by its inability to compete effectively with intensive production in other regions (Brouwer, 2006). According to Baldock et al. (1996), marginalisation occurs in areas where farming ceases to be viable under an existing land use and where other agricultural options are not available leading to land abandonment and driven by a combination of social, economic, political and environmental factors. Regions which are potentially most vulnerable to marginalisation and possibly to abandonment fall into two main categories – regions where extensive systems predominate and those characterized by small-scale agriculture (COM, 1999). Organic farming is another trend observed in EU agriculture. Indeed, in areas with a high proportion of permanent grassland or environmentally sensitive regions organic farming can be an interesting alternative (COM, 1999). In 2012, the area of organic land, the number of organic farmers and the organic market continued to grow in Europe. There were more than 250 000 organic producers in the EU while 320 000 globally (Willer, 2014).

10.2.3 What Is Considered Sustainable Agriculture In Europe?

Sustainable agriculture is often cited as the answer for the question how to produce more food with fewer resources, ensure food security and reduce poverty. It is a challenge not only for Europe, but also for the whole world. Sustainable agriculture is therefore a key for long-term and inclusive growth, especially in developing countries in which agriculture is still the major backbone of the economy (EC, 2012). In this context, sustainable intensification of agriculture to levels that optimize the highest yield with minimum possible adverse impacts on the environment has been proposed as a strategy to reconcile increasing demand for food and protection of natural resources (Godfray et al., 2010; Foresight, 2011). There is a number of studies that demonstrate a yield gap between current and potential sustainable production levels and that achieving these higher production levels would enable to feed future population and conserve natural environment under climate change (Licker et al., 2010; Mueller et al., 2012; Sakschewski et al., 2014, Strassburg et al., 2014; West et al., 2014; Fig. 1). Indeed, because intensification means increasing yield per hectare it may result in land sparing for nature or for other agricultural uses (Balmford et al., 2012). As mentioned in the section above, concurrent to the need for sparing land for biodiversity, the European Commission introduced three ‘greening measures’: establishing Ecological Focus Areas (EFAs) on 7% of farmed area, maintaining existing permanent grassland, and growing a minimum of three different crops on any farm with >3 ha of arable land. To this end, 30% of direct payments to farmers were to become conditional on compliance with these measures.

These greening measures were introduced to address concerns expressed in the latest CAP reform in 2010, in which preserving the environment was outlined as one of the three main challenges (the other two are food security and maintaining the territorial balance and diversity of rural areas). Following a 3-year negotiation, these measures were somewhat melted down and are now set at 5%, instead of 7%, and only on farms with >15 ha of arable land. Countries can further reduce the requirement to 2.5% or lower in some regions. Moreover, EFAs now apply only to roughly 50% of EU farmland and most farmers are exempt from deploying them. Nevertheless, if intensification continues, and evidence from parts of Europe characterized by less intensive farming demonstrates that this process will continue (e.g. Królczyk et al., 2014) combined with land sparing for nature, it should be performed in a way that does not compromise producers' economic returns and increased yields. However, in order to realise the full potential of sustainable productivity increase, complementary policies such as territorial planning should be put in place to both avoid undesirable outcomes of intensification (such as rebound effect – see e.g. Lambin and Meyfroidt, 2011) and to maximise the positive outcomes for biodiversity (establishing protected areas in places that provide landscape connectivity). Furthermore, management of agricultural land designed for sustainable intensification should be performed in ways that reduce negative impacts on the environment (Godfray et al., 2010).

Another approach with the aim to achieve sustainable agriculture and reconcile agricultural production and biodiversity is 'land sharing' (Perfecto and Vandermeer, 2010), at which heart lays coexistence of biodiversity and agriculture on the same land area. Examples of such an agricultural matrix can be found worldwide (Mendenhall et al., 2013) and are common in Europe. Besides biodiversity-related arguments defending land-sharing, proponents of this strategy often bring attention to other ethical and esthetical aspects linked with landscape mosaics. In that respect, intensification is sometimes linked with environmental and social unfairness, while 'spared land' for biodiversity may be unavailable to be appreciated by poorer parts of society. There is vast literature defending these opposite trends, while others show that different approaches may work in different circumstances (Godfray, 2011). It is generally agreed that, in order for land sparing to achieve its intended benefits for people and environment, they must be supported by monitoring and relevant legislation (Phalan et al., 2011; Balmford et al., 2012). Concurrently, Pe'er et al. (2014) in response to dilution of new environmental prescriptions of the EU, proposes actions to benefit biodiversity within a EU legislation scheme that includes allocating sufficient funding and effort within the Farm Advisory System in order to deliver ecological expertise to farmers and provide budget through budget modulation, prioritizing context-specific measures shown to support biodiversity and ecosystem services. They also propose to set clear and measurable targets that are coherent with the EU Biodiversity Strategy (for other recommendations see Pe'er et al., 2014).

Similarly to the dichotomy between land sparing and land sharing, various authors defend different approaches to land management, intensity of agriculture and used inputs. A classic example is organic agricultural production (usually small-scale holder) versus intensive (usually large-scale agriculture, Fig. 2.). Although in general organic agriculture is considered more environmentally friendly, some conventional farmers defend their practices precisely as being sustainable. Furthermore, some authors defend an approach that evaluates sustainability of a system taking into account not inputs and level of intensity but pressures exerted on environment by a farm (see also section 'Discussion' below; Gaudino et al., 2014). Although often driven by pragmatic motives, rather than environmental concern, a producer may have consciousness that his/her practices, if inappropriate, may compromise his/her production (thus profits) in the long term. Seemingly driven by economic benefits, it may lead to sustainable management of the farm and awareness of the impacts on the ecosystem services that a farm provides. Some farmers also claimed (personal communication) that even if not labelling themselves as eco-friendly and openly admitting the priority of economic aspects in their farm management, they do not necessarily behave against the sustainability principles. Quite on the contrary, a good farm manager, independently on being organic farmer or a conventional one, may be aware that excessive inputs are not sustainable for economic and social reasons. For instance, excess of fertilizer is avoided on account of inefficiency of money spent on it, if excess infiltration occurs ('fertilization costs, why use excess?'; personal communication). Similarly, the farmer may be aware of the adverse effect of agrochemicals on his health and his family and therefore opt for rational use of agrochemicals.

Although farm management is perceived through the lens of a business and being able to run life at good level, the environment may benefit as well. Importantly, sustainability is also associated *per se* with long-term thinking and many 'intensive' farmers are aware that if they lead to soil degradation, it will undermine their future production. In relation to 'greening measures', lack of environmental (and biodiversity in particular) consciousness must not necessarily prevent actions towards land sparing, if a farmer is aware for example that pollination is good for yields. For practical reasons, for a large-scale farmer leaving a part of his farm (usually area of lower yields) may also not be a problem (personal communication). We therefore conclude that both organic farming, driven directly by farmer's environmental and social concerns, or rational conventional farming, if applied appropriately, may contribute to sustainability of agricultural systems.

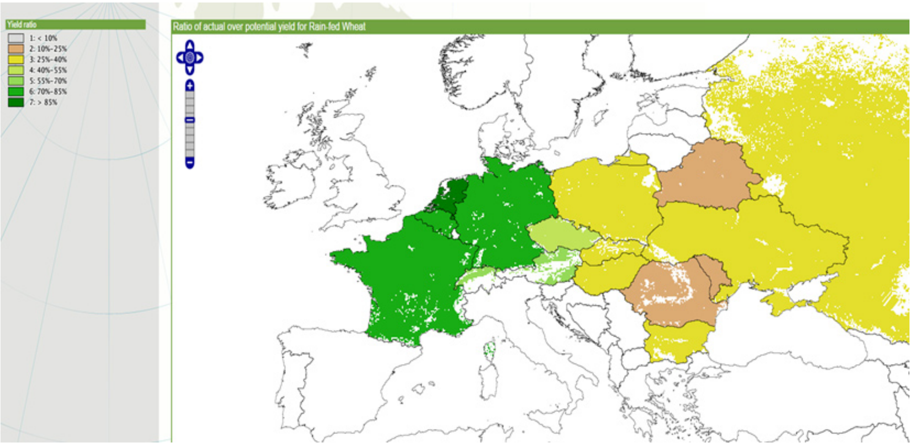

Figure 1: Yield gap between current and potential sustainable wheat production (based on GAEZ). Visible is the large gap for central and eastern European countries highlighting the scope to improve agricultural productivity.

10.3 Which Indicators Are Used Within The EU To Assess Sustainable Agriculture?

A vital step towards making agriculture sustainable is evaluating the effects of different farming systems around the world (Sachs et al., 2010). There have been many attempts to agree at the global scale on a list of sustainable development indicators including agriculture aspects. The results are however not conclusive. There are many different indicators among various organizations such as United Nations, Organisation for Economic Co-operation and Development - OECD, European Union - EU, Food and Agriculture Organization of the United Nations - FAO. OECD developed an overall framework and approach to establish a set of agri-environmental indicators. A pilot survey on 37 (currently 13) agri-environmental indicators in OECD Member countries was conducted in 1995 (FAOSTAT, 2011). The coverage of agri-environmental indicators from the year 2013 in the OECD compendium of agri-environmental indicators is presented in Table 1.

Within the EU, the IRENA (Indicator Reporting on the integration of Environmental concerns into Agricultural policy) project was a joint exercise between several Commission Directorates-General (Agriculture and Rural Development, Environment, Eurostat and Joint Research Centre) and the European Environment Agency, which has led to substantial progress in the development of 35 agri-environmental indicators. Further to the IRENA operation, the European Commission identified 28 agri-environmental indicators (AEI) (UNECE, 2012). Indicators have different ranges based on different approaches to sustainability definition. Only some of the indicators can be compared at the global level. The reason for this is sometimes lack of statistical data and also that many of the interactions between environment and agriculture are not well understood or are difficult to assess and capture in a single framework. Moreover, socio-economic factors are independent of the policy and they can determine changes in farming systems and rural areas and can also affect the environment. A further problem with the current system is that the data collected are rarely comparable across ecological zones because of inconsistencies in methodologies or in the spatial scale at which observations are made (MEA, 2005; IPCC, 2007; McIntyre et al., 2009).

Agri-environmental indicators are a useful tool for monitoring and analysing the relationship between agriculture and the environment and identifying trends in interaction. In January 2000 the European Commission published the communication 'Indicators for the Integration of Environmental Concerns into the Common Agricultural Policy" (COM, 2000). The need for appropriately developed agri-environmental indicators is highlighted. Indicators improve transparency, accountability and ensure the success of monitoring, control and evaluation. Agri-environmental indicators have to assess positive and negative effects of agriculture and should be sufficiently differentiated to be able to capture regional differences in environmental conditions (COM, 2000). Further, they should provide information: on the state of the environment in agriculture; on the wider context, particularly concerning the diversity of the EU's agri-ecosystems; for understanding and monitoring the linkages between agricultural practices and their effects on environment; to support the global assessment process of agricultural sustainability; to assess the extent to which agricultural and rural development policies promote environmental friendly farming activities and sustainable agriculture (COM, 2006).

The basis for an agri-environmental indicator framework is provided by the OECD's DSR (Driving force-State-Response) framework and the European Environment Agency's DPSIR (Driving force-Pressure-State-Impact-Response) framework. 'At the centre of the framework is the current state of the agricultural environment and how this has changed over time. State indicators bring to the fore any undesirable changes which need to be combated (for example, nitrate or pesticide concentrations in water) as well as particularly desirable states which should be preserved (for example, many agricultural landscapes or valuable habitats)' (COM, 2000, p. 9). The second step is to identify negative and positive impacts on the environment by assessing the pressures that have brought about undesirable change and environmental benefits resulting from farming that have helped to preserve or enhance the environment. The next step is to connect these pressures and processes to the driving forces in the economy (farmers' activities) as this where the integration process is applied. In the final step society's response to these issues is monitored (COM, 2000).

Figure 2: Intensive rapeseed production (yield of approximately 4 tonnes per hectare). Four tonnes of rapeseed per hectare is considered the upper limit for sustainable agriculture given high input levels (Królczyk et al., 2014). Photo courtesy of Jan Białas.

In the policy paper Communication COM (2006) 508 final, the European Commission adopted 28 agri-environmental indicators (AEIs) to assess the interaction between the CAP and the environment (COM, 2006). Indicators are identified under the DPSIR (Driving forces - Pressures and benefits - State/Impact - Responses) analytical framework (Table 2). The agricultural DPSIR framework is meant to capture the key 'factors' involved in the relationships between agriculture and the environment and to reflect the complex chain of causes and effects linking these factors (COM, 2006). Actions that are important for the EU economy, policy, environmental and sociological issues, like the IRENA project, are performed by many institutions. Eurostat is

Table 1: Coverage of agri-environmental indicators in the OECD compendium of agri-environmental indicators (adapted from OECD, 2013).

Theme	Indicator title	Indicator definition
I. Soil	**Soil erosion**	1. Agricultural land affected by water and wind erosion
II. Water	**Water resources**	2. Freshwater withdrawals for agriculture
		3. Irrigated area
		4. Irrigation water application rate
	Water quality	5. Pesticide, nitrate and phosphorus pollution
III. Air and climate change	**Ammonia**	6. Ammonia emissions from agriculture
	Greenhouse gases	7. Agricultural greenhouse gas emissions (methane and nitrous oxide, but excluding carbon dioxide)
	Methyl bromide	8. Methyl bromide use, expressed in tonnes of ozone depleting substance equivalents
IV. Biodiversity	**Farmland birds**	9. Populations of breeding bird species that are dependent on agricultural land for nesting or breeding
	Agricultural land cover	10. Agricultural land cover types (arable crops, permanent crops and pasture areas)
V. Agricultural inputs and outputs	**Production**	11. Agricultural production volume
	Nutrients	12. Agricultural nitrogen and phosphorus balances, surplus or deficit
	Pesticides	13. Pesticide sales
	Energy	14. Direct on-farm energy consumption
		15. Biofuel production to produce bioethanol and biodiesel
	Land	16. Agricultural land-use area
		17. Area of certified organic farming
		18. Area of transgenic crops

developing the 28 AEI in cooperation with the EU Member States, the Directorates-General for Agriculture and rural development, and for the Environment, the Joint Research Centre, the European Environmental Agency, as well as with the OECD and the FAO (UNECE, 2012).

10.4 Discussion

There is extensive literature on developing sustainable agriculture. Multiple and sometimes contradictory perspectives are proposed within this academic debate, which is mirrored in the various indicators and frameworks proposed to monitor steps towards sustainable agriculture (Pannell and Glenn, 2000; Piorr, 2003; Haberl et al., 2004; Lomba et al., 2014; West et al., 2014). For example,

Table 2: Agri-environmental indicators proposed by the European Commission in the IRENA project – 28 indicators (adapted from EUROSTAT, 2010; COM, 2006; see more UNECE, 2012).

Domain	Sub-domain	Explanation	Nr	Title
Responses	**Public policy**	Farming is strongly influenced by agricultural and environmental policies and sensitive to input and product price. In addition, changes in farmers' skills, technology , and consumers' and producers' attitudes affect agricultural practices	1	Agri-environmental commitments
			2	Agricultural areas under Natura 2000
	Technology and skills		3	Farmers' training level and use of environmental farm advisory services
	Market signals and attitudes		4	Area under organic farming
Driving forces	**Input use**	A crucial characteristics of different farming systems and indication of farming intensity is the use of inputs (pesticides, fertilisers, energy and water)	5	Mineral fertiliser consumption
			6	Consumption of pesticides
			7	Irrigation
			8	Energy use
	Land use	Land-use change, cropping and livestock patterns influence land-use intensity and trends in the agricultural sector	9	Land-use change
			10.1	Cropping patterns
			10.2	Livestock patterns
	Farm management	Farm management includes rotation patterns, soil cover, different tillage methods and the handling of farm manure	11.1	Soil cover
			11.2	Tillage practices
			11.3	Manure storage
	Trends	Key trends in farming activities at an aggregate level, for instance regional or national, can be expressed in terms of intensification or extensification, specialization, risk of land adandonment	12	Intensification/ extensification
			13	Specialisation
			14	Risk of land abandonment
Pressures and benefits	**Pollution**	Agriculture can lead to excessive nutrient and pesticide residues in soil and runoff to water. It can also lead to ammonia and methane emissions	15	Gross nitrogen balance
			16	Risk of pollution by phosphorus
			17	Pesticide risk
			18	Ammonia emissions
			19	Greenhouse gas emissions
	Resource depletion	Inappropriate use of water and soil in agricultural sector as well as changes in genetic diversity may lead to environmental pressures	20	Water abstraction
			21	Soil erosion
			22	Genetic diversity
	Benefits	Agriculture may provide environmental benefits through management of farmlands of high nature value and via the production of renewable energy sources	23	High nature value farmland
			24	Renewable energy production

Continued Table 2: Agri-environmental indicators proposed by the European Commission in the IRENA project – 28 indicators (adapted from EUROSTAT, 2010; COM, 2006; see more UNECE, 2012).

Domain	Sub-domain	Explanation	Nr	Title
State/ Impact	**Biodiversity and habitats**	Species diversity in farmed areas can be measured via the state of farmland birds	25	Population trends of farmland birds
	Natural resources	The quality of key natural resources needs to be monitored	26	Soil quality
			27.1	Water quality - Nitrate pollution
			27.2	Water quality - Pesticide pollution
	Landscape	Agriculture influences strongly the state of Europe's landscapes through, for example, landscape elements such as hedgerows, cropping patterns, grazing of upland areas.	28	Landscape - state and diversity

Overmars et al. (2014) developed a spatially explicit methodology for a species-based indicator for biodiversity on agricultural land in the EU. The indicator combines potential occurrence of 132 species of plants and vertebrates with information on the influence of environmental pressures on these species. Based on this indicator, the authors show that biodiversity in agricultural areas in the south and east of the EU is in a better state than in the west and north, but they also observed high spatial variability. Binder et al. (2010) review a range of indicator-based assessment methods for sustainable agriculture and show that there are different trade-offs encountered when selecting an assessment method. For example, a clear, standardized, top-down procedure allows for potentially benchmarking and comparing results across regions and sites but compromises system specificity. They also showed that bottom-up, regional participatory approaches contribute best to filling the current needs of agricultural sustainability assessment, and address the applicability of the results, by involving the stakeholders in the assessment procedure and providing them with a space for the decision making system (Binder et al., 2010).

The process of selecting the sustainability indicators in agriculture should be adjusted to specific conditions. For instance, in the end of 2012, the United Nations Economic Commission for Europe proposed three additional indicators – irrigation, cropping and livestock patterns, gross nitrogen balance, which could be added to the Guidelines for the Application of Environmental Indicators in Eastern Europe, Caucasus and Central Asia (36 indicators) (UN, 2007; UNECE, 2009; UNECE, 2012). Many of the EU AEI have been developed according to the specificity of the European Union agricultural policy and at present could hardly be produced by these countries. The analysis of the agri-environmental indicators used by OECD and EEA has shown that some of these indicators have already been included in the Guidelines for the Application of Environmental Indicators in Eastern Europe, Caucasus and Central prepared by the United Nations Economic Commission for Europe (UNECE). Some indicators could also be produced using basic statistical data collected for indicators already included in the Guidelines. In countries of Eastern Europe, Caucasus, Central Asia and South-Eastern Asia it was recommended to use the following thirteen indicators based on the Guidelines (UN, 2007; UNECE, 2009):

1. Fertilizer consumption.
2. Pesticide consumption.
3. Irrigation: new indicator.
4. Energy use in agriculture: sub-indicator of indicator ‘final energy consumption’.
5. Agricultural land-use change: can be developed on the basis of data collected for indicator ‘land uptake’.
6. Cropping and livestock patterns: new indicator.
7. Gross nitrogen balance: new indicator.
8. Atmospheric emissions of ammonia from agriculture: can be developed on the basis of data collected for indicator ‘emission of pollutants into the atmospheric air’.
9. Emissions of methane and nitrous oxide from agriculture: sub-indicator of indicator ‘greenhouse gas emissions’.
10. Water abstraction: can be developed on the basis of data collected for indicator ‘freshwater abstraction’.
11. Soil erosion: indicator ‘area affected by soil erosion’.
12. Nitrates in water: can be developed on the basis of data collected for indicators ‘nutrients in freshwater’ and ‘nutrients in coastal seawaters’.
13. Share of agriculture in greenhouse gas emissions: can be developed on the basis of data collected for the indicator ‘greenhouse gas emissions’.

There is a strong demand for selecting sustainability indicators for many reasons such as: agri-environmental reports, international comparability of environmental concerns, national and global development plans and development strategies, national feedback on international regulations, conventions and environmental initiatives and evaluation of progress in the achievement of environmental goals. The process of selecting sustainability indicators for agriculture is a difficult process. Indicators should have many attributes (Table 3).

Farm management practices are defined as the decisions and operations that shape the practical management of farms, such as cropping methods, soil cover and tillage. Soil cover and tillage can be considered important agri-environmental indicators. But only limited data is available at farm level about cultivation methods except for few countries or regions. According to Piorr (2010) in places where the

Table 3: Agri-environmental indicators attributes (adapted from Piorr, 2010).

Scope of indicators	- inform about status and development of complex systems - provide sufficient information about sustainability of land-use systems - be responsive to changes related to human activities - show trends over time
Policy relevance	- provide a representative picture of environmental, agricultural and rural conditions - simple and easy to interpret for different users - provide a basis for national and international comparisons - assist decision-makers of the private sector as well as trade and industry
Analytically sound	- theoretically well founded in technical and scientific terms - based on international standards and international consensus about its validity - linked to economic models, forecasting and information systems
Measurability and data required	- have to be controllable - readily available or made available at a reasonable cost/benefit ratio - adequately documented and of known quality - updated at regular intervals - have a threshold or reference value against which to compare it.

links between the environmental effects of farming and management practices are tracked, the monitoring of farm management can be considered an early indication of likely changes in environmental impacts from farming before they can be measured by other indicators, such as soil and water quality. Information on farm management practices is also pertinent to other indicators, such as nutrient balances, soil erosion, soil fertility, water contamination, among others.

Other authors, however, demonstrate indicators that decouple the level of agricultural inputs from pressure on the environment and show that higher inputs do not necessarily result in higher pressures on the environment (Gaudino et al., 2014; see also chapter 9). Therefore, the pressure on the environment should be measured on the basis of impacts rather than by mere analysis of the level of intensification of agriculture and regulations should preferably be based on pressure indicator thresholds instead of on system inputs (Gaudino et al., 2014).

Another useful indicator can be soil cover on arable land by green crops, which measures the number of days in a year that the soil is covered with vegetation. Some authors even point out that permanent soil coverage throughout the year should be the aim (OECD, 2001).

Due to many problems with data availability and quality such as: representativeness, geographic coverage, timeliness, accuracy and precision or reliability, Eurostat (the statistical service of European Commission) has launched the project DireDate to get recommendations for setting-up a sustainable data collection system, based on best practices, for developing the agri-environmental indicators of the EU (Piorr, 2010).

During the ‘OECD workshop - agri-environmental indicators lessons learned and future directions’ a few recommendations concerning AEI has been made. One of them was to respond to policy makers’ demands with fewer but easier to understand indicators (OECD, 2010). Sill public and policy makers’ awareness of AEIs in many countries is at a low level and this is limiting their use. Some AEI, in particular related with biodiversity, farm management and cultural landscape, do not have consistent definitions. They can vary depending on the region and scaling up indicators from the farm to the country is a complex problem. Moreover social indicators have been found to provide a weak link in assessing the sustainability of agriculture (OECD, 2010). It has been articulate that there is a strong need to move beyond national level in reporting AEIs to present a spatial distribution on environmental effects, especially identifying areas at most environmental and/or human health risk (OECD, 2010). On the other hand indicators should be location specific, constructed within the context of the contemporary socioeconomic situation (Dumanski and Pieri, 1996). Indicators used in one country are not necessarily applicable to other countries due to variation in biophysical and socioeconomic conditions (Rasul and Thapa, 2003). Undoubtedly to provide a useful policy tool, a set of indicators must be taken into account to understand the relation between farm input use, farm management practices and impacts on ecosystems related to agriculture (OECD, 2010).

10.5 Conclusions

Based on the literature discussed here we can conclude that agriculture is a complex system and pursuing sustainable agriculture requires addressing various features of such a system: resilience, dynamics and adaptation; the features discussed in detail in chapter 2 of this book. The main messages of this chapter are that:

1. Pursue of sustainable agriculture is up in agendas worldwide and is also a priority in EU;
2. Many indicators are proposed and many different approaches to what sustainable agriculture is are in place;
3. The way towards sustainable agriculture is complex and should be adjusted to local circumstances. Fortunately, there are cases of success in literature on well-functioning agriculture we can learn from.

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11 Sustainability And Air Quality

Agnieszka Bartocha

11.1 Introduction

The problem of poor air quality is complex because it involves atmospheric physical and chemical processes, multiple emission sources and is linked to other large global issues like energy and transportation. The understanding of the causes of poor air quality, their consequences and measures that should be taken to improve and maintain good air quality is one of the key challenges for sustainable development. The problem with air quality is that it is often not obviously visible (except the cases with very high concentrations) like other environmental concerns (e.g. solid waste or sewage), and this poses further challenges to combating air pollution.

Harmful impact both on people and ecosystems is one of the most important consequences of air pollution. Poor air quality causes human health problems associated with respiratory conditions (such as asthma) and exacerbated cardiovascular diseases and is also responsible for acidification and eutrophication effects in ecosystems. It was estimated that air pollution was responsible for over 400 000 premature deaths in 2010 making it the number one environmental cause of premature deaths in the European Union (EU) and ten times more than the toll of road traffic accidents (Amann et al., 2012). Sixty two percent of EU's ecosystem area exceeds critical loads for eutrophication (Amann et al., 2012). According to the European Commission, the external costs of air pollution health impact range between €330-940 billion a year and direct economic losses like damage to crops and buildings are estimated at about €23 billion a year[10]. The problem of air quality is not only a concern in Europe. A report by the Organisation for Economic Co-operation and Development (OECD) (Sigman et al., 2012) states that "*without new policies, by 2050, air pollution is set to become the world's top environmental cause of premature mortality*" with projection of 3.6 million premature deaths from exposure to particulate matter a year globally in 2050. A recent report by Gurreiro et al. (2012) suggests that air pollution is responsible also for:

– Material losses including agricultural crops, buildings, cultural heritage due to soiling and exposure to acidifying pollutants and ground level ozone (O_3);
– Impacts of specific pollutants like heavy metals and persistent organic pollutants on ecosystems, due to their environmental toxicity and bioaccumulation;
– Contribution to climate change;
– Impact on atmospheric visibility.

10 http://ec.europa.eu/environment/air/index_en.htm, EC

In Europe air quality is one of the main threats to environmental and human health. Even though emissions of nitrogen dioxide (NO_2), sulfur dioxide (SO_2) and particulate matter (PM) have been decreasing regularly since the last few decades, the air quality standards especially for NO_2 and PM are not met in many European countries. High pollution levels of these chemicals are observed especially in urban areas. In 2011, more than 30% of the European urban population was exposed to concentrations of PM10 (10 micrometers in diameter or smaller) in excess to daily limit and about 5 % of the EU urban population was exposed to concentrations of NO_2 in excess to annual limits (Guerreiro et al., 2013). The figure below presents annual mean concentrations of NO_2 and PM10 measured at European monitoring stations to demonstrate spatial distribution of the pollution in 2011.

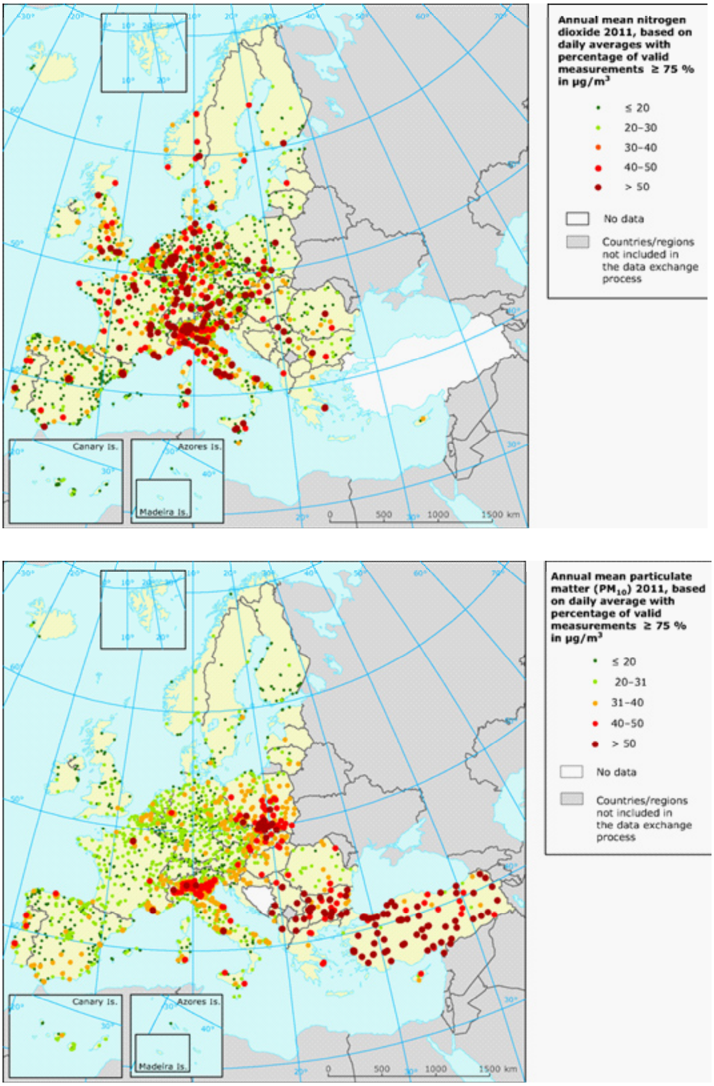
This chapter presents general reflections on the air quality in the EU and their linkages with sustainability issues. It focuses on a case study relating to air quality in Poland and presents the measures implemented by authorities to tackle the specific problem of emissions sources from domestic heating. These efforts were considered a step towards sustainability and this chapter will look more closely into the sustainability indicators used to evaluate air quality. Poland is an interesting case study due to its problems with small individual combustion sources especially in connection with coal resources and coal-based economy. Emissions from domestic heating sources are also of concern in other transition countries or developing countries therefore issues discussed here have wider implications.

11.2 General Reflections On Air Quality And Sustainability In The EU

11.2.1 When Sustainability Meets Air Quality

Although sustainability is not often mentioned within the discussions on air quality management, many air quality aspects are strictly linked to sustainability, such as:

a) Harmful impacts on people and ecosystems;
b) Material losses due to pollution;
c) Connection to the climate change policy;
d) Long range pollution transport;
e) Control strategies including:
   - Emission reduction from different sources (industrial, transportation, energy sector, agriculture and other);
   - Links to energy policy;
   - Links to a transport system;
   - Measure selection and optimization and cost-benefit analysis (CBA);



(a)

(b)

Figure 1: Annual mean concentration in 2011: a - NO_2, b - PM10 (red and dark red dots indicate stations reporting exceedances). Source: European Topic Centre on Air Pollution and Climate Change Mitigation (EIONET), European Environment Agency (EEA)[11].

11 http://acm.eionet.europa.eu/databases/airbase/eoi_maps/eoi2012/index_html - 7.05.2014, European Topic Centre on Air Pollution and Climate Change Mitigation (EIONET), EEA.

Short-term and long-term harmful impacts on people and ecosystems are the most important consequences of poor air quality. Health effects of pollution can be expressed as:

- Years Of Life Lost (YOLLs) due to exposure to pollutant from anthropogenic sources (years);
- Loss of Life Expectancy (LLE) due to exposure (months)[12];
- Premature deaths attributable to exposure (cases per year);
- Premature mortality attributable to exposure (cases per year)[13].

Estimation of the costs resulting from the avoidance of health care and the lack of absence at work are also useful metrics. The way of expressing the health impact is important to achieve the best understanding of the problem and citizens' acceptance for the measures implemented. The impact on ecosystems is estimated as an ecosystem's area with excess of critical loads for eutrophication or acidification. Material losses due to pollution are expressed as costs and they include for instance: agricultural crops losses due to O_3 exposure or building damages due to high level of acidifying pollutants and O_3. Human population exposure to air pollution, calculated mainly using dispersion modelling tools is also one of the useful air quality indicators. All above mentioned metrics are linked to sustainability aspects and, in some circumstances, can be considered as sustainability indicators.

Another issue worth mentioning in regard to sustainability is better integration of air quality and climate change policies which has been investigated and considered recently. The problem is complex due to the synergies and antagonisms existing between air pollution and greenhouse gases (GHG). The impact of climate change on air quality is related to the changes of atmospheric chemistry and meteorological conditions (Fowler et al., 2013). For instance, the duration and frequency of stable meteorological conditions have a strong impact on the dispersion of the pollution. The other example is the influence of temperature increase on ozone formation in troposphere and on enhancement of the biogenic emission of ozone precursors (VOC). On the other hand, emissions of black carbon (light absorbing particles, part of the particulate matter), methane (GHG), sulphur dioxide and ozone have influence on both air quality and climate change (Fowler et al., 2013). Further research is necessary on the air quality and climate change synergies and antagonisms due to the complexity of the chemistry

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12 LLE can be used for spatial analysis, when YOLLs is useful for estimation of at aggregated level (for instance national level)

13 Premature deaths can be define as a number of deaths in the given year among persons age between 0 and the year of average life expectancy and dividing by the estimated population for all age groups in the same year. Mortality is the risk of dying in a given year, measured by the death rate — the number of deaths occurring per 100,000 people in a population (http://www.societyhealth.vcu.edu/Page.aspx?nav=64&scope=0).

of atmosphere processes. There is large potential for policy integration. Many actions like direct emission reduction of pollutants like PM, VOC, methane, or energy efficiency measures and renewable energy implementation have a positive impact simultaneously on both the air quality and climate change mitigation. The adverse effect is also possible in some cases like biomass burning regarded as a renewable energy source and simultaneously causing the emission of PM and other pollutants.

Long-range pollution transport in the hemisphere of ozone, particulate matter, its precursors and other pollutants requires cooperation between countries especially those which are responsible for high level emissions. Local emissions can influence the air quality in different continents. The case of ozone in Europe shows the importance of the problem. Reduction of ozone precursors emission (about 30% for NO_2 and VOC) in the last decades in Europe resulted in reduction of ozone episodes but the trend of annual ozone concentration (background) is increasing because of intercontinental flows (Jonson et al., 2006). The United Nations Economic Commission for Europe (UNECE) Convention on Long-range Trans-boundary Air Pollution is a good example of international cooperation.

Emission reduction is the main measure of achieving good air quality and to decrease impacts on people and ecosystems. The new EU air quality policy (mentioned below) focuses on emission reductions to decrease the background concentration. Energy policy has large impact on the level of emissions and the structure of combustion sources and finally on the air quality including impacts on both background concentration and local high pollution episodes. Appropriate energy policy is crucial for air quality management. Transportation is the second most important emission source responsible for exceedances of NO_2,PM10 and PM2.5 limit values in many European cities. Other sustainability measures for transportation include: intelligent transportation systems, Park&Ride, bicycle paths, car sharing systems, fees systems, low emission zones and development of public transportation.

Selecting and optimising air quality measures and indicators is a very important process enabling achieving improvements in three pillars of sustainability: social, environmental and economic. In other words, selecting right indicators may allow directing actions towards sustainable management of air quality. Selection of the measures depends on source apportionment analysis which indicates the sources responsible for poor air quality. Many issues have to be analyzed to choose the best package of actions including:

- emission reduction optimization - measures with the best emission reduction effect;
- costs of measures and cost-effectiveness analysis (often used in Polish Air Quality Action Plans - AQAPs);
- barrier analysis including socio-economic implications of adopting selected actions and indicators;
- implementation time;

– availability of the measures;
– law requirements and limitations;
– cost-benefit analysis.

The cost-benefit analysis (CBA) is a good tool for a measure selection and optimization. CBA methodology is used in the EU for revision of the air quality policy and to design the best policy option taking into account costs of policy implementation and benefits due to better air quality after implementation (for instance: Amman et al., 2013).

Air quality management is complex. The impacts of air quality range from local to global (hemispheric) and short-term and long-term influence on human health and ecosystems. Similarly to other chapters of this book, this complexity is directly linked to and reflected in sustainability: climate change, energy, transportation policy and socio-economic aspects.

11.2.2 Tools For Improving Air Quality In Europe

Air quality management in Europe has three main components: air quality measures, emission measures and supporting instruments (Fig. 2.).

Air quality and emission measures are the main direct instruments for improving air quality. Air quality measures focus on impact of air pollution on human health and ecosystems and on establishing air quality standards. The measures concern areas where air quality limits are exceeded (mainly urban areas in the case of PM10, PM2.5 and NO_2) and regard measures at local, regional and national level (the type of the measures depends on the nature and source of pollution exceedances). The Directive 2008/50/EC on ambient air quality and cleaner air for Europe (CAFE Directive) – the main act in air quality management and Directive 2004/107/EC relating to arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air define air quality limits or targets. The air quality standards are established for: PM10, PM2.5, NO_2, NOx, SO_2, benzene, ozone, heavy metals, benzo(a)pyrene and other pollutants. The CAFE Directive describes the basic principles of how air quality should be assessed and managed using the following key instruments:
– assessment and monitoring of air quality;
– air quality plans – prepared for zones (areas) where exceedances were observed; including identification of sources responsible for poor air quality and plan of actions for improving air quality (called also Air Quality Action Plans - AQAPs);
– short term action plans – in the case of exceedances of alarm thresholds;
– reporting, dissemination and public information.

The emission measures focus on controlling the emissions from different sectors (i.e. combustion, transport, chemistry - solvents and paints, waste incineration and others). They include:

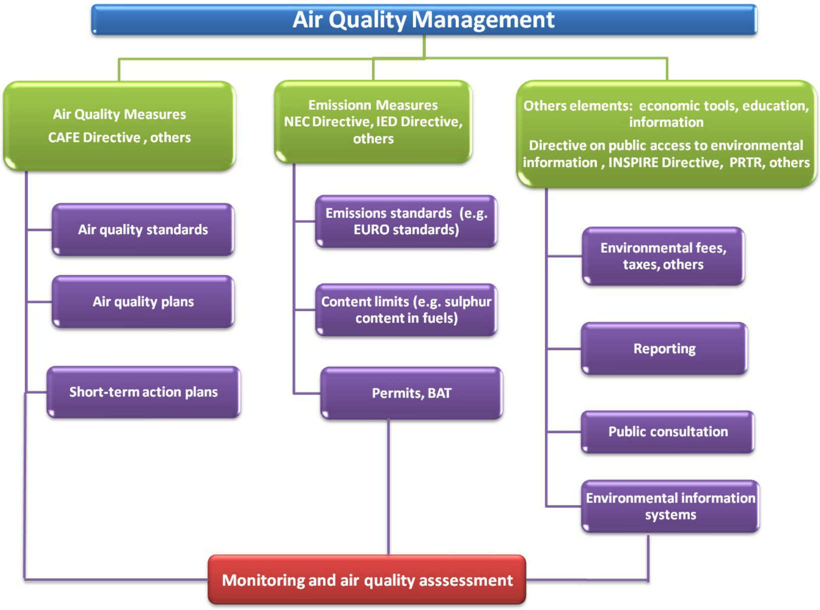

Figure 2: The main elements of air quality management in Europe (Bartocha et al., 2012).

- The national emission limits for Member States (imposed by National Emission Ceiling (NEC) Directive 2001/81/EC; new proposal is currently being prepared and consulted);
- Industrial emission control system: integrated pollution prevention and control, Best Available techniques (Reference Documents [BREFs], emission standards for Large Combustion Plants (LCP) and others; the new Directive 2010/75/EU on industrial emissions (IED) brings together Integrated Pollution Prevention and Control (IPPC), LCP and other directives concerning industrial emission control);
- The EURO standards for vehicles;
- Limits concerning content of substances in the products (like content of the volatile organic compounds [VOC] in the paints - Paint Directive or content of the sulphur in the fuel - Sulphur Content of Liquid Fuels Directive).

Other instruments that support the air quality strategy are:
- Public consultation to ensure and to achieve citizens' participation in environmental activities;
- Financial instruments like the "*polluter pays*" principle, environmental fees, taxes;
- Reporting and monitoring;
- Environmental information systems and appropriate tools like INSPIRE;
- The European Pollutant Release and Transfer Register (E-PRTR).

In 2013 the new European air pollution policy proposal was adopted (A Clean Air Policy Package for Europe – CAPE). The main aim of the package is to tackle the problems of compliance with present air quality requirements and it regards mainly measures at national and local levels. It focuses on problems with emissions from transportation (problem of complying to the NOx EURO standard under real driving conditions), on adopting tools for national and local actions, especially concerning transportation and public information, and on ensuring financial support. A long-term objective of the CAPE is to deliver further reduction of air pollution concentration towards the level of the World Health Organization (WHO) guidelines and reducing the burden of the pollution on ecosystems. It focuses on decreasing the emissions at the sources to reduce background concentration (in particular: national limits, emissions from domestic combustion sources, industrial emissions, emissions from non-road machinery sources, emissions from Medium Combustion Plants, ammonia emissions from agriculture, and emissions from shipping).

11.3 The Air Quality In Poland

11.3.1 What Is The Problem With Air Quality In Poland?

Let the story about air quality in Poland begin in a small touristic town located in the mountains in the south of Poland. Winter in the morning, people are getting up, preparing themselves for daily activities. Weather is nice but it is rather cold outside. No wind, the air is still and the sun is slowly rising from behind the mountains warming the air above the valley. In the bottom of the valley air is still cold, waiting for the sun. It is so cold that it is time to switch on the heating systems, especially for tourists sleeping in the guest houses around and it is also time to prepare food in regional restaurants and bars. Part of the town has got gas and geothermal heat distribution systems and the hotels, larger guest houses and municipal houses or buildings use them for heating purposes. But many individual houses and guest houses have their own boilers fed by coal or wood as this is much cheaper. After one or two hours the air quality begins to be more and more visible and cloudy air is not the fog… This situation is presented in Fig 3. The pictures illustrate the formation of smog due to emissions from the domestic combustion sources and weather conditions: low wind and temperature inversion (an increase in temperature with height and inversion layer formation which traps the air pollution near the surface) during one winter morning in 2013.

Dust pollution (PM10, PM2.5 and benzo(a)pyrene) is the main and most serious air quality problem in Poland. According to the official annual air quality assessment report (Mitosek et al., 2013) almost all zones[14] (in 2011 - 38 from 46) are classified

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14 Zone – part of the territory of a Member State, as delimited by that Member State for the purposes of air quality assessment and management – CAFE Directive (in Poland regional districts or cities).

Figure 3: Smog in Zakopane Basin (south of Poland, Malopolska Region). Pictures were taken in the winter morning within one hour in 2013. The first two pictures (1a and 1b) illustrate the emission from guest houses (smoke from chimneys). The next two (2a and 2b) present smog formation due to emissions and weather conditions. The last ones (3a and 3b) present a general view (smaller zooming) of the smog (inversion layer) covering the Zakopane Basin.

as a C[15] class regarding measured exceedances of 24-hour PM10 concentration. The situation is even worse when looking at the target value for benzo(a)pyrene (which is a part of the dust particle) – 42 of the zones are classified as a C. The ozone target

15 C class means that assessment (e.g. measurements) showed exceedances of limit value or target value in the area of the zone during the year and the air quality action plan is required for the zone.

value was exceeded in 5 zones. High pollution of NO_2 is observed in a few large cities, there are also some local exceedances of limits for other pollutants (arsenic, benzene) connected to industry emissions. In 2012 concentrations above annual SO_2 limit value were measured in 3 zones in the south of Poland. Fig. 4 below presents the zone classification for PM10 (A), PM2.5 (C) and ozone (D). The (B) picture presents measurements of PM10 (90.4 percentile for 24-hour concentration in 2011) to compare the spatial distribution of concentration.

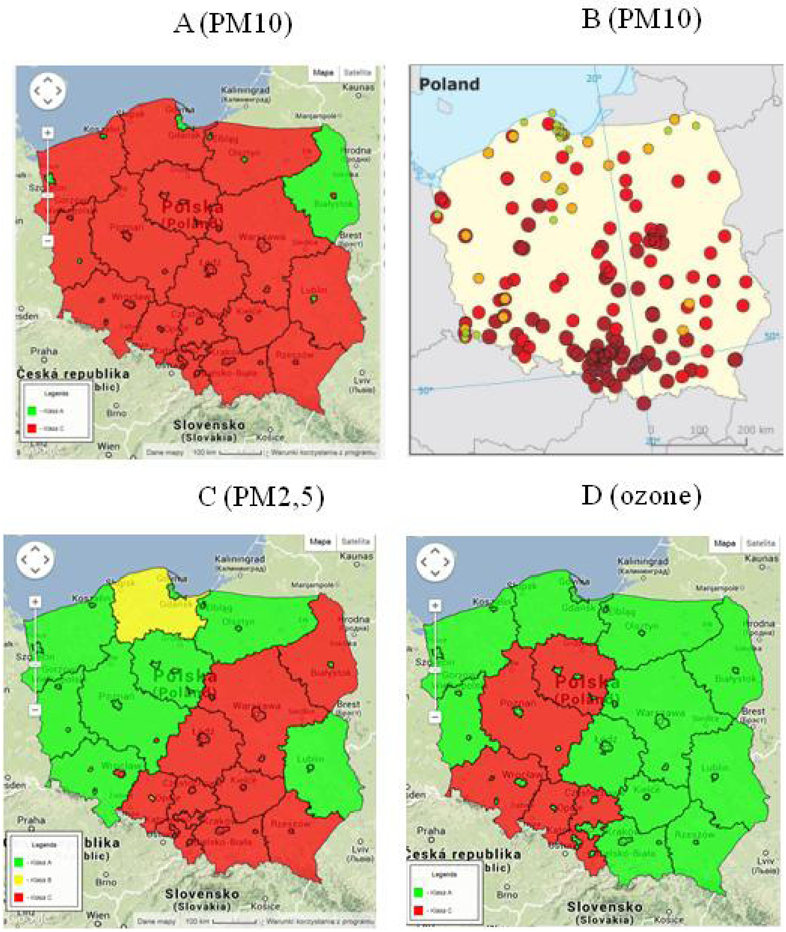

Figure 4: Zones classification and measurements results for three pollutants (bright and dark red dots and zones indicate exceedances of limit value) (Państwowy Monitoring Środowiska (PMŚ) - Inspekcja Ochrony Środowiska, 2013[16], EEA, AirBase v.7, 2011).

16 State Environmental Monitoring – Environmental Protection Inspection

The situation of high PM pollution in Poland has not changed much since the year 2000 (Fig. 5). Differences in concentration between years are due to the weather conditions rather than changes in emissions. In particular unfavorable meteorological factors (low wind and thermal inversions) when pollutants dispersion is slow were observed in the years 2003 and 2006.

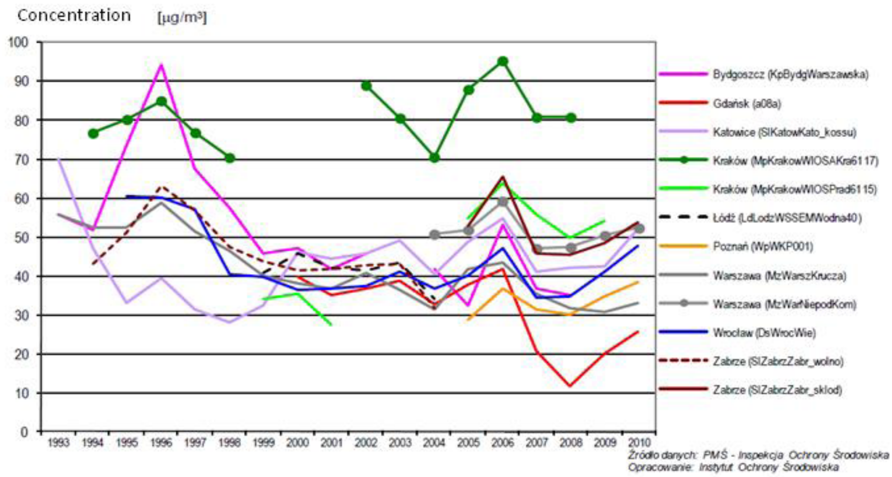

Figure 5: Series of annual concentration of PM10 measured at monitoring stations in different cities (PMŚ, Inspekcja Ochrony Środowiska, 2011 – State Environmental Monitoring, Environmental Protection Inspection – Iwanek et al., 2011).

The percentage of urban population exposed to air pollutant concentrations above the PM10 air quality objectives (24-hour, 50 µg/m^3) is estimated to be about 79-86% (2009-2011) (Air pollution fact sheet 2013 Poland, European Environment Agency, 2013)[17]. It means that every year about 80% of people living in the Polish towns/cities are exposed to the significant adverse impact of PM pollution.

The worst situation regarding PM pollution is observed in the south of Poland (Silesia and Malopolska Regions). According to The New York Times analysis, based on EEA data, the cities located in these regions (Krakow, Nowy Sacz, Katowice, Gliwice, Zabrze and Sosnowiec) are on the 10 top list of the most polluted cities in Europe (taking into account average number of days in 2011 when particulate concentrations exceeded the EU limit)[18]. In all the above mentioned cities the average number of days

17 http://www.eea.europa.eu/themes/air/air-pollution-country-fact-sheets, 7.05.2014
18 (http://www.nytimes.com/interactive/2013/10/15/business/international/europe-air-quality.html?ref=international&_r=0, 7.05.2014)

with exceedances of PM10 limit value is more than 120 days which means that citizens are inhaling much polluted air during more than one third of the year. The mixture of many anthropogenic factors like high population density, coal mines, location of heavy industry and power plants and transportation intensity result in high emissions from all sources. These sources include: industrial and energy sectors, domestic heating and small combustion and transportation in the regions. Additionally Silesia Region and Malopolska Region are characterized partly by difficult orographic and meteorological factors such as: hilly areas with valleys where low wind speed and thermal inversions are observed, as presented in Fig. 3.

11.3.2 Why Are There Problems With Air Quality In Poland?

The most important question is to find out the main sources of high level of pollution. One of the elements of the Air Quality Action Plan is the source apportionment analysis, whose main objective is to indicate the emission sources responsible for exceedances. The mean percentage of source apportionment based on analysis of all Polish AQAPs is presented in Fig. 6. According to the AQAPs analysis more than 60%[19] of PM10 concentration is due to domestic heating and other small combustion sources.

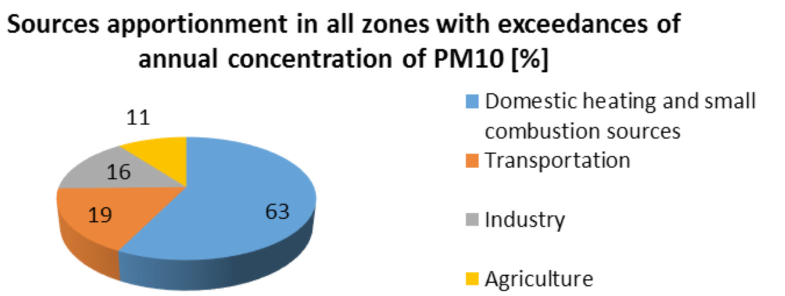

Figure 6: Sources apportionment in all zones with exceedances of annual concentration of PM10 (Bartocha et al., 2013)[20].

The Zakopane Basin (Kotlina Zakopianska in Polish) described at the beginning of the section is a touristic mountain town inhabited by about 30 thousand citizens, without heavy industry. In 2011 there were 101 days measured with exceedances of PM10 daily

19 Excluding the regional background
20 Assessment of the Thematic Strategy on Air Pollution and CAFE Directive objectives realization with special regard to air quality standards for particulate matter PM2.5

limit value. Exceedances of target value for benzo(a)pyrene (annual concentration of 8,7 ng/m^3 comparing to the target value - 1 ng/m^3) and exceedances of annual concentration for PM2.5 (36 ug/m^3 compared to 28 ug/m^3 of limit value) were also observed (Pajak et al., 2012). In 2012 the results were similar. The Air Quality Action Plan prepared for Malopolska zones in 2012 indicates that more than 90% of PM10 and PM2.5 concentrations was due to emissions from domestic heating (Lochno et al., 2012).

Transportation is often the second emission source responsible for poor air quality. In big cities its share is significant - according to the AQAP for Warsaw established by the regional authorities[21], transportation is indicated as a main emission source responsible for high level of PM10 concentration.

Poland complies with the national emission limits imposed by the NEC Directive, nevertheless air quality trends have not significantly changed. A lot has been done, but the problem is very complex and integrated with energy policy, especially with fuel prices for individuals and with a fast growing transport sector.

11.3.3 Sustainability And The Problem Of Domestic Heating In Poland

Understanding the current energy policy, social situation, fuel resources available in Poland and its spatial distribution is necessary to tackle the air pollution problem in the country. Historical facts have also some impact on the whole issue. Energy and heat production is based on solid fuels (hard and brown coal). According to Energy Regulatory Office central heating production is based on more than 70% of solid fuel (coal) including large power plants and small units >5 MWt (Bunczyk, 2013). The central heating system delivers about 23% of required heat to citizens (up to 80%, on average 50% in the cities), 63% of heat demand is produced in small domestic stoves and boilers using solid fuels, mainly coal (author’s calculation based on statistical data from Central Statistical Office of Poland and ATMOTERM Corp. internal analyses). The location of coal resources and coal mine industry in Silesia Region caused industrial development and strong mining traditions and hence the common usage of coal for heating purposes in the south of Poland.

Explanation of some confusion with terminology is also necessary. “Low emission” term is used to define emission from domestic heating and small combustion sources in air quality management in Poland and at the same time is used for CO_2 (GHG) emission sources by the energy or economy sector concerning climate change issues. In Europe the “low emission” is often used if emissions from transport are considered (e.g. Low Emission Zones). Therefore there are many misunderstandings between different sectors concerning the climate change and air quality management which makes the

21 http://www.bip.mazovia.pl/sejmik/uchwaly-sejmiku/uchwala,2602,18613.html 7.05.2014 action implementation, policy integration or raising the air quality awareness more difficult. For example it is difficult for many people to distinguish different objectives of the Low Emission Reduction Program and Low Emission Economy Plans (in other words: Low-Carbon Plans) conducted by local authorities. In this chapter "low emission" term means emissions from domestic heating and small combustion.

Low Emission Reduction Program (LERP) is the third issue specific for Poland worth an explanation. LERPs were set up by local authorities and they enable to fund investment costs of changing domestic heating systems to limited emission ones such as gas, oil boilers, modern coal boilers, central heating or electrical systems. Some Programs include also the funding of insulation of buildings or solar systems. The Programs funding of replacing old coal heating system into new solid fuel boilers are specific for the south of Poland, where coal mining traditions are strong. The decision to fund the replacement to the new solid fuel boilers was based on socio-economic analyses. The operational costs of using solid fuels (coal, wood) in domestic heating system are the lowest (see Fig. 7.). The modern solid fuel boilers ensure much lower emissions due to better combustion efficiency than old models. If an exhaust system (a chimney) is also renovated the emission reduction is even better. To ensure that emissions decrease the Programs are more complex and include the Operator responsible for the quality of solid fuel boilers and the quality of construction works. The retort boilers including the automation of the combustion process and quality of the fuels are often used in LERPs to achieve appropriate emission reduction effect. LERPs are the main measures used to tackle the problem of "low emission" in Poland.

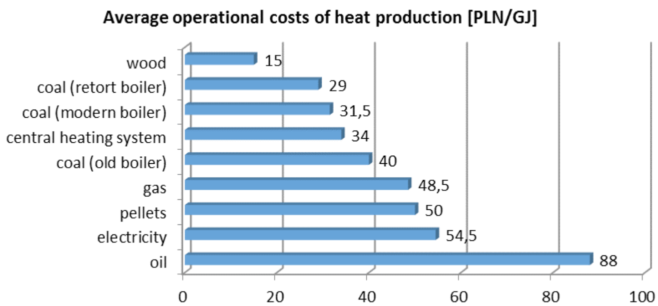

Figure 7: Example of average operational costs of heat production calculated for Silesia Region in 2011 (Lochno et al., 2011)[22]. It can be observed that average operational costs are the cheapest for solid fuels boilers and not everybody can afford heating systems other than solid fuel boilers. Therefore there is no incentive to switch to more environmentally friendly systems.

22 (Program ochrony powietrza dla stref gliwicko-mikołowskiej, i częstochowsko-lublinieckiej województwa śląskiego, w których stwierdzone zostały ponadnormatywne poziomy substancji w powietrzu, Katowice, 2011)[23].

23 Air Quality Action Plan for Gliwice-Mikolow and Czestochowa-Lubliniec Zones in Silesia Region where air quality stadards exceedances were observed.

Air quality measures concerning the pollution from domestic heating sources are implemented at all levels: local, regional and national, however the intensity of the actions is diverse.

11.3.3.1 Local Level

Pollution from domestic heating sources was governed first by local authorities from the most polluted regions namely Malopolska and Silesia. In 1992 Low Emission Reduction Program in Krakow was funded by the U.S. Agency for International Development, in 2000 the Regional Environmental Fund supported the LERP in Tychy and since then a lot of Programs have been launched especially in Malopolska and Silesia Regions (about 50% of municipalities in Silesia Region have carried out already the LERP). The structure of the Programs differs and depends on the type of financial support. Because of the operational costs modern solid fuel boilers are the most popular option in the LERPs in the south of Poland. The simplest Programs enable the receipt of funds on the basis of documents giving evidence of changing the heating system (e.g. invoice of boiler purchase and/or construction works). Programs including supporting the change to modern limited emission solid fuel boilers are often more complicated. Local or Regional Environmental Funds are usually used.

11.3.3.2 Regional Level

LERPs are included in the air quality action plan (AQAP) prepared by regional authorities as one of the proposed measures. Regional Funds are used to support LERPs. In 2013 the regional authorities established, for the first time in Poland, Solid Fuel Prohibition Act for Krakow City. Regarding socio-economic consequences, the Act is a difficult decision and raising Krakow citizens' awareness, as well as local authorities' support has made this decision possible to be established. There are still a lot of problems ahead for implementation of the Act including the setting up of the financial support system of operational costs of heat production for citizens with low income.

11.3.3.3 National Level

In 2013 the new KAWKA Program (Low emission reduction program promoting energy efficiency growth and development of small-scale renewable energy) was launched by the National Environmental Fund. The Program allocates funds (about 100 MLN €) to the Regional Environmental Funds to support low emission reduction actions included in the AQAPs. Financial support is dedicated to the regions where exceedances of PM10 and B(a)P have been measured and depends on the population exposed to air pollution. The resulting indicators of the Program include emission reduction of PM10, PM2.5 and CO_2.

The changes to modern solid fuel boilers assure the wider participation in the Program because of their low operational costs, tradition and easy access to the coal. Cases with more funds than citizens willing to change the heating system have happened. This happened especially when the financial support included only the replacement to more expensive types of heating (the limited emission modern coal fuel boilers were excluded from the financial support). This example shows that sustainable solutions towards better air quality must include also socio-economic aspects (such as costs and tradition) besides merely emission reduction. The solution is indeed not perfect and some problems are listed below (Table 1).

A large effort has been put into low emission reduction actions but the air quality has not shown improvement (Fig. 5.). The following list of problems at various levels explains why the air quality in Poland remains unimproved and shows the barriers to sustainability of air quality.

Air quality in Poland is becoming a priority yet further actions are needed to increase the efficiency of existing measures and adoption of new ones in order to achieve the improvement of air quality over the long term and including all aspects of sustainability: environmental, social and economic.

At the national level measures regarding the energy policy should be considered to decrease the costs of heat production from central systems. The cost-effectiveness calculation is a very important indicator to consider besides emission reduction effect, the socio-economic aspects in Low Emission Reduction Programs. The integration of air quality policy with climate change policy is needed for good management of synergies and antagonisms in both policies. Impact of the ETS system on the costs of heat production in power plants is a good example of strong interactions between policies. All measures concerning the energy efficiency and renewable energy sources have a positive effect on reducing the emissions from small combustion sources. The integration process has already started (e.g. KAWKA Program or AQAP for Malopolska) and it includes analysis of CO_2 emissions. In that, both air quality and climate change indicators are used to monitor the Program's results. Legal support is needed to enable efficient control and monitoring the quality of solid fuels used by individuals and to establish emission standards for domestic boilers. Control of emission levels from chimneys can be regarded as a very important indicator for achieving and sustaining the real improvement of air quality at local level.

Another very important measure is raising air quality awareness and enhancing citizens' participation in the air quality policy. Knowledge dissemination, education campaigns and citizen science actions should be carried out at national, regional and local levels. CBA analyses and citizens science can be very useful tools to raise citizens' awareness.

Improvement of LERP management is needed at local level including optimization of actions focusing not only on emission reduction but also on air quality. Project long-term objectives should include decrease of air pollution concentration and the project baseline concentration should be estimated (or measured). The PM10/

Table 1: Problems and barriers of air quality management at local, regional and national levels.

Local	Regional	National
- The level of the emission reduction due to the implemented LERPs is still insufficient in the municipalities to achieve air quality standards - Lack of monitoring of change of individual heating systems not taking part in the LERP from gas/oils or central heating systems to solid fuel boilers at the same time. Therefore there is no information about overall emission changes in the municipality. The trends for using the solid fuel boilers intensify when the prices for gas or central heating systems are increasing or during economic crisis - Insufficient emission reduction in modern coal low-emission boilers due to using poor quality solid fuel and inappropriate operation of the boiler leading to ineffective combustion process - Using emission reduction indicators – there is need for better indicators for assessing and monitoring the LERPs' results	- Cooperation and participation of local authorities and stakeholders during AQAP preparation by regional authorities	- High operational costs of other fuels and central heating. The poor quality coal is much cheaper then gas and oil, it is also cheaper than the central heating system (Fig. 7.). Poorer citizens use also waste as fuel (this is forbidden but controlling system is insufficient) - Insufficient central financial support for the regions (before KAWKA Program) - Lack of a control and monitoring system of the solid fuels' quality used by individuals, lack of emission standards for domestic boilers and other small combustion sources - Lack of integration between energy policy, climate change policy and air quality policy Confusion with "low emission" term is the best example. Some level of integration can be obtained using integrated project result indicators like in KAWKA Project: emission of PM pollutants and CO_2
Still low public awareness of air quality impact on human health and ecosystems and real costs of this impact		

PM2.5 concentration change or population exposure to these pollutants can be used to monitor the project's long-term results concerning the health effect of the air quality. The reduction of pollution concentration monitoring has advantages such as the fact that it indicates the final expected project result and can be based on a measurement instead of theoretical data (such as emission calculations). On the other hand, these measurements are still very expensive and the concentration level is influenced by weather conditions. Hence long-term measurements are needed for trend identification, of utmost importance for observing sustained effect.

Integration with climate change measures like insulation of the houses and implementation of individual renewable energy sources to decrease the heat demand will lead to better air quality effects in LERPs. The integrated projects therefore should be preferred in the process of funding attribution. Adverse effects of changing domestic heat systems to cheaper but more polluting ones can be better governed by introducing local taxation or rules in local developments plans with a good control and monitoring system. Many municipalities in Poland have already introduced a range of these improvements and the best practices should be disseminated.

11.4 Conclusions

Air quality management is strongly linked to sustainability due to its complexity, short and long-term harmful impact of pollution on humans and ecosystems and a global range of the impact.

Although much effort has been put into emission reduction, the air quality has not changed much in Poland. Implementation of the following further measures and actions is recommended:

1. Decrease of the costs of heat production from central systems by integrating energy policy and air quality policy at the national level.
2. Integration of air quality and climate change policy for a better management of synergies and antagonisms in both policies.
3. Legal support to enable efficient control and monitoring of the quality of solid fuels used by individuals and to establish emission standards for domestic boilers.
4. Raising air quality awareness and enhancing citizens' participation in the air quality policy. Knowledge dissemination, education campaigns and citizen science actions at national, regional and local level are necessary.
5. Improvement of LERP management including optimization of actions focusing not only on emission reduction but also on air quality.
6. Including in LERP decrease of air pollution concentration objective and the project baseline concentration.
7. Adverse effect of changing the domestic heat systems to cheaper but with high-emission ones can be governed by introducing local taxation or rules in local development plans and good control and monitoring systems.
8. Integrating energy efficiency and renewable energy sources measures in LERPs to achieve better results in air quality.
9. Dissemination of best practices at local, regional and national levels.

The above-mentioned recommendations should be supported by implementation of relevant indicators. Proposition of such indicators is listed in the Table 2.

First two indicators (concentration or population exposure change) focus on air quality trends. They are long-term output indicators. They suppose to monitor if emission reduction plan (i.e. LERPs) result in improvement of air quality. The concentration indicator can be measured or modeled, population exposure is

Table 2: Proposition of indicators useful for monitoring of sustainability in "low emission" reduction actions and for improving air quality in Poland

Indicator objective	Indicator	Unit	Description
Optimization of LERPs management Rising stakeholders understanding (local level)	Concentration of PM10, PM2.5, benzo(a)pyrene changes	[μg/m^3, ng/m^3] or [%]	Concentration trends in LERPs long-term result
	Population exposure to exceedances changes	[no. of people exposed to exceedances]	Changes of population living in the area of exceedances long-term result in LERPs
Monitoring of adverse effects (local level)	"Low emission" balance	[Mg/year]	Monitoring of "low emission" to estimate actual overall "low emission" changes at LERPs area
Integration with climate change measures (local, regional national level)	CO_2 reduction	[Mg/year]	Integration of climate change indicators in the air quality/pollution projects
	Reduction of energy demand in buildings [energy efficiency increase] and/or share of renewable energy in the building	[GJ/year] and/or [%]	Integration of climate change measures in the air quality/pollution projects
Raising air quality awareness and enhancing citizens' participation, Measures' optimization (local, regional national level)	Benefits resulting from improving air quality	[U$/year]	Calculation of costs savings thanks to i.e. smaller number of job dismissals and less frequent stays in hospital for the air quality/pollution projects
	e.g. Cost/Benefits Indicator	[U$/U$]	Comparison of costs and benefits in the air quality/pollution projects
	e.g. Loss of Life Expectancy (LLE)	[Months/year]	Calculation of external costs in LEPRs the air quality/pollution projects
Monitoring of measures implementation for "low emission" reduction and air quality improvement (national level, local level)	e.g. Operational costs of heat production	[U$/GJ]	Comparison of costs of heat production
	Emission standard for domestic boilers	[Yes/No]	Monitoring of frame support

calculated using modeling. The measures indicate the main objective of emission reduction projects and help in better understanding of the project.

Climate change indicators (CO_2 reduction, energy demand in buildings, renewable energy share) estimate the broader positive impact of the air quality/pollution projects (LERPs). The actions concerning also climate change issues decrease the operational costs of heating systems and help to achieve more sustainable effects in air quality projects (stop adverse trends in changes of heating systems caused by high operational costs).

Indicators related to the costs and benefits analyses are very important for monitoring of the cost-effectiveness of measures (what type of heating system replacement would be the most effective). They are also useful in consultation process to illustrate citizens the benefits of air quality improvements and for monitoring the level of benefits achieved after project implementation.

"Low emission" balance measure estimates the main projects' result - overall emission changes due to the LERP and replacements of individual heating systems not taking part in the LERP at the same time in the municipality. The measure has an impact on other indicators - good monitoring of emission changes is the basis for calculation of many of the proposed indicators.

The methodology of the indicators can be different from the proposed according to the data availability or other specific local conditions (especially for cost/benefits indicator, health impact indicator or operational costs calculation for which there are many various methods). It is important to remember the main objectives of the indicator when the methodology is being chosen.

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12 How To Measure Wastewater Systems' Sustainability?

Dariusz Latawiec

12.1 Introduction

Environmental pollution can be grouped into three main classes: water pollution, air pollution and soil pollution. Water pollution can originate from multiple sources, but common among them is wastewater discharged from domestic, agricultural or industrial facilities and processes. The management of domestic wastewater pollution is one of the targets of the European Council Urban Wastewater Directive (91/271/EEC). Pollution reduction can be achieved in different ways and the most commonly used strategy is the use of sewerage systems built in areas that meet defined criteria of technical and economic rationality. Processes of wastewater systems' construction are quite advanced in the European Union (EU) but currently there is a big gap between Western and Central-Eastern EU regions due to financial reasons and historical political divisions. It is estimated that over 80% of the population of France is provided with sewers (wastewater system is in operation together with the proper treatment system) (Mascarau, 2008; Eurostat, 2014) compared to 97% in the United Kingdom, 94% in Austria and 96% in Germany both in urban and rural areas (Eurostat, 2014). The number of people connected to wastewater systems (also equipped with the proper treatment systems) has been very rapidly increasing in Poland in the past 10 years due to external financing by European funds. Yet only 65% of the population is served by sewers – out of which 89% is served in urban areas and only 30% in rural areas (Polish Chamber of Waterworks, 2013). It must be noted that in 2002 the total population served by sewers was 57% with 83% and 14% in urban and rural areas respectively. Where there are no sewerage systems usually septic tanks or small wastewater treatment facilities are used by the population. Both systems (small wastewater treatment plants and septic tanks) are difficult to regulate and can cause environmental pollution if not serviced properly. Therefore increasing the number of economically justified, well-designed and functioning wastewater systems is important for addressing environmental and sustainability challenges. Article 3 of the Urban Wastewater Directive points out that all agglomerations (as defined by the Directive, see below) should be provided with collecting systems for urban wastewater. If the establishment of a collecting system is not justified either because it has no environmental benefit or because it is associated with excessive cost, individual systems or other appropriate systems which can achieve the same level of environmental protection can be used. Although care should be taken with poor people, increased costs must not necessarily lead to unsustainability (given that the more affluent part of the population is willing to bear these costs).

The term “agglomeration” is defined as “an area where the population and/or economic activities are sufficiently concentrated for urban wastewater to be collected and conducted to an urban wastewater treatment plant or to a final discharge point” (European Council, 1991). The very definition of agglomeration has caused a problem in the Polish context. As it can be understood from the above, the wastewater systems should be built mainly in the agglomerations. The reasons for such an assumption are simple – only sufficiently concentrated activities will produce sufficient quantities of wastewater that would be economically justifiable to collect and treat in a single facility (wastewater treatment plant). The implementation of the definition of agglomeration in some cases resulted in perceived exclusion from the society, in particular in rural areas where wastewater systems would not be provided, as they were not considered an agglomeration (agglomerations were not established).

As a consequence, pressure was put on local authorities to include all areas and dwellings into the agglomerations, irrespective of the geographical conditions and distance from the main pipelines. Due to political and legal changes the established agglomerations were never properly verified by the higher level authorities. An effect of this is that the established agglomerations produced excessive costs of wastewater collecting systems that would have to be built in order to meet the Urban Wastewater Directive goals. In addition, Polish settlements and villages are more scattered than in some other EU countries and this could have been the reason for miscalculations of the agglomerations’ areas as well. These factors have led to “a bad use of a good science” – seemingly good implementation of the Directive has led to wrong actions. Thus a question is raised: whether the term “agglomeration” should rather be translated as “basin”?

Fig. 1 shows the schemes for sufficiently concentrated settlements, where establishment of the agglomeration would be relatively easy (Fig. 1a) and the examples of scattered settlements where agglomeration would probably not meet any economical standards (Fig. 1b).

It is only now, 10 years after the term “agglomeration” was firstly introduced in the Polish legal system, that the knowledge of the methodology for defining the borders of agglomeration properly becoming widely understood. The local authorities became aware that establishing agglomeration rationally requires a technical concept that has to be calculated, drawn and paid for. The population should be carefully assessed, as well as tourist activities and the amount of industrial loads must be known. The agglomerations established in the past as the effect of political decisions by the district council (the authority responsible for this initiative) are now being replaced by more rational and technical-based solutions.

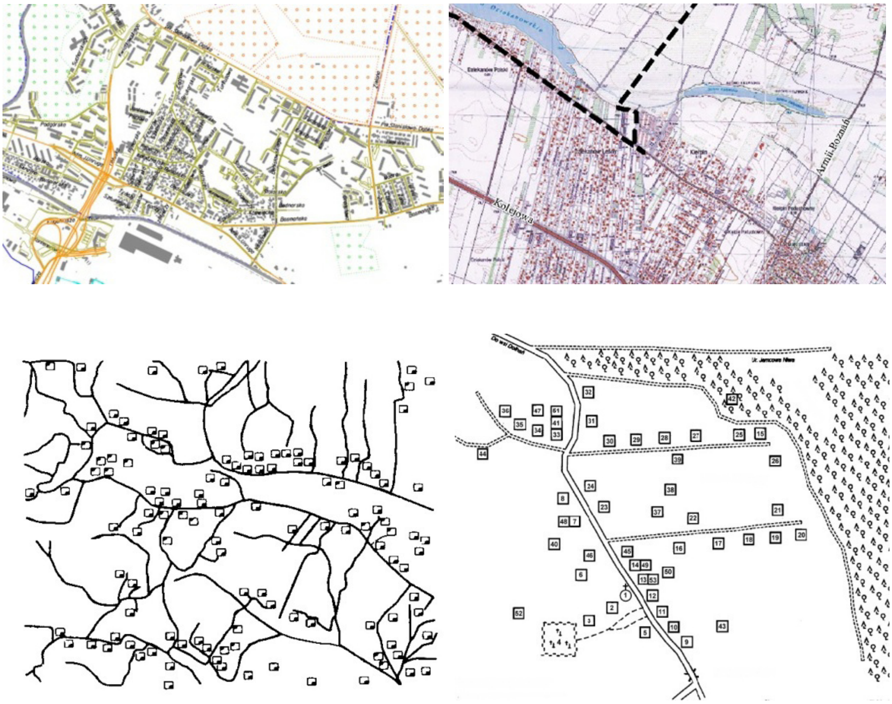
a)
b)

Figure 1: Concentrated (a) and scattered (b) settlements.

12.2 Indicators For Sustainability Assessment Of Wastewater System

The implementation of the Urban Wastewater Directive in the Polish legal system was done in a few Acts. One of the Acts was the directive of the Ministry of Environment concerning the methodology for establishing agglomerations' borders. The Act adopted the definitions of agglomerations from the Wastewater Directive also introducing the methodology for defining which areas are sufficiently inhabited (concentrated) to justify wastewater system construction. It was stated that it was rational to construct sewerage only if the population density is 120 PE (population equivalent) for every kilometer of pipelines built (with a few exceptions defined in the Polish directive) (Polish Ministry of Environment, 2010). Such a statement has not appeared in the European Union Directive and was new to the Polish legal system. Currently, there is an ongoing discussion on whether such an indicator is needed and if it was defined properly. For the sake of this article it will be called population density indicator – PDI, and the population density will be stated as PD.

Some other indicators can be used to assess economic and/or technical rationality of wastewater collection systems. The most important economic indicators would cover the cost of wastewater system construction (investment) per capita, the annual maintenance costs per capita and the costs of household connection to wastewater system. The technical indicators could be defined as the collection and/or connection rate (the volume of sewage collected by a wastewater system compared to the volume produced by a population of agglomeration, the number of households connected to wastewater system compared to a total number of households in the agglomeration).

It may seem that the subject of sustainability should be quite obvious considering the abovementioned indicators. However, discourse on sustainability of sewerage is seldom on top of the agenda during various discussions in Poland. Sustainability of wastewater systems (both wastewater collection and treatment facilities) is considered a subject when technologies of renewable resources, water reuse, wastewater separation at the source or pollutants discharge loads minimization are taken into account (ISO, 2007). On the contrary, sustainability is not an issue when discussing economic justification of wastewater systems' investments.

Considering the relatively low wastewater service availability (compared to other well-developed European countries) and a need for its increase, Poland is an interesting example to analyze sustainability and sustainability indicators in this sector of economy. Prior to joining the European Union in 2004, Poland had signed the Adhesion Treaty that imposed certain duties on Polish government. Environmental protection and control was one of the major issues and wastewater management was also included. Currently, there are vast investment works in wastewater systems all across Poland due to increasing pressure on the need to meet the Adhesion Treaty requirements in 2015. Wastewater pipelines were built covering several kilometers but the rationality behind some of them can be questioned. If an appropriate feasibility study had been performed the sustainability of such systems could have been undermined. The case study described here is therefore a good opportunity to observe, compare and analyze various effects (economic, technical, environmental, social) of a very rapid infrastructure growth and its impacts on sustainability of wastewater systems.

This chapter presents practical experiences of wastewater systems management in Poland. It aims to highlight sustainability issues by showing that some indicators, while commonly used, are sometimes significantly underestimated by decision-makers. The chapter shows that financial and technical guidelines for designing and constructing wastewater collection systems are practice-based and also illustrates the potential consequences of ignoring them.

12.3 Context Of The Case Study

All the examples in this chapter are taken from a company that manages wastewater systems in 10 southern Poland districts, Minor Poland province. All these Districts are characterized by rough alpine climate, mountain landscape, and are mostly rural areas. The Company deals with investment processes and every-day running activities of 16 small and middle-sized wastewater works and about 600 km of sewerage.

As discussed in the introduction to this chapter, the rationality of infrastructural investments should be that the pipelines are built in the areas with the highest population densities first. And only after those areas are successfully covered with wastewater collecting systems can areas of lower population densities be served. Unfortunately this logic often does not go together with the practice. The wastewater systems were built in quite remote and deserted spots and improperly established agglomerations have only enhanced such irrational activities.

Fig. 2 presents the pipelines that were actually built in one of the Districts managed by the Company. 600 meters of wastewater pipelines were built connecting six dwellings to a system. The resulting population density indicator was 40 PE/km and the costs were PLN 50,000 for each dwelling(€ 12,000, $ 16,700; 1 € =4.20 PLN, 1$=3.00 PLN). It must be pointed out that according to current Polish experience and engineering practice the rational cost of connecting sewage pipes to a single dwelling should be no more than PLN 15,000 (€ 3,570, $ 5,000). Also the cost of high quality on-site wastewater treatment construction is some PLN 15,000 in Poland. It is obvious that the realization of this investment is quite controversial.

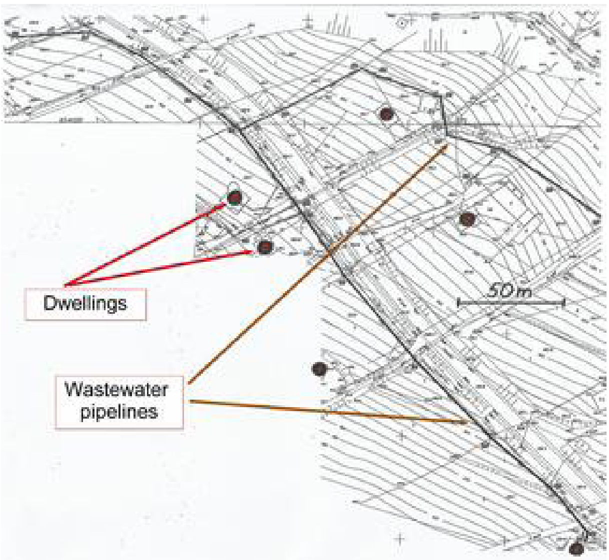

Figure 2: Wastewater pipelines in low concentration area, Minor Poland province.

On the contrary, Fig. 3 below shows a very rational concept of agglomeration borders where many dwellings had to be excluded from the agglomeration due to economic and technical reasons (Jazwiec, 2014).

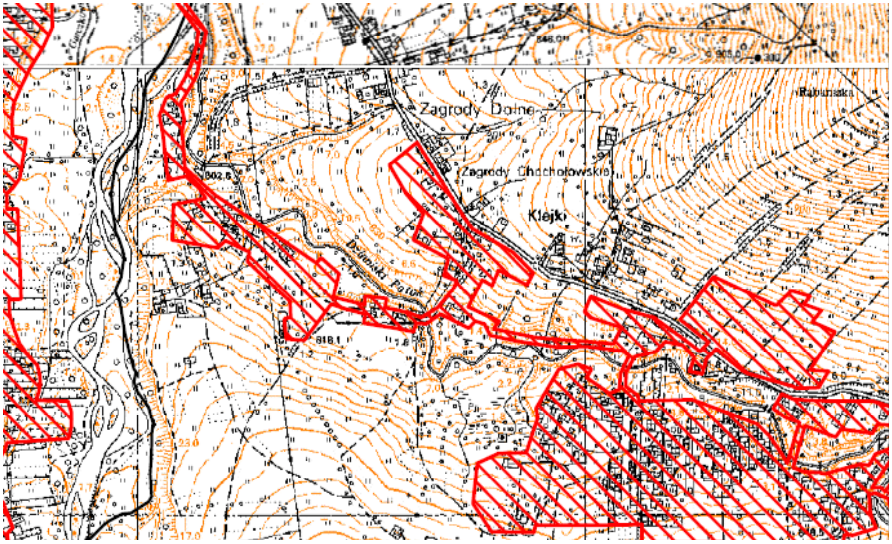

Figure 3: The proposal of agglomeration borders (Jazwiec, 2014), Minor Poland province. The red area stands for agglomeration, the borders are very irregular to cover the areas of economic rationality only. Many dwellings had to be excluded for the agglomeration.

12.3.1 The Polish Tariffs System And Wastewater System’s Indicators

The “*Polluter Pays Principle*” is an environmental policy principle which requires that the costs of pollution be borne by those who cause it. Concerning the wastewater management, the general rule of “*the polluter pays*” is being realized by covering all and every costs during the process of calculating the tariffs. The principles of water and wastewater tariffs calculation are the subject of yet another Polish directive issued in 2006 (Polish Ministry of Infrastructure, 2006. Tariffs Directive). According to this directive the costs that have to be considered during tariffs calculations include: amortization (depreciation), taxes (land, property, income taxes and others), repairs and renovations, energy, materials, external costs, payroll, financial costs (interests), and any others that can contribute to overall costs of running all the activities (Polish Ministry of Infrastructure, 2006). Thus the tariffs are meant to cover not only running costs but all the investment costs and the costs of future reconstructions. And this should be treated as meeting the requirements of wastewater systems’ sustainability. The Company calculates ten separate wastewater tariffs (one tariff for each District) and draws ten separate investment plans (investments are also the subject of tariffs calculation).

However, this is theoretical and the reality is that due to the specificity of the Polish legal regulations, not all the companies in Poland include all the costs in tariffs. In addition, not all of the organizations responsible for water supply and wastewater

Table 1: The Company's costs by type [%]. The columns "benchmarking I and II" show average values taken from two different benchmarks the Company is involved in. Benchmarking II contains more companies that have finished big investments, what can be observed by increased amortization and financial costs (loan interests) (Latawiec, 2014).

No.	Costs by type	Structure [%]	Benchmarking I [%]	Benchmarking II [%]
	Amortization and depreciation	28.1	~ 20	~ 22
	Materials and energy	7.0	~ 16	~ 18÷19
	External services	26.2	~ 17	~ 10÷12
	Taxes and charges	9.7	~ 13	~ 13
	Payroll	21.8	~ 33	~ 27÷30
	Financial and other costs	3.8	~ 1	~ 6÷7

collection are subject to the Tariffs Directive, which means that they do not have to include all the costs in tariffs calculation. This alone means that the systems in operation are not sustainable. The Company calculates the tariffs according to the Directive. The Company has completed the investment that was supported with a grant from the European Union of 17 million Euros (while the annual turnover of the Company was 3 million Euros at the time of the investment). The resulting costs of amortization and property taxes (assets-related costs) are now all calculated in the tariffs. The structure of costs by type is shown in Table 1. The table also shows the structures of some companies according to a benchmarking with which the Company is involved. It can be easily noticed that amortization and depreciation highly influence the overall costs of the Company, compared to the other organizations. This is a scale of indirect investment influence on tariffs. It can be seen how irrational investments can influence future tariffs for the whole community.

As a result of such a structure of costs as well as due to poor optimization of the investments in the past, the tariffs calculated by the Company for most of the Districts are quite high. In some of the Districts they can even exceed 3% of household's available income. It is noted that 3% of household's available income is the maximum to be paid by citizens for water and wastewater services (JASPERS, 2006). The situation is even more complicated due to the fact that there is a heavy touristic load on some of the areas covered by the Company activities and this generates some wastewater loads that are not simple to determine. The wastewater tariffs as well as some of the important factors influencing tariffs are shown in Table 2.

Table 2 shows some relations between tariffs and population density although the overall picture is blurred due to differences in amortization and investments realized in the past amongst the Districts. For example PDI is very similar in Districts 2 and 7 but the tariffs are much different and the reasons are differences in amortization and number of connections. Fig. 4 shows a visual interpretation of Table 2. As a very

Table 2: Tariffs for sewage services in 2014 and some characteristic factors in the Districts.

District	Tariff [PLN] net	Gross	*Number of connections*	*Length of sewerage[km]*	*PDI [PE/km]*
1	13.73	14.83	750	37.5	120
2	24.67	26.64	884	56.3	63
3	13.65	14.74	1705	85.2	90
4	23.51	25.39	148	12.0	62
5	13.27	14.33	1061	55.5	76
6	12.50	13.50	1681	79.8	84
7	9.39	10.14	2264	140.9	65
8	12.47	13.47	1660	142.3	82
9	12.70	13.72	737	53.6	55
10	10.38	11.21	1141	32.9	169

general rule one may note that the higher the population density the lower tariffs are. Of course lower tariffs are more acceptable by society.

Therefore some selection and processing of data was made that included finding common cost factors all across the Company and considering the technological and technical specificity of wastewater treatment plants and investments realized in the past. As a result of data processing it was calculated that the running costs of wastewater collection alone is some 5,500 PLN/km (1,310 €/km) as average cost for the whole Company and all the Districts. Considering the costs of wastewater treatment the running costs are approximately 12,000 PLN/km (2,890€/km) of sewerage. These include the costs of minor renovations, energy, materials, payroll and do not include assets-related costs (amortization and property tax). These were calculated based on data available for the Company and should not be applied for any other organization. Furthermore, these costs will vary when the technologies within the Company change so they should not be applied even for the Company in the future should any major operational conditions change.

The above data processing and calculations allowed calculating theoretical tariffs that would be generated for any new wastewater pipelines built that would be only the function of the investment costs and the number of people using newly built lines. Thus the assets-related costs may be calculated from investment's costs and any other costs would be calculated using the running costs as stated above. It was also assumed that the water consumption would be 2.5 m^3/(PE* month). This is consistent with the long-term observations made within the Company.

The results of the calculations performed for the investments that had been already realized are shown in Fig. 5. The investments that are currently being done or are planned in the forthcoming years are shown in Fig. 6. The calculations were made according to the formula below:

$$Tariff = \frac{arc + rc}{(connections) * 4 * 2.5 * 12} [PLN/m^3]$$

where:
arc – annual asset-related costs calculated as investment costs multiplied by the factor 0.04 [PLN]
rc – annual mean running costs – here calculated as 12,000 PLN/km multiplied by the length of pipelines [PLN]
4 – number of inhabitants using a single connection to a system (people living in a dwelling) [persons]
2.5 – monthly consumption of water per person [m^3/(person*month)]
12 – number of months per year [month]

Now it can be observed more clearly that the higher PE/km indicator is, the lower the tariffs are. Some exceptions can be observed and they are due to a large increase of investment costs caused by the obligatory roadwork. The preliminary thesis could be expressed:
– if PDI (population density indicator) is some 100the PE/km, the tariffs would be some 10 PLN/m^3,
– if PDI is ≈ 50÷70PE/km, the tariffs would be some 15÷20 PLN/m^3,
– and of course if PDI is above 120PE/km, the tariffs would be below 10 PLN/m^3.

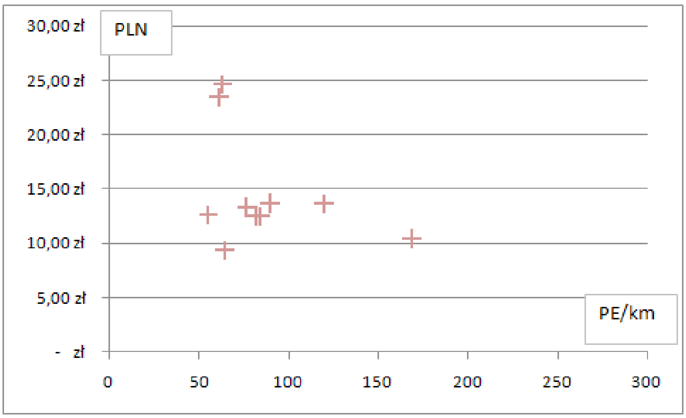

Figure 4: Relationship between wastewater tariffs and population density indicator. The X-axis shows Population Density Indicator (PDI) and the Y-axis shows tariffs level in PLN.

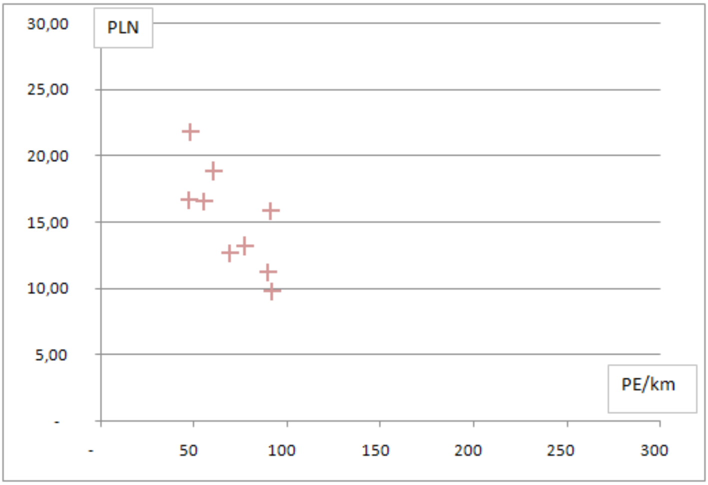

Figure 5: The relationship between wastewater tariffs (Y-axis) and Population Density Indicator (PDI on X-axis as population equivalent per km) – calculated for the investments finished by the Company in the past.

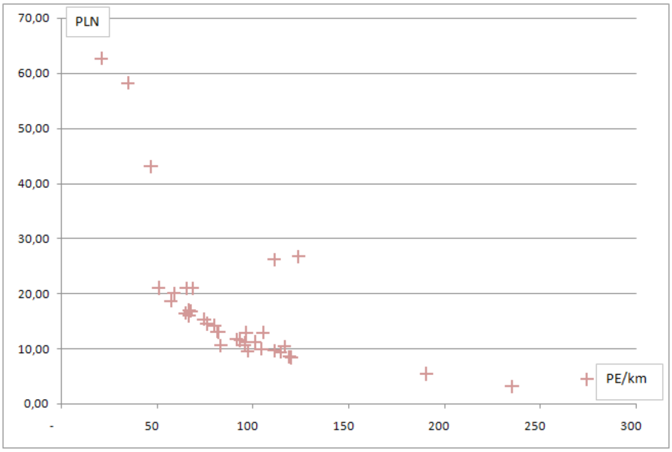

Figure 6: Relationship between wastewater tariffs (Y-axis) and population density indicator (PDI shown in PE/km) – calculated for the investments that have just been realized or are planned for tendering in the foreseeable future.

Fig. 6 shows the relationship between wastewater tariffs and population density indicator, calculated for the investments that are being realized or planned for realization. Due to a higher amount of investments the thesis postulated above can be clearly observed. All of the calculations were performed for the specific Company's conditions and local determinants in the same manner as shown above. It must be pointed out that the lesser investment the more it is prone to disturbances, such as obligatory roadwork or necessity for wastewater pumping stations. Such situations can be observed in some cases where despite high PDI the tariff is high due to necessity of pumping station). Also the investments can be very easily rationalized. Sometimes it is enough to reduce the investment scope by one or two elongated pipelines (usually connecting individual dwellings) to increase PDI and reduce wastewater tariff. Such optimizations should be every-day practice during investments planning, tendering and realization.

Analyzing Fig. 6 the tariff-rules seem to shape as follows:

– if PDI is ≈ 50÷70PE/km, the tariffs would be some 15÷20 PLN/m^3, or even more
– if PDI is ≈ 80÷110PE/km, the tariffs would be some 10 PLN/m^3,
– if PDI is ≈ 120 PE/km or more, the tariffs would be lower than 10 PLN/m^3, and the higher the indicator is the lower the tariff.

The question arises now whether such relationships would occur all across the country. Analyzing various data sources (Polish Chamber of Waterworks, 2012, http://www.cena-wody.pl, http://www.cenywody.pl) it seems that intensive investments conducted on rural areas in conjunction with the assets assigned to companies (and not to districts or other local authorities) may contribute to tariffs increase. This seems to be a general rule with the exception as below.

In the case of suburbs and rural areas adjacent to bigger towns or cities where the tariffs are calculated altogether, the above-mentioned principles will not necessarily hold true. The mechanism of costs subsidizing occurs, which means that inhabitants from city centers pay a bit higher wastewater prices in order to keep a bit lower (than would be calculated) prices in the suburbs. One may assume that this would be the social cost of agglomerations' development – if population of urban areas wishes to be provided with nearby ecosystem services such as leisure, greens, camping, etc., they would have to agree on paying more for wastewater services that would subsidize neighboring rural areas.

12.3.2 Household's Available Income And Tariffs System

Household's available income (or disposable income) is the amount of money that households have available for spending and saving after income taxes have been deducted. The available income varies depending on the region of Poland and on the size of city/town. It is assumed that water and wastewater charges combined should not be higher than 3% of household's available income (JASPERS, 2006). It was calculated that available income in the region where the Company operates (southern Poland) and taking into account the size of settlements was PLN 913.58 monthly. This means that water and wastewater services expenditures should not be higher than PLN 27.41 per person per month.

Table 3 shows the expenditures for water and wastewater services per person as a function of tariff's level and the difference between these costs and calculated 3%

Table 3: Available income vs. water and wastewater tariffs.

Tariffnet [PLN/m³]	Tariff gross [PLN/m³]	Monthly charges [PLN/person]	3% avail. inc. – sewage charges [PLN]	3% avail. inc. – (sewage+water) charges [PLN]
4.63	5.00	12.50	14.91	7.41
6.48	7.00	17.50	9.91	2.41
9.26	10.00	25.00	2.41	-5.09
11.11	12.00	30.00	-2.59	-10.09
13.89	15.00	37.50	-10.09	-17.59
18.52	20.00	50.00	-22.59	-30.09

of available income. The negative values mean exceeding the 3% threshold and this means non-accomplishment of the very basic assumption – that the services should be financially available for the population. It was assumed that the typical water tariff is some 3.00 PLN/m^3. But this figure will be lower for most rural areas due to individual water supply solutions, with the mean of 3.43 PLN/m^3 for the whole of Poland (Polish Chamber of Waterworks, 2012). It must be also noted that the tariffs calculated above were given in net values and available income is given in gross values, which means that for the sake of comparability value added tax (VAT) had to be included.

The conclusions from the above do not seem to be optimistic:

1. The high population density (PD=120 PE/km and more) will produce the overall tariffs of 5-7 PLN/m^3 and this would include all the obligatory costs as specified in the Polish legal system. Unfortunately, such population density will be very rare to appear in rural areas.
2. The investment realization will produce the tariffs of 10 PLN/m^3 and more in most of the Districts serviced by the Company.
3. The wastewater tariffs at the level of 10 PLN/m^3, together with water tariffs can cause the charges for these services to exceed 3% of available income. Thus the completion of "*the polluter pays*" rule will cause this indicator of service availability to be exceeded.
4. The entire system sustainability achieved by the full realization of "*the polluter pays*" rule and covering of all the costs (including amortization, taxes, payroll, materials, and all others) leads to inability to provide wastewater services or its limiting in rural areas.
5. The abovementioned can lead in practice to:

– abandoning of wastewater investments in rural areas,
– worsening of environmental conditions due to a lack of rational and good alternatives for sewage collection and treatment systems (the septic tanks system does not meet the legal and environmental requirements in the Polish conditions for now).
– the necessity of tariff subsidizing by the local authorities.

It should be emphasized very strongly that while the general conclusions drawn can be extended to other organizations, the specific numbers must be applied to the Company exclusively and having in mind the local conditions and factors. A change in any factor may lead to strictly different conclusions. Also note that the calculated available income at the level of PLN1,300 (310 €), as it was calculated for the other parts and regions of Poland(JASPERS, 2006), will cause different values in Table 5 and so the 3% threshold would not be exceeded for the tariffs as high as 12 PLN/m^3.

12.3.3 And What Now?

Taking into consideration the conclusions above one should think of some means to change the current situation concerning rural areas. The most costs generating the tariffs arise from the assets-related costs amortization and depreciation and property tax. In some of the cases they will contribute to as far as 70÷80% of tariffs. What would the tariffs be if the assets-related costs were not included? The theoretical calculations are shown in Fig. 7. All the calculations were performed based on the Company situation, using the formula stated above and following all the abovementioned assumptions.

It is very clear that if the assets-related costs were not included into the calculation it is highly unlikely that the tariffs would exceed 10 PLN/m^3 and even 7 or 8 PLN/m^3 is reached in a couple of cases only. As it can be concluded from Table 3 only those cases can lead to overcrossing of 3% available income indicator. As a consequence it leads to the conclusion that if the assets-related costs were not included into tariffs

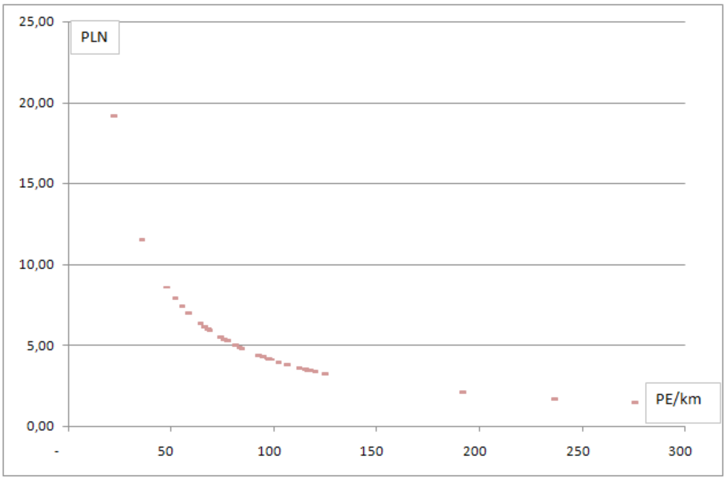

Figure 7: The wastewater tariffs (Y-axis) with the assets-related cost excluded (the costs of investments do not influence the tariffs).

Table 4: The consequences of a shift in tariffs calculation methodology.

No.	Positive effects	Negative effects
	Tariffs lowering, better services availability	Legal nonconformity – in the case of the Company (and any company based on the Polish commercial code)
	Decrease of potential environmental pollution	Funds collection for future reconstructions is not possible
	No need for subsidizing, will release some investment potential for the local authorities	Local authorities have to fund investments.
	Reduction of local communities' tension and discontent	-

calculation, the water and wastewater services availability would greatly increase. On the other hand, all the implications of such a paradigm shift must be clearly stated. Table 4 summarizes the most important implications of such a potential change in the calculation methodology.

It seems that there are social effects on the one hand and economic and environmental effects on the other. It has to be remembered that resigning of assets-related costs calculation must go along with the strategic planning of investment politics within districts. Abandoning of investments and renovations would lead to complete depreciation of all assets and negligence of their basic functions. And this would be a straight way to increase environmental pollution. Such situations have already occurred. The cases of investments finished 10÷20 years ago and then "forgotten" in operation are visible and are now subject of total modernization and renovation works.

The author of this chapter encountered various solutions for the problems stated above. The author's personal opinion is simply that there is not a single or best remedy that can be unambiguously recommended. Most likely, a specific solution is highly dependent on local conditions, financial situation of districts and local authorities, wellness of community, or debt level of the organization responsible for water and wastewater systems operation. The current conditions of water and wastewater services in rural areas in Poland are quite disadvantageous – both for contractors (local authorities) and consumers. It seems that deep and essential system changes are necessary. If no changes are applied in the near future this may lead to a situation where services' financial availability will be limited due to very high costs of water and wastewater services and this would lead to increasing loads of contaminants and nutrients discharged to the environment.

Also, a better caution should also be imposed on the use of alternative methods for sewage management, namely septic tanks and small wastewater treatment plants. Proper management and control systems should be introduced. It is very common that the so called septic tanks in rural areas are not sealed and/or they are emptied over the farmlands. These acts are of course forbidden but the lack of control systems allows for such practices quite often. The reason for such misdoings lies in the costs of proper septic tank management. Emptying and transporting of the full tank load to wastewater treatment plants would amount to some PLN 300 while the cost of spreading the content over the farmland is near zero. And this is why the amount of the sewage transported to wastewater treatment facilities from septic tanks is some 2% of the amount generated by the population. The law enforcement does not exist in this case (but this is the subject for yet another discussion). Taking into consideration some of the tariffs calculated in the tables above it has to be noted that some of them exceed 20÷30 PLN/m^3. The prices for wastewater collection and treatment by the pipelines and by transport from septic tanks would be equal with the tariff of 35 PLN/m^3. This would undermine the sensibility of wastewater pipelines construction and point the necessity of septic tanks management and control system – provided that the wastewater stored in them is transported to treatment plants.

As pointed above – all the services have a price. The price must be calculated rationally according to legal regulations and allowing to cover the expenditures borne by the investor. The realization of wastewater systems requires that beneficiaries of these systems cover the costs of the system. In the author's understanding these should cover the running costs at least but full realization of *the polluter pays* rule will rather require that they cover investments costs as well. And this would lead to calculation of amortization into the tariffs too. Considering all the above, in the author's opinion the demands of the *Right2water* initiative (e.g. guaranteed sanitation for all, global access to sanitation) would lead to unsustainability in water and wastewater services and failure in applying *the polluter pays* principle.

One has to be aware of the limitations of the method shown in this chapter. All the numbers, calculations and specific data are applicable for the Company alone and if such calculations were to be conducted for any other organization the specific numbers and values should be recalculated according to local conditions. Also, the described implications may not occur in towns or cities and in cases where the investments do not contribute to a major expansion of existing wastewater infrastructure or the communities are better-off.

12.3.4 Possible Recommendations

Based on the author's experience, effects of the described situation and proposals for some changes that could lead to the increase of wastewater services availability are listed below. These should not be treated as the sole solutions nor the ones that would be successful in every situation.

1. The current model for tariffs' calculation according to the Polish legal system may lead to an extraordinary increase of the costs that have to be covered by the end users. Therefore full realization of "*the polluter pays*" principle and achieving full sustainability of wastewater systems is a big challenge in rural areas.

2. It seems to be very rational that all the running costs are calculated (included) in the tariffs and this could be a measure for wastewaters' systems sustainability. On the other hand, the decision whether or not the assets-related costs are included in the tariffs should be made by local authorities largely accounting for local conditions. Unfortunately this would require some changes in the existing legal system in order to cancel the necessity of assets-related costs calculations, in parallel with pointing out of sources for investment and reconstructions planning. Abandoning obligatory property tax calculation (2% of assets value) alone would lead to a remarkable tariffs decrease in some cases.
3. If the construction of wastewater collecting systems was to be fully sustainable and to cover all the costs of wastewater management, this may lead to tariffs increase beyond the level of social acceptability and financial availability. And this would cause maintaining the present, relatively high level of environmental pollution.
4. It seems that the population density indicator at 120 PE/km is quite rational. The investments realized at the areas of lower PDI may cause the increase of costs and tariffs and this could lead to crossing the limit of 3% of available income for water and wastewater services – as shown in the examples in this chapter. Of course PDI should be treated with care because in some situations even despite very low PDI the 3% limit would not be exceeded and therefore PDI can be treated as a very fast and auxiliary indicator of sustainability. Tariffs and economic availability of services (measured by percentage of available income spent on water and wastewater services) should be the ultimate indicators of sustainability.
5. This chapter shows the discrepancy between environmental effects of the investments in rural areas and the possibilities to meet all the related costs by the local communities. Clearly, the social costs are contradictory to environmental effects.
6. Cost balancing for large-scale systems (e.g. the province-scale, or more than one agglomeration) could lead to averaging the costs between cities and villages, lowering the tariffs in rural areas and thus increasing the accessibility of wastewater services.
7. Such solutions exist in Poland but they are quite rare. Such solutions may also be opposed because they do not really meet the "*polluter pays*" rule – citizens of higher population density areas would pay for those from rural areas – and nobody likes to pay for anybody else. Nevertheless tariffs calculation for wider areas covering a few cities/towns/agglomerations/villages could be beneficial for environmental protection.
8. As the matter of fact the current system of tariffs calculation in Poland makes it very difficult to assess the degree of sustainability of the infrastructure built due to the fact that environment-related decisions are sometimes taken without considering technical and economic aspects.

Having said that it may be beneficial for the local communities to have the wastewater investments completed by local governments and then handing the ready-for-operation assets over to managing organizations (assets-related costs would be omitted). Also individuals that are interested in collecting systems' development could build pipelines under the supervision of a responsible organization and then hand them over to the organization.

As the ultimate solution, local subsidizing in any case when the tariffs exceed 3% of available income could be obligatory. Currently this solution is difficult to implement in some districts due to internal Polish regulations for financing and subsidizing local authorities. This solution would make the picture of sustainability even more frayed but the availability of the sanitation services would likely increase and so would the environmental gains.

12.4 Conclusions

The Company described in this chapter is a good example of the unwanted side-effects of environmental investments. The calculated tariffs are too high to bear by the local communities. It seems that maintaining the full sustainability of wastewater systems in the rural areas as shown in the chapter may cause tariffs increase to a level beyond social and financial acceptability. If the sustainability indicators as shown (PDI – population density indicator, available income) had been used and applied in a manner described in this chapter the financial and social problems could have been avoided. Alternatively, subsidizing or large scale cost balancing could be introduced but prior to this some law changes may be necessary.

Wastewater systems seem to be the best solution but the technical, technological, economic and legal issues must be strongly addressed. The use of septic tanks or small wastewater treatment plants will rarely secure environmental protection on the same level as wastewater pipelines connected to large-scale wastewater treatment plant.

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13 Conclusions - Sustainability Indicators In Practice: Lessons Learned From The Past, Directions For The Future

Agnieszka E Latawiec and Dorice Agol

This book presents various case studies from around the world and we hope that for the reader it will serve as a handbook for lessons learned on practical use of sustainability indicators. Evaluating development projects using sustainability indicators continues to be a dynamic research field and those practicing sustainability are also creating it. As emerged from many of the case studies discussed here, there is rarely, if at all, an ideal indicator that fully encompasses all the desired qualities and features of 'a perfect indicator'. Indicators are always subject to controversy, subjectivity and preferences of their users. Yet as shown in this book there are indicators that, from expert opinion, address the critical issues of sustainability in a specific context (being it fishery, wastewater, forestry) in a more (or less) comprehensive manner. These indicators can aid in the understanding and measurement of the progress of development better than 'traditional' indicators, for reasons discussed in specific chapters. Moreover, as often discussed throughout the book, 'traditional' indicators are not wrong *per se*. Quite the opposite, they may provide important information and should accompany what is claimed to be a 'sustainability indicator'. In addition, the data used for delivering a specific indicator are often readily available and also can show progress over time. Sometimes it is also the interpretation and application of the data behind a traditional indicator that transform it into something that gives a notion of sustainability. For example, data on income as a traditional indicator can be used to provide purchase parity, which tells us more about the social sustainability of the system.

In decision-making processes, choosing the right indicator is often a challenge. There are always trade-offs when selecting and using sustainability indicators and it is important to be transparent and acknowledge limitations. Many of the chapters in this book suggest solutions that worked in individual case studies. This does not mean that we have a 'silver bullet', but it shows that a range of practitioners around the world (Europe, Latin America and Africa) faced similar challenges and proposed solutions that worked in practice.

This book covers a range of topics from different case studies but what became apparent is that many of the practical challenges associated with sustainability outlined were common for Brazil, Poland and Kenya. This is despite different economic, cultural and biophysical contexts. Solutions to the challenges were often common too. Moreover, although goals of the projects were different and set in different sectors and frameworks, there are some general conclusions that can be drawn:

1. There are discrepancies between social, economic and environmental priorities within the projects, hence capturing these priorities by indicators is challenging (e.g. chapters 5, 7 and 12);
2. What is measured by sustainability indicators should depend on the goal that was set before a certain intervention (e.g. chapter 1, 8 and 11). For example, if ecological restoration is the goal, the mere reforested area may not be sufficient to indicate the success of restoration. Instead, rather the diversity of species planted and their survival should be taken into consideration (chapter 8);
3. Indicators are needed for decision making but they are only useful when, in addition to being placed in a specific cultural context, there is also a clear understanding of previous interventions. In other words, if indicators are used without understanding the processes and people they are relevant to, they may be easily misused (even if an indicator is good itself). This aspect was discussed virtually in all chapters of this book and strongly stressed in chapters 5, 10 and 12;
4. Monitoring is fundamental for development and use of sustainability indicators. The importance of monitoring is demonstrated, in particular, in chapters 3, 4 and 7;
5. Indicators can be both quantitative and qualitative (e.g. chapter 1). Yet, they should both be a measure rather than a vague approximation of what is intended to be measured. It is extremely important that indicators are well developed and carefully chosen in order to avoid wrong representations and decisions, which might result in negative consequences for sustainable development. Quantitative assessments are particularly important when dealing with issues such as pollution as discussed in chapters 9, 10, 11 and 12;
6. Most of the chapters also refer to conceptual problems with interpretations of sustainability and its subjectivity. Practitioners are often compelled to act on behalf of future generations, hence the indicators should capture future scenarios. Yet, how to define what should be left for the future generations can be difficult. How do we know what the future generations will value? In that respect, how can one define what is a ''fair'', ethical and ''the sustainable thing to do''? And how can we think of future generations if so many current generations do not meet their basic needs? There are no obvious answers on how this should be dealt with and it is the practitioner who often needs to make the ultimate decision. Being aware of these issues is however necessary. Acknowledging limitations and subjectivity can certainly help negotiation processes and consensus on various debates and real-world situations within which sustainability indicators are often used. The reader can find more on this topic in chapters 1 and 2.
7. The previous point on subjectivity is heavily linked to issues with values. The conflicts between human wellbeing, environmental conservation and economic development are a challenge when developing sustainability indicators. Wellbeing is dependent on culture, society or spiritual traditions and it may rise as more resources are used. Although human wellbeing is the ultimate end for

measuring sustainability, assessing what wellbeing actually means has been a question for thousands of years and a subject of great philosophical deliberation. That discussion has not produced one answer. To various people wellbeing means something different, it is a matter of values, culture and other complex factors. So how to measure the most qualitative, personal, culture-bound subjective yet the most important part in sustainability debate? And if we cannot assess what wellbeing is, how can we ever get there? How do we know if we are going in the right direction? This topic is discussed in depth in chapter 2.

8. Acknowledging multiplicity and ambiguity in relation with the use of sustainability indicators was also discussed throughout the chapters (e.g. chapters 1, 2, 9 and 10). Our conviction strongly affects our perception of the world and understanding of it is critical to the discussion on sustainability indicators. We therefore need to acknowledge and correct for this diversity. The subjectivity of the perception of indicators and preference for different sustainability indicators is heavily embedded in the ways in which we perceive the world. There is therefore a need to acknowledge these multiplicity and ambiguity when working with indicators, and understand and accommodate multiple views on sustainability.

Notwithstanding subjectivity, multiplicity, ambiguity and complexity associated with the use of sustainability indicators, we have already enough evidence from practice about what may and what may not lead to sustainability. We hope that this book, which shows some of these examples, will enrich the existing knowledge on the use of sustainability indicators. This may help practitioners in the field, by applying some of the solutions that are suggested here, and spur more interest in sustainability indicators. Obviously, we are not exhaustive and we present only a snap shot of a bigger picture. Yet we believe that a range of environmental aspects discussed (water, land, air) from different places and cultures, brings some representative picture at least for a part of the world. Although much remains to be done on the search for desirable indicators we hope that our book will encourage more work, participation and collaboration in this extremely relevant, complex and interesting topic. Just because sustainability is difficult and many do not really know how to make up a truly sustainable world (especially as different views exist on what a sustainable world is), that does not mean that we cannot do things to be less unsustainable, and pursue sustainability.

As many scientific articles, books and reports of a similar scope after discussing the selection, uses and misuses of indicators propose characteristics of an ideal indicator, we also initially thought that given the information from the chapters we would also present a table listing features of a good and a bad indicator. Instead, however, we propose to the reader, to make their own decision on choosing their "ideal" sustainability indicator, based on the succinct information on indicator use in practice that each chapter has provided. Easy or not, working with sustainability indicators is a fascinating adventure and we invite everybody to join.

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More at: http://www.iis-rio.org/en/equipe/alvaro#sthash.B2xOm9aI.dpuf

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