ELSEVIER

TREATISE ON COASTAL AND ESTUARINE SCIENCE - CONTRIBUTORS? INSTRUCTIONS

PROOFREADING

The text content for your contribution is in final form when you receive proofs. Please read proofs for accuracy and clarity, as well as for typographical errors, but please DO NOT REWRITE.

At the beginning of your chapter there is a page containing any author queries, keywords, and the authors? full address details.

Please address author queries as necessary. While it is appreciated that some chapters will require updating/revising, please try to keep any alterations to a minimum. Excessive alterations may be charged to the contributors. *Publication is scheduled for March 2012. As you are among the authors who are receiving their proof in 2010, you will be offered an opportunity to receive an updated proof prior to publication, in order to ensure that your chapter is up to date. We will be in touch at a later date about this.*

The shorter version of the address at the beginning of the chapter will appear under your author/co-author name(s) in the published work and also in a List of Contributors. The longer version shows full contact details and will be used to keep our internal records up-to-date (they will not appear in the published work). For the lead author, this is the address that the honorarium and any offprints will be sent to. Please check that these addresses are correct.

Titles and headings should be checked carefully for spelling and capitalization. Please be sure that the correct typeface and size have been used to indicate the proper level of heading. Review numbered items for proper order ? e.g., tables, figures, footnotes, and lists. Proofread the captions and credit lines of illustrations and tables. Ensure that any material requiring permissions has the required credit line, and that the corresponding documentation has been sent to Elsevier.

Note that these proofs may not resemble the image quality of the final printed version of the work, and are for content checking only. Artwork will have been redrawn/relabelled as necessary, and is represented at the final size.

PLEASE KEEP A COPY OF ANY CORRECTIONS YOU MAKE.

DESPATCH OF CORRECTIONS

Proof corrections should be returned in one communication to **Laura Jackson, at the Elsevier MRW Production Dept**, by **29-Dec-10** using one of the following methods:

1. **PREFERRED**: Corrections should be listed in an e-mail to **Laura Jackson** at Elsevier MRW Dept at ESCO-Proofs@elsevier.com. Please do not send corrections to the Editors.

The e-mail should state the chapter code number in the subject line. Corrections should be consecutively numbered and should state the paragraph number, line number within that paragraph, and the correction to be made.

2. If corrections are substantial, send the amended hardcopy by courier to **Laura Jackson, Elsevier MRW Production Department, The Boulevard, Langford Lane, Kidlington, Oxford, OX5 1GB, UK**. If it is not possible to courier your corrections, please fax the relevant marked pages to the Elsevier MRW Production Department **(fax number: +44 (0)1865 843974)** with a covering note clearly stating the chapter code number and title.

Note that a delay in the return of proofs could mean a delay in publication. Should we not receive corrected proofs within 10 days, Elsevier may proceed without your corrections.

CHECKLIST

Author queries addressed/answered? ❑

Affiliations, names and addresses checked and verified? ❑

?*References*? section checked and completed? ❑
Please take this opportunity to bring the critical references in your chapter up to date

Permissions details checked and completed? ❑

Outstanding permissions letters attached/enclosed? ❑

Figures and tables checked? ❑

If you have any questions regarding these proofs please contact the Elsevier MRW Production Department at: ESCO-Proofs@elsevier.com.

Author Query Form

Treatise on Estuarine and Coastal Science

Article: 01206
Dear Author,

Please respond to the queries listed below. You may write your comments on this page, but please write clearly as illegible mark-ups may delay publication. If returning the proof by fax do not write too close to the paper's edge.

Thank you for your assistance.

AUTHOR QUERIES

AU1	Please check the long affiliations for accuracy. These are for Elsevier's records and will not appear in the printed work.
AU2	Please provide telephone number and fax number of all the authors, if available.
AU3	Please supply the chapter number, along with its title, intended here.
AU4	'Yeo et al. (2002)' is not listed. Please provide complete details of this reference.
AU5	Do you mean US dollar?
AU6	'Carpenter et al. (2008)' is not listed. Please provide complete details of this reference.
AU7	'Duarte (1991)' is not listed. Please provide complete details of this reference.
AU8	Please provide biographical sketch of the authors 'SD Hacker', 'EW Koch', and 'AC Stier' with color photographs.
AU9	'Koch (1996)' is not listed. Please provide complete details of this reference.
AU10	'Prager and Halley (1999)' is not listed. Please provide complete details of this reference.

AU11 OK as edited?

AU12 'Silliman and Bertness (2003)' is not listed. Please provide complete details of this reference.

AU13 'Zimmerman et al. (2007)' is not listed. Please provide complete details of this reference.

AU14 OK as edited?

AU15 'Valiela and Teal (1979)' is not listed (cited more than once). Please provide complete details of this reference.

AU16 'Bromberg and Silliman (2009)' is not listed (cited more than once). Please provide complete details of this reference.

AU17 'Buckely (1987)' is not listed. Please provide complete details of this reference.

AU18 'Ruggiero et al. (2008)' is not listed. Please provide complete details of this reference.

AU19 'Liu et al. (2005)' is not listed. Please provide complete details of this reference.

AU20 OK as edited?

AU21 'Petersen et al. (2003)' is not listed. Please provide complete details of this reference.

AU22 'Rountree and Able (2007)' is not listed (cited more than once). Please provide complete details of this reference.

AU23 'Aguilar-Perera et al. (2008)' is not listed (cited more than once). Please provide complete details of this reference.

AU24 OK as edited?

AU25 OK as edited?

AU26	References 'Beer and Koch (1996)', 'Glynn (1997)', 'Petersen et al. (2008)', 'Ruggiero (1998)', 'Silliman and Bertness (2002)', 'Yeo (2002)', and 'Zimmerman et al. 2000' are not cited in the text. Please check.
AU27	Please provide place of publication for all the book and book chapter type references, where missing.
AU28	Please provide page range and place of publication of the reference 'Henry and Jefferies (2009)'.
AU29	Please provide the editor name(s) for the reference 'Millennium Ecosystem Assessment (2005)'.
AU30	Please check if permission is required for all figures and tables. If yes, please supply permission letters.

Non-print Items

Author's Contact Information

AU1 **Edward B Barbier**
Department of Economics and Finance
AU2 University of Wyoming
Laramie
WY
USA
E-mail: ebarbier@uwyo.edu

Sally D Hacker
Department of Zoology
Oregon State University
Corvallis
OR
USA
E-mail: hackers@science.oregonstate.edu

Evamaria W Koch
Horn Point Laboratory
University of Maryland Center for Environmental Science
Cambridge
MD
USA
E-mail:koch@hpl.umces.edu

Adrian C Stier
Department of Biology
University of Florida
Gainesville
FL
USA
E-mail: astier@ufl.edu

Brian R Silliman
Department of Biology
University of Florida
Gainesville
FL
USA
E-mail: brs@ufl.edu

Keywords: Coral reef; Economic value; Ecosystem service; Estuarine and coastal ecosystem; Mangrove; Salt marsh; Sand dune; Seagrass; seascape

Biographical Sketches

Edward B. Barbier, PhD, is the John S Bugas Professor of Economics, Department of Economics and Finance, University of Wyoming. He was formerly at the Environment Department, University of York, UK, and previously served as Director of the London Environmental Economics Centre of the International Institute for Environment and Development and University College London. Professor Barbier has over 25 years' experience as an environmental and resource economist, working on natural resource and development issues as well as the interface between economics and ecology. He has served as a consultant and policy analyst for a variety of national, international, and non-governmental agencies, including many United Nations (UN) organizations and the World Bank. Professor Barbier serves on the editorial boards of several leading economics and natural science journals, and he appears in the 4th edition of *Who's Who in Economics*. He has authored over 150 peer-reviewed journal articles and book chapters, written or edited 17 books and published in popular journals. Some of his well-known works include *Blueprint for a Green Economy* (with David Pearce and Anil Markandya, 1989), *Natural Resources and Economic Development* (2005), and the UN Environment Programme report, *A Global Green New Deal* (2009).

AU8

Dr. Brian R. Silliman received his PhD degree from Brown University, Providence, RI, USA, in 2004, after finishing his thesis Top-down vs. Bottom-up Control of Western Atlantic Salt Marshes, after which he started as an Assistant Professor at the University of Florida. Since then, he was awarded the American Society of Naturalists' Prestigious Young Investigators' Prize, an Andrew Mellon Foundation Young Investigator Grant and has become a Fellow, both with the Nature Conservancy as a David Smith Conservation Post Doctoral Research Fellow and at the Center for Advanced Studies in Ecology and Biodiversity, Catholic University, Santiago, Chile. Brian Silliman has become internationally renowned for his work on top-down control by fungal-feeding snails on salt-marsh vegetation. In particular, he has demonstrated that the regionwide collapse of salt-marsh vegetation along the eastern and southern-US marshes is driven by the interaction of climatic stress and grazer pressure, amplified by human-related environmental impacts. Key in understanding vegetation collapse was the formation and movement of snail fronts that greatly amplified the effects of grazing and led to runaway vegetation collapse at large spatial scales. Currently, he is leading an international network studying how climate change shifts the importance of facilitation, disease, and predation in coastal plant ecosystems. This effort has been funded jointly by the National Science Foundation and the Andrew Mellon Foundation, and culminated in publication of the book *Human Impacts on Salt Marshes*, of which Dr. Silliman is the primary editor.

The Department of Spatial Ecology of the NIOO-KNAW studies the processes that determine the spatial structure and dynamics of estuarine ecosystems, focusing on salt marshes and intertidal flats. The department's central theme is to study how the interaction of organisms with physical processes determines the spatial structure of estuarine ecosystems, for instance, the patchy vegetation structure typical of salt marshes. We propose to combine the expertise of Dr. Brian Silliman (experimental food web ecology) and the expertise of the department of Spatial Ecology, which includes detailed studies on the interaction between vegetation and hydrodynamic processes (Bouma), the use of geographic information systems (GIS) techniques (Van der Wal), and modeling spatial self-organization (Van de Koppel) to study the spatial dynamics of both US and European salt marshes. Dr. Silliman's experience in setting up experiments to approach ecological interactions from multiple angles compliments our expertise in modeling and GIS techniques to build predictive ecological theory on the impacts of climate change on salt-marsh ecosystems with a solid foundation in both mathematics and experiments, as is witnessed by past publications in *Science and Ecology Letters* (pubs 43 and 44).

a0005 # 12.06 Estuarine and Coastal Ecosystems and Their Services

EB Barbier, University of Wyoming, Laramie, WY, USA
SD Hacker, Oregon State University, Corvallis, OR, USA
EW Koch, University of Maryland Center for Environmental Science, Cambridge, MD, USA
AC Stier and BR Silliman, University of Florida, Gainesville, FL, USA

Abstract

The global decline in estuarine and coastal ecosystems (ECEs) is affecting a number of critical benefits, or ecosystem services. We review the main ecological functions and their services across a variety of ECEs, including marshes, mangroves, near-shore coral reefs, seagrass beds, and sand beaches and dunes. We cite estimates of the key economic values arising from these services, and discuss how the natural variability of ECE impacts their benefits, the synergistic relationships of ECE across seascapes, and the management implications.

s0005 ## 12.06.1 Introduction

p0005 Estuarine and coastal ecosystems (ECEs) are some of the most heavily used natural systems globally; it is now recognized that the cumulative impacts from a range of human activities are threatening many of the world's remaining ECEs and the many benefits they provide (Lotze et al., 2006; Worm et al., 2006; Halpern et al., 2008). More than one-third of the world's population lives in coastal areas and small islands, which together make up just 4% of the Earth's total land area. Coastal human population densities are nearly 3 times that of inland areas, and they are increasing exponentially (UNEP, 2006). The long-term sustainability of these populations depends on ECEs and the critical services they provide, such as storm buffering, fisheries production, and enhanced water quality.

p0010 Yet, despite the importance of these services, the loss of ECEs is intense and increasing, such that 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and 29% of seagrasses are either lost or degraded worldwide (Valiela et al., 2001; Millennium Ecosystem Assessment, 2005; Orth et al., 2006; UNEP, 2006; FAO, 2007; Waycott et al., 2009). For example, the global decline of ECEs is known to affect at least three critical ecosystem services (Worm et al., 2006): the number of viable (noncollapsed) fisheries (33% decline); the provision of nursery habitats such as oyster reefs, seagrass beds, and wetlands (69% decline); and filtering and detoxification services provided by suspension feeders, submerged vegetation, and wetlands (63% decline). Loss of filtering services is also linked to declining water quality and the increasing occurrence of harmful algal blooms, fish kills, shellfish and beach closures, and oxygen depletion. The decline in biodiversity and ecosystem functions in ECEs may have contributed to biological invasions and vice versa. Increasingly, the loss or change of coastal vegetation in ECEs has affected these systems' ability to protect against coastal flooding and storm events (Braatz et al., 2007; Cochard et al., 2008; Koch et al., 2009).

Such widespread transformation of ECEs and their ecosystem services suggests that it is important to understand further what is at stake in terms of critical benefits and values. The purpose of the following chapter is to provide an overview of the main ecological functions and their services across a variety of ECEs, including marshes, mangroves, near-shore coral reefs, seagrass beds, and sand beaches and dunes. Where possible, we cite estimates of the key economic values arising from the services provided by these ECEs. In addition, we discuss how the natural variability in these systems, both in space and time, can produce nonlinear functions and services that greatly influence their economic value (Barbier et al., 2008; Koch et al., 2009). We discuss some of the synergistic effects of ECE. Because they occur at the interface between the coast, land, and watersheds, ECEs can produce cumulative benefits that are much more significant and unique than the services provided by any single ecosystem. Finally, we end by highlighting the main management implications of this review of ECE services and their benefits to humankind. p0015 AU3

s0010 12.06.2 Definitions of Services and Values of ECE

p0020 In classifying the ecosystem services provided by various ECEs, we adopt the broad definition of the Millennium Ecosystem Assessment (2005) that "ecosystem services are the benefits people obtain from ecosystems" (see also 01202). Thus, the term 'ecosystem services' is usually interpreted to imply a variety of benefits, which, in economics, would normally be classified under three different categories (Daily, 1997; Barbier, 2007): (1) goods (e.g., products obtained from ecosystems, such as resource harvests, water, and genetic material); (2) services (e.g., recreational and tourism benefits or certain ecological regulatory functions, such as water purification, climate regulation, and erosion control); and (3) cultural benefits (e.g., spiritual and religious beliefs, and heritage values) (see also 01207, 01208, 01209, 01210, 01211, and 01212).

p0025 Regardless of how one defines and classifies ecosystem services, "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values" (Heal et al., 2005: 2). Table 1 provides some examples of the links between regulatory and habitat functions and the ecosystem services from ECEs that ultimately benefit humankind.

p0030 Thus, one pertinent feature of many ECEs is that they provide multiple benefits, or values, to surrounding coastal populations and communities. As Table 2 indicates, these benefits of ECE services cover a wide variety of use and nonuse

t0005 **Table 1** Some services provided by ecosystem regulatory and habitat functions

Ecosystem functions	*Ecosystem processes and components*	*Ecosystem services (benefits)*
Regulatory functions		
Gas regulation	Role of ecosystems in biogeochemical processes	Ultraviolet-B protection Maintenance of air quality Influence of climate
Climate regulation	Influence of land cover and biologically mediated processes	Maintenance of temperature, precipitation
Disturbance prevention	Influence of system structure on dampening environmental disturbance	Storm protection Flood mitigation
Water regulation	Role of land cover in regulating runoff, river discharge, and infiltration	Drainage and natural irrigation Flood mitigation Groundwater recharge
Soil retention	Role of vegetation root matrix and soil biota in soil structure	Maintenance of arable land Prevention of damage from erosion and siltation
Soil formation	Weathering of rock and organic matter accumulation	Maintenance of productivity on arable land
Nutrient regulation	Role of biota in storage and recycling of nutrients	Maintenance of productive ecosystems
Waste treatment	Removal or breakdown of nutrients and compounds	Pollution control and detoxification
Habitat functions		
Niche and refuge	Suitable living space for wild plants and animals	Maintenance of biodiversity Maintenance of beneficial species
Nursery and breeding	Suitable reproductive habitat and nursery grounds	Maintenance of biodiversity Maintenance of beneficial species

From Adapted from Heal, G.M., Barbier, E.B., Boyle, K.J., Covich, A.P., Gloss, S.P., Hershner, C.H., Hoehn, J.P., Pringle, C.M., Polasky, S, Segerson, K., Shrader-Frechette, K., 2005. Valuing Ecosystem Services: Toward Better Environmental Decision Making. The National Academies Press, Washington, DC., table 3-3.

t0010 **Table 2** Various values provided by tropical coastal and marine ecosystems

Use values		*Nonuse values*
Direct values	*Indirect values*	*Existence and bequest values*
Fishing	Nutrient retention and cycling	Cultural heritage
Aquaculture	Flood control	Resources for future generations
Transport	Storm protection	Existence of charismatic species
Wild resources	Habitat for species	Existence of wild places
Water supply	Shoreline stabilization	
Recreation		
Genetic material		
Scientific and educational opportunities		

From Adapted from Barbier, E.B., 1994. Valuing environmental functions: tropical wetlands. Land Economics 70, 155–173 and Heal, G.M., Barbier, E.B., Boyle, K.J., Covich, A.P., Gloss, S.P., Hershner, C.H., Hoehn, J.P., Pringle, C.M., Polasky, S, Segerson, K., Shrader-Frechette, K., 2005. Valuing Ecosystem Services: Toward Better Environmental Decision Making. The National Academies Press, Washington, DC., table 2-1.

values, which can have direct, indirect, and existence and bequest subcomponents (see also 01203).

p0035 For example, typical direct-use values, which refer to both consumptive and nonconsumptive uses that involve some direct physical interaction with the ecosystem and its services, include harvesting of fish and wild resources, transportation by waterways, recreation, and tourism. Some unique estuarine and coastal habitats are also important stores of genetic material and have educational and scientific research value as well. However, in developing regions, some of the more important uses of ECEs tend to involve both small-scale commercial and informal economic activity to support the livelihoods of local populations, for example, through fishing, hunting, and fuelwood extraction.

p0040 Some of the regulatory and habitat functions of ecosystems identified in Table 1 underlie key services provided by ECEs. As indicated in Table 2, these include indirect use values such as nutrient retention and cycling, flood control, storm protection, habitat for species, and shoreline stabilization. Such values are considered to be indirect because they are derived mainly from the support and protection of economic activities and livelihoods that have directly measurable values (Barbier, 1994). For example, in the case of mangrove ecosystems, the mangrove swamps may serve as a nursery and breeding habitat for many important fish species, some of which may migrate as adults to offshore fisheries. In addition, mangroves can provide storm protection by reducing the economic damages inflicted by tropical storms on coastal property and communities. Finally, mangrove systems are thought to prevent coastal erosion, thus preserving valuable agricultural land and coastal properties.

p0045 Many unique ECEs are considered to have substantial nonuse values, even in developing regions. These include existence and bequest values, which refer to people benefiting from the knowledge that an ecosystem simply exists or that it will be around for future generations to enjoy. These values may be particularly important among indigenous communities in rural areas, as they see their culture, heritage, and traditional knowledge closely intertwined with the surrounding environment. Even some of the poorest rural communities have expressed interest in seeing their way of life passed on to their heirs and future generations (Berkes, 1999).

p0050 In the following section, we provide an overview of the main ecological functions and their services for five ECEs. We describe the key services of the ecosystem and elaborate on its special ecological features, components, and functions that correspond to key services. We also cite, when possible, some estimates of key economic values from these services.

s0015 12.06.3 Descriptions of Services and Values of ECEs

s0020 12.06.3.1 Near-shore Reefs

p0055 Coral reefs are structurally complex limestone habitats that form in shallow coastal waters of the tropics. Reefs can form near-shore or extend hundreds of kilometers in shallow offshore environments. Primarily generated by calcium carbonate fixed by sedentary cnidarians (corals), coral reefs flourish in oligotrophic tropical waters by supplementing their diet with excess sugar generated by photosynthetic dinoflagellates inside their tissues. Thus, the majority of the reef structure is dead coral skeleton laid down over millennia, covered by a thin layer of live coral tissue that slowly accretes new limestone. As a generalization, coral species can be functionally divided into primary and late successional species. Primary successional species are massive boulder-shaped corals that have dense skeletons and are therefore resistant to disturbance and form the foundation of the reef (e.g., Montastrea in the Atlantic and Porites in the Pacific). As time passes, initial colonists are outcompeted by fragile but structurally complex late successional species (e.g., corals of the genus *Acropora* and *Pocillopora*), which form much of the habitat occupied by the diverse suite of invertebrates and fishes. The benthic composition of reefs can, however, vary substantially across oceanic basins, longitude, latitude, distance for biodiversity hot spots, and across physical disturbance gradients. The composition of reef communities can additionally be affected by ecological processes such as predation, competition, and mutualisms (Glynn, 1976; Connell et al., 1997; Pandolfi, 2002; Bellwood et al., 2005). For example, the crown-of-thorns sea star, a voracious coral predator, can substantially alter the composition of coral communities during spikes in population size (Porter, 1972).

Coral reefs can provide a number of important ecosystem services to humans including building materials, shoreline protection, fisheries, tourism and recreation, medicinal products, habitat for a variety of species of heavily harvested invertebrates and fishes, cross-ecosystem nutrient transfer to birds and pelagic fishes, a window into geologic history, and an opportunity for education and scientific inquiry (Moberg and Folke, 1999; Table 3). These services are collectively estimated as having a value of approximately $30 billion annually worldwide (Cesar et al., 2003). p0060

Reefs have been providing services to humans as a coastal resource for millennia, facilitating the colonization of rocky land that is largely depauperate in terrestrial resources such as food and building materials. Indeed, the harvesting of fishes and invertebrates provided the provisional backbone (along with pigs and chickens) for early Polynesian expansions to rocky, eastern Pacific islands (though, notably, reefs were also treacherous to ships navigating uncharted waters). Historically, the mining of live reefs provided an excellent and reliable source of valuable lime, which served as both an essential material in the manufacturing of mortar and cement as well as used to control soil pH in agriculture (Dulvy et al., 1995). Presently, excavation of live reefs for lime is uncommon due to the obvious destructive nature of this resource extraction; however, terrestrial limestone deposits (geologic reefs that formed during times of higher sea level) are still heavily mined on land and continue to provide goods for building construction and agriculture endeavors. As one of the most diverse marine ecosystems on the planet, coral reefs offer opportunity for the discovery and sequestration of medical products (Paulay and Meyer, 2002). For example, azidothymidine (AZT) and Ara-c, compounds derived from sponges on coral reefs, are antiviral drugs used to fight human immunodeficiency virus (HIV) (Loya et al., 1993; Jain, 2009). A suite of other compounds (e.g., sunscreens) are sequestered from seaweeds, corals, gorgonians, mollusks, and sea anemones and used to treat a variety of other human diseases such as cancer, heart disease, and inflammation (Carté, 1996). p0065

Coral reefs also provide an important ecosystem service by facilitating island formation and persistence in the face of intense storms. For example, reefs buffer shorelines from severe p0070 t0015 **Table 3** Ecosystem services, processes and functions, important controlling components, and human drivers of service loss for near-shore coral reefs

Ecosystem services	*Ecosystem processes and functions*	*Important controlling components*	*Human drivers of service loss*
Coastal protection	Attenuates or dissipates waves, beach retention	Wave height, distance from reef crest	Climate change, blast or cyanide fishing, lime mining, eutrophication, sedimentation, coastal development, dredging, pollution, biological invasion
Tourism, recreation, education, and research	SCUBA and snorkel operations, aesthetic reefscapes and coastlines, water sports, tour boats	Lagoon size, beach area, wave height, shell availability, education and research funding, diversity	
Maintenance of fisheries	Suitable reproductive habitat and nursery grounds, suitable living space for species	Reef composition and density, habitat quality, water supply and quality	
Nutrient cycling	Biogeochemical activity, $CaCO_3$ sequestration, sedimentation, biological productivity	Reef composition and density, sediment deposition, subsidence, coastal geomorphology	
Cross-ecosystem nutrient transfer	Provides feeding and migration grounds for shorebirds and transfers nutrients to pelagic fisheries	Food availability, reef type, quality and availability, proximity to shore and pelagic zone, outer reef slope	

weather, making it possible for coastal housing and moderate levels of island-based agriculture to develop and persist on landmasses largely composed of rock and sand. Erosion and bioerosion of reefs also contributes to island growth, and on old sunken volcanic islands (atolls), old reef and sand reef makes up the entire livable space for humans. Finally, by altering the physical environment (i.e., reducing waves and currents), corals can engineer the physical environment for entire ecosystems, making it possible for other coastal ecosystems such as seagrass beds and mangroves to develop, which, in turn, serve their own suite of services to humans (detailed below).

p0075 Shoreline protection and island growth also provide picturesque placid lagoons for tourism and recreational activities, which can make up a high percentage of gross domestic product (GDP) for developing countries. Resorts sell the aesthetically turquoise lagoons, white sandy beaches, and underwater opportunities on the reef to tourists. The high biological diversity and clear waters of tropical reefs also support an abundance of recreational activities such as scuba diving, snorkeling, island tours, and sport fishing. These tourism activities can be highly lucrative for individual economies; for example, in 2002, the earnings of ~100 diver operators in Hawaii were estimated at \$50–60 million yr^{-1} (van Beukering
AU4 and Cesar, 2004). In addition, Yeo et al. (2002) estimated tourism in the Pulau Payar Marine Park at \$390 000 yr^{-1}. However, estimates of the recreational value of individual reefs should be interpreted with caution as a recent meta-analysis suggests substantial bias in author's estimates of individual recreation values (Brander et al., 2007). In addition to tourism and recreation, reefs also provide substantial services through research opportunities for scientists, providing rich geologic climate history in tree-like bands embedded in coral skeletons (Greenstein and Pandolfi, 2008). Warm, clear, tropical waters also give scientists the rare opportunity to spend extended time conducting subtidal research on extremely diverse fish, algal, and invertebrate communities, work that is essential to basic and applied science.

In addition to providing building materials, altering geo- p0080
morphology, stimulating tourism, recreation, and research, coral reefs also enhance production of ecologically and economically important fishery species by providing shelter space and substrate for smaller organisms and food resources for larger epibenthic and pelagic organisms. Increases in fishing technology and transport have transformed reef fisheries that initially functioned solely for subsistence, into commercial operations that serve international markets. Coral reef fisheries consist of reef-associated pelagic fisheries (e.g., tuna, mackerel, mahi-mahi, and sharks), reef fishes (e.g., jacks, snappers, groupers, and parrotfishes), and large invertebrates (e.g., giant clams, conch, lobsters, and crabs). In 1978, reef fisheries were estimated to make up 9–12% of the world's fish market (Smith, 1978). This market has undoubtedly grown in the past 40 years, though recent estimates of economic value focus on individual markets. For example, Hawaiian reefs in the US are estimated to contribute \$1.3 million yearly to the Hawaiian economy (Cesar and van Beukering, 2004). Additional fishery harvests consist of the live animal aquarium trade where corals, small fishes, and invertebrates are collected from reefs. The aquarium trade has substantially expanded in the past 20 years, listed in 1985 as making \$20–40 million yr^{-1} as a AU5
world market (Wood, 1985) and expanding to an estimated \$90–300 million yr^{-1} in 2002 (Sadovy et al., 2002). The export and sale of shells and jewelry also makes up a substantial portion of fisheries on reefs; giant clams, conch shells, coral, and pearls are all among the many heavily harvested organisms.

Coral reef ecosystems also perform important services by p0085
cycling organic and inorganic nutrients. Unlike the other coastal systems (e.g., salt marshes discussed below), reefs play

a relatively minor role in the global carbon budget. Indeed, despite housing a great deal of inorganic carbon in the limestone skeleton that makes up the structure of the reef, carbon budgets for reefs are thought to be estimated as a small fraction of the global carbon budget (Hallock, 1997; Gattuso et al., 1998). Reefs do, however, serve as major players in the global calcium carbonate budget, estimated as 26% of coastal marine $CaCO_3$ and 11% of the total $CaCO_3$ precipitation. Reefs additionally play a role in cross-ecosystem nutrient cycling, transferring a variety of nutrients through trophic links with pelagic organisms and marine birds. Of particular importance is transference of excess nitrogen production from cyanobacteria and benthic microbes on the reef to the pelagic environment (Moberg and Folke, 1999). Though poorly quantified, the sequestering of $CaCO_3$ to form the foundation of the reef is the primary reason for such high abundance and diversity of organisms; the consequences of losing this foundation species are discussed below.

p0090 Despite the numerous economic, health, and nutritional benefits coral reefs provide, reef ecosystems are under threat of irrevocable decline worldwide due to a suite of anthropogenic stressors. Localized stressors (i.e., within reefs or archipelagos) include overfishing, dynamite fishing, poisoning animals with cyanide treatments, pollution, eutrophication, and disease (Hoegh-Guldberg, 1999; Gardner et al., 2003; Bellwood et al., 2004; Hoegh-Guldberg et al., 2007). Though stressors may occur concurrently and combine in unpredictable nonadditive ways to affect reef ecosystem services, individual case studies attempting to place an economic value on these stressors provide some insight into the economic impact of coral degradation. For example, a study comparing tsunami wave heights and subsequent depths in the 2004 Sumatra tsunami found that areas where dynamite fishing had occurred suffered 70% greater wave heights than undisturbed areas. This ultimately led to the deaths of 1700 people whose train was derailed behind these degraded reef locations (Fernando et al., 2005). Blast fishing can also have negative effects on local economies by reducing the amount of available reef for tourisms; a study in Indonesian reefs estimated blast-fishing-induced habitat destruction led to the loss of reef that was valued at \$306 800 km^{-2} (Pet-Soede et al., 1999).

p0095 Overfishing has important cascading consequences on both reef ecosystem function by inducing phase shifts (Mumby et al., 2006, 2007) and through sustainable production. Overharvesting in the aquarium industry has also been documented at local levels. For example, Lubbock and Polunin (1975) reported severe depletion of a butterflyfish in Sri Lanka. Another example of overharvesting is in the live fish-food trade in Hong Kong, where it makes up 40% of the Indo-Pacific market and is estimated to be a yearly global market of \$1 billion yr^{-1}. Here, harvesting exceeds production of the Indo-Pacific fish populations by up to sixfold. In another example, fishing in Oceania for local consumption is estimated to be 20% less than the capacity, which underlines the importance of export in driving demand for reef ecosystem services (Warren-Rhodes et al., 2004). Though reefs may be somewhat capable of mitigating human impacts such as petroleum spills (Peterson and Lubchenco, 1997), a recent model estimates millions of dollars in lost tourism revenue through eutrophication-induced algal blooms in Hawaii (van Beukering and Cesar, 2004).

Though local-scale stressors and fisheries export can have devastating effects on reef services, perhaps of greatest concern are global-scale climate-change threats to reefs, which include coral bleaching, disease, and ocean acidification. A recent study placed 32.8% of 702 examined coral species at elevated risk of extinction due to bleaching, diseases, and local anthropogenic stressors (Carpenter et al., 2008). Current projections of reef decline at present-day levels of increase in temperature and CO_2 (2 °C and 500 ppm by 2100, respectively) are catastrophic for reefs (Hoegh-Guldberg et al., 2007). Such declines have the potential to render reefs crippled in their ability to provide economic services to humans. A number of studies have evaluated the economic consequences of climate-change-driven reef degradation on fisheries and tourism. p0100 AU6

The response of fisheries to climate change may be variable, depending on the direct dependence of fished species on live coral, structure, or trophic dependencies (reviewed in Pratchett et al. (2008)). Species dependent on live coral are the first to disappear with severe reductions in live coral cover; however, in the case of bleaching and disease, the limestone structure often remains well beyond the death of the coral, generating a 'lag-effect' whereby populations of fishes that depend on the reef for structure only, or mobile upper trophic levels (e.g., sharks, jacks, and barracuda) that use the reef for prey, may not decline for a number of years after bleaching or disease events (Graham et al., 2007). Though the economic effects of climate change on fisheries harvested for food remain somewhat unclear, what is clear is that reefs are likely to shift heavily in composition in response to benthic composition and, therefore, fisheries will at least change in composition, if not in overall take (Pratchett et al., 2008). The effects of climate change on the aquarium trade are much more obvious. Loss of reefs may substantially reduce the availability of the most-valued fishes collected in the aquarium trade. Of the species harvested by the aquarium trade, live coral-dependent species (those most susceptible to reef shifts from climate change) are, on average, worth 12% and 18% more in fishes per dollar than fishes that are solely habitat-dependent or reef-associated fishes (Pratchett et al., 2008). Reef tourism is also expected to be heavily impacted by climate change. Economic costs of bleaching on tourism are not present in all areas, though a number of surveys suggest that divers are more likely to pay higher prices to dive recreationally on healthy reefs (\$202 and \$52 per dive in the Philippines (Pratchett et al., 2008)). Reductions in tourism due to recent climate change-driven coral-bleaching events are estimated in billions of dollars. For example, the estimated losses in revenue from the 1998 bleaching event in the Indian Ocean are thought to be as much as \$3.5 billion in tourism revenue in a 20-year time period (Wilkinson et al., 1999). p0105

Considering the number of services coral reefs provide, current projected levels of reef degradation will lead to enormous economic losses for humans in the future. Though the mechanisms by which each of the reef stressors can cause decline are still being revealed, the solutions needed to reduce human impacts on reefs are relatively straightforward. They include combating climate change by reducing CO_2 emissions, alleviating destructive fishing practices and overfishing, and stemming terrestrial runoff of fertilizers, sediments, and pollutants. Putting these types of changes into effect is much more difficult than identifying the solutions; however, by communicating the benefits (and potential losses) of reefs to p0110 governmental officials, fishermen, and consumers, it may be possible to preserve the essential services reefs provide.

s0025 12.06.3.2 Seagrass Beds

p0115 Seagrasses are flowering plants that colonize shallow marine and estuarine habitats. With only one exception (the genus *Phyllospadix*), seagrasses colonize soft substrates (e.g., mud, sand, and cobble) and grow to depths where approximately AU7 11% of surface light reaches the bottom (Duarte, 1991). Seagrasses also prefer wave-sheltered conditions as sediments disturbed by currents and/or waves lead to patchy beds or their absence (Koch et al., 2006). Despite being among the most productive ecosystems on the planet, fulfilling a key role in the coastal zone (Duarte, 2002) and being lost at an alarming rate (Orth et al., 2006), seagrasses receive little attention when compared with other ECEs (Duarte et al., 2008).

p0120 Anthropogenic influences such as nutrient enrichment of coastal waters, sediment runoff, algal blooms, commercial fisheries and aquaculture practices, physical disturbance, and global warming are among the causes for the decline of seagrasses worldwide (**Table 4**; Orth et al., 2006). With the disappearance of seagrasses, valuable ecosystem services are also lost (McArthur and Boland, 2006).

p0125 In the past, seagrasses were highly valued for their provision of materials (**Table 4**). In the seventeenth century, coastal villagers made a variety of foods from flour produced from seagrass seeds (Felger et al., 1980). In the eighteenth century, large amounts of seagrass leaves were harvested to manufacture pillows, mattresses, and commercial insulating products for homes and some of the first skyscrapers (Wyllie-Echeverria and Cox, 1999). Modern direct uses of seagrasses are rather limited. For example, seagrasses are harvested in Tanzania, Portugal, and Australia and are used as fertilizer (Hemminga and Duarte, 2000; de la Torre-Castro and Rönnbäck, 2004). In Chesapeake Bay, USA, seagrass by-catch or beach-cast is used to keep crabs moist during transport. In East Africa, *Halophila* spp. is used as salad, whereas other species are used in magic potions (for good and bad influence) and rituals against ghosts and devils (de la Torre-Castro and Rönnbäck, 2004). Although human use of seagrasses has dwindled since the seventeenth century, they continue to be a main food source for organisms such as manatees, dugongs, and turtles. Actually, the common name of many seagrasses reflects the organisms that feed on them: for example, turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), widgeon grass (*Ruppia maritima*), dugong grass (*Halophila ovalis*), and spiny dugong grass (*Halophila spinulosa*).

Regulating ecosystem services provided by seagrasses include carbon sequestration, water purification, and coastal protection (**Table 4**). Seagrasses use carbon dissolved in the seawater (mostly in the form of CO_2 but also HCO_3^-) to grow. Once the plants complete their life cycle, a portion of these materials is then buried in the sediment in the form of refractory detritus. It has been estimated that detritus burial from vegetated coastal habitats contributes about half of the total carbon burial in the ocean (Duarte et al., 2005). Therefore, the decline in seagrasses could lead to an important loss in the global CO_2 sink capacity. p0130

Water purification by seagrasses occurs via two processes: nutrient uptake and particle removal (**Table 4**). Although seagrasses tend to remove most of their nutrients from the sediments, their leaves are colonized by algae (epiphytes), which also remove nutrients from the water column. The nutrients incorporated into the tissue of seagrasses and algae are slowly released back into the water column once the plants decompose, or are removed from the nutrient cycle when buried in the sediment. Nutrient cycling by seagrasses has been valued at \$19 004 ha yr^{-1} (Mathews, 1983). In addition to reducing nutrients, seagrass beds also decrease the concentration of suspended particles (e.g., sediment and microalgae) from the water. Leaves in the water column provide an p0135

t0020 **Table 4** Ecosystem services, processes and functions, important controlling components, and human drivers of service loss for seagrasses

Ecosystem services	*Ecosystem processes and functions*	*Important controlling components*	*Human drivers of service loss*
Raw materials and food	Biological productivity and diversity	Seagrass species and density, habitat quality, water depth, light and nutrient availability	Eutrophication, overharvesting, coastal development, vegetation disturbance, aquaculture, climate change, sea-level rise
Coastal protection	Attenuation and/or dissipation of waves	Water depth, wave height, wind climate, beach slope, seagrass species and density, reproductive stage, distance from shore	
Erosion control	Sediment stabilization	Sea-level rise, subsidence, tidal stage, coastal geomorphology, seagrass species and density	
Water purification	Nutrient uptake, particle deposition	Seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, light availability	
Maintenance of fisheries	Suitable reproductive habitat and nursery grounds, suitable living space and sheltered substrates for species	Seagrass species and density, habitat quality, proximity to other ECE, food sources	
Carbon sequestration	Photosynthesis	Seagrass species and density, water depth, light availability, burial rates, biomass export	

obstruction to water flow and, as a result, currents and waves are reduced within seagrass canopies (Koch et al., 2006) causing particles to be deposited. This water-purification effect can be quite dramatic, with clear water in vegetated areas compared to those without vegetation (Rybicki, 1997).

p0140 Coastal protection is often listed as an ecosystem service provided by seagrasses (Hemminga and Duarte, 2000; Spalding et al., 2003), but due to nonlinearities in wave attenuation, caution about generalizations is needed (Koch et al., 2009; see also Section 12.06.4). Seagrasses tend to attenu- AU9 ate waves (Fonseca and Cahalan, 1992; Koch, 1996; Prager and AU10 Halley, 1999) and, as a result, smaller waves tend to reach the adjacent shoreline. Wave attenuation is an inverse function of water depth (Fonseca and Cahalan, 1992); therefore, coastal protection will be highest when the plants occupy the entire water column such as at low tide or when plants produce long reproductive stems (Koch et al., 2006). When small seagrasses colonize rather deep waters, their contribution to wave attenuation and coastal protection is rather limited. What appears to be more important is sediment stabilization by seagrass roots and rhizomes as well as by their beach-casted debris (Hemminga and Nieuwenhuize, 1990). Therefore, the benefit seagrasses provide in terms of coastal protection is more localized and occurs mainly where the seagrasses grow (via sediment stabilization) and to a lesser extent at the adjacent shoreline (via wave attenuation).

p0145 Seagrasses are well known for their value as habitat for ecologically and economically important species such as scallops, shrimp, crabs, and juvenile fish (**Table 4**). Seagrasses protect them from predators and provide food in the form of leaves, detritus, and epiphytes. The potential shrimp yield in seagrass beds in Western Australia was estimated to be between \$684 and \$2511 ha yr^{-1} (Watson et al., 1993) and the fish, shrimp, and crab yield in southern Australia at US\$1436 ha yr^{-1} (McArthur and Boland, 2006).

p0150 Although not often included in the list of ecosystem services provided, seagrasses are also a source of recreation and scientific discovery (**Table 4**). Waters inhabited by seagrasses are cleaner and have higher species diversity compared to unvegetated areas; therefore, they provide beauty to be enjoyed by divers and snorkelers as well as fishes to be caught by anglers. Seagrass systems have also fostered 4 decades of scientific discovery employing faculty, students, technicians, and managers.

p0155 Considering all the benefits seagrass systems provide to humankind, their decline leads to tremendous losses, which range from the impacts on coastal villages that depend on fisheries facilitated by seagrass habitat to the global effects of carbon removal from the oceans via burial of seagrass detritus. For example, the loss of most seagrasses in Long Island in the 1930s due to wasting disease led to the collapse of the scallop industry (Orth et al., 2006). Management and preservation of seagrass beds is needed to assure the continuation of provision of ecological and social benefits to humankind.

s0030 12.06.3.3 Salt Marshes

p0160 Salt marshes are intertidal grasslands that form in low-energy, wave-protected shorelines along continental margins. Extensive salt marshes (many >2 km in width) establish and grow both behind barrier-island systems and along the wave-protected shorelines of bays and estuaries. These systems are generated primarily from foundation plants at lower tidal levels such as *Spartina*, Phragmites, and a variety of rushes. At higher reaches of salt marshes, woody plants such as *Iva*, *Myrica*, and *Baccharus* may also play key bioengineering roles (Bertness, 2006). Salt marshes are characterized by sharp zonation of plants and low plant species diversity. Individual AU11 marshes typically have 5–15 plant species, with the worldwide total reaching over 400 (Marc Spalding, personal communication). The structure and function of salt-marsh plant communities (and thus their services) were long thought to be regulated by physical processes, such as elevation, salinity, flooding, and nutrient availability (Odum and Smalley, 1959; Mitsch and Gosselink, 2008). Over the past 25 years, however, experiments have shown that competition (Bertness, 1991) and facilitation (Hacker and Bertness, 1995) among marsh plants is also critically important in controlling community structure. More recently, research has revealed the presence of strong trophic cascades driven by habitat-destroying herbivorous grazers (Silliman and Bertness, 2003; Silliman et al., 2005; AU12 Henry and Jefferies, 2009).

p0165 Among coastal ecosystems, salt marshes provide a high number of valuable benefits to humans including water filtration, buffering of storm waves and surges, carbon sequestration and burial, critical habitat for both adult and juvenile fishes and birds, grasses for building houses and baskets, land for grazing ungulates and development, and scientific and educational opportunities (Morgan et al., 2009; **Table 5**).

p0170 For over 8000 years, humans have benefited greatly from salt marshes and relied on them for direct provisioning of materials (**Table 5**, Davy et al., 2009). In the Fertile Crescent of the Middle East, as well as in Europe and North America, salt marsh grasses were harvested to provide thatch for roofs, wall material for small houses, and baskets for holding food and goods. Additionally, salt marshes in these areas were used as grazing pastures for cattle and sheep, and other domesticated animals. Starting roughly 2000 years ago and to this day, marsh grasses are still purposely planted and protected by the Dutch so as to act as buffers against storms surges and as natural-engineering tools to reclaim shallow seas and build up sea barriers to facilitate greater human reclamation and development (Davy et al., 2009). Indeed, over 40% of the land in present-day Netherlands was once estuarine habitat and was reclaimed with the help of the engineering services of salt marsh plants (Davy et al., 2009). Although the harvesting of marsh grasses and the use of salt marshes as pasture lands have decreased today, these services are still important locally in both developed and developing areas of the world (Bromberg-Gedan et al., 2009).

p0175 Besides directly providing humans with building materials for shelter, feed for cattle, and ways of reclaiming estuarine habitat, salt marsh ecosystems also help boost production of economically and ecologically important fishery species, such as shrimp, oysters, clams, and fishes (MacKenzie and Dionne, 2008; Boesche and Turner, 1984). For example, salt marshes may account for 66% of the shrimp and 25% of the blue crab production in the Gulf of Mexico (Zimmerman et al., 2007). AU13 Field research has revealed many mechanisms by which salt marshes boost production in these fisheries. Because of their complex and tightly packed plant structure, marshes provide habitat that is mostly inaccessible to large fishes, thus providing a predator-free or predator-reduced habitat for the increased t0025 **Table 5** Ecosystem services, processes and functions, important controlling components, and human drivers of service loss for salt marshes

Ecosystem services	*Ecosystem processes and functions*	*Important controlling components*	*Human drivers of service loss*
Raw materials and food	Biological productivity and diversity	Marsh grass species type and density, marsh quality and area, healthy predator populations	Marsh reclamation, vegetation disturbance, climate change, sea-level rise, pollution, altered hydrological regimes, biological invasion
Coastal protection	Attenuates or dissipates waves	Tide height, water depth, wave height, wind velocity, marsh area and width, marsh grass species and density, healthy predator populations	
Erosion control	Provides sediment stabilization and soil retention in vegetation root structure	Sea-level rise, fluvial sediment deposition and load, subsidence, coastal geomorphology, marsh grass species and density, distance from sea edge, healthy predator populations	
Water purification	Provides nutrient and pollution uptake, as well as retention	Marsh grass species and density, marsh quality and area, nutrient and sediment load, water supply and quality, healthy predator populations	
Maintenance of fisheries	Provides suitable reproductive habitat and nursery grounds, suitable living space for species	Marsh grass species and density, marsh quality and area, primary productivity, healthy predator populations	
Maintenance of birds, other species for recreational uses	Provides suitable reproductive habitat and feeding and rearing grounds	Marsh grass species type, and density, habitat quality and area, prey species availability, healthy predator populations	
Carbon sequestration	Generates biogeochemical activity, carbon sequestration, sedimentation, biological productivity	Marsh grass species and density, sediment type, primary productivity, healthy predator populations	

growth and survival of young shrimp, fishes, and shellfish (Boesche and Turner, 1984). In addition, young animals utilizing marshes also benefit from increased food supply of small invertebrates and primary producers, such as plants, detrital plant particles, and diatoms (Boesche and Turner, 1984). Finally, salt marshes provide juvenile animals with a stable and relatively low-energy physical regime, which reduces chances of dislodgement or death due to heavy-wave energy (Boesche and Turner, 1984).

p0180 In addition to the direct provisioning of resources for humans, salt marshes also provide important regulatory ecosystem services such as carbon sequestration, coastal protection, and water purification. By being one of the most productive ecosystems in the world (up to 3900 g cm^{-2} yr^{-1}), and one that is widely distributed, salt marshes sequester millions of tons of carbon per year (Mitsch and Gosselink, 2008). AU14 Because of the anoxic nature of the marsh soils (as in most wetlands), carbon sequestered by salt marsh plants during photosynthesis is often lost from the short-term carbon cycle (10–100 years) to the long-term carbon cycle (1000 years) as buried, slowly decaying biomass in the form of decaying root material (Mitsch and Gosselink, 2008; Mayor and Hicks, 2009). This cycle-shifting capability is unique among many of the world's ecosystems, where carbon is mostly turned over quickly and does not often move into the long-term carbon cycle. Thus, salt marshes should be elevated along with ecosystems such as Arctic tundra and boreal forests, as those that provide the important ecosystem service of carbon sequestration and transition into the long-term carbon cycle (Mayor and Hicks, 2009). Currently, however, this critical ecosystem service generated by salt marshes is just beginning to be recognized by managers, economists, and scientists.

For over 2000 years, salt marshes have been considered as p0185 natural barriers that protect our coastlines from waves and storm surge (Davy et al., 2009). By generating higher, accreting land between sea and terrestrial ecosystems, providing baffling vertical structures (grass), and stabilizing sediment, salt marshes reduce impacts of incoming waves by reducing their velocity, height, and duration (Morgan et al., 2009). In addition, because bioengineering by both burrowing marsh crabs (Bertness, 1984) and grasses provides increased interstitial space and porosity for holding greater amounts of water per unit volume of soil, marshes are also likely to reduce storm surge duration and height by providing extra water uptake and holding capacity in comparison to the highly compact sediments of unvegetated mudflats. Recent research in coastal protection services cautions, however, that there are nonlinearities in the generation of coastal protection by salt marshes and other coastal ecosystems such that larger areas do not necessarily produce greater protection, especially for larger waves such as those produced by tsunamis and hurricanes (Barbier et al., 2008; Koch et al., 2009). Regardless, there is little doubt that salt marshes provide coastlines with significant protection from the erosion and destruction caused by boat- and mid-sized storm-generated waves. The latter economic value can be substantial, as a study of the protection against hurricanes by coastal wetlands along the US Atlantic and Gulf coasts reveals (Costanza et al., 2008). Coastal protection should thus be included as a critical service generated by salt marsh ecosystems. The functional components and the quantitative details of its generation, however, must be explored to a greater degree.

p0190 Like riparian wetlands and coastal forests, salt marshes act as natural, vertically oriented filters that purify water entering the estuary (Mitsch and Gosselink, 2008). As water (e.g., from rivers, terrestrial runoff, ground water, or rain) passes over marshes, it slows due to the baffling and friction effect of upright grasses (Morgan et al., 2009). Suspended sediments then deposit on the marsh surface. Those sediments contain nutrients as well as toxic pollutants. Both of these deposits are taken up by salt marsh grasses and sequestered in plant tissue
AU15 (Valiela and Teal, 1979). Some of the bio-sequestered pollutants may enter the estuarine food web, but many are buried in the sediment during the biodeposition process described above. After this filtration process, water leaving the marsh either in tides, groundwater, or as overland flow is significantly less polluted and clearer (Valiela and Teal, 1979). This water-filtration service benefits humans, who use nearby waters for swimming and fishing, as well as adjacent ecosystems and the services they provide, such as seagrasses, which are more easily overwhelmed and killed by nutrients (Valiela and Teal, 1979).

p0195 Despite this list of abundant and valuable critical services, salt marshes are currently severely threatened worldwide (50% are now lost or degraded) by an extensive portfolio of human-
AU16 generated threats (Bromberg and Silliman, 2009). Most of these threats are currently underestimated or even overlooked by coastal conservation managers (Table 5) because marsh-preservation practitioners have historically worried most about stopping reclamation efforts (Silliman et al., 2009). Current threats to salt marshes include human-precipitated species invasions, small- and large-scale eutrophication and accompanying plant species declines, runaway grazing by snails, geese crabs, and nutria that denude vegetated marsh substrate over vast extents, climate-change induced effects including sea-level rise, increasing air and sea-surface temperatures, increasing CO_2 concentrations, altered hydrologic regimes, and a wide range of pollutants including nutrients, synthetic hormones, metals, organics, and pesticides (Silliman et al., 2009). Already, about 50% of the value of services marshes provide have been lost as salt marsh ecosystems have been degraded or lost (Bromberg-Gedan et al., 2009). On some coasts, such as the West Coast of the US, this number rises above 90%, for both marsh area and their services (Bromberg and Silliman, 2009). Without proper conservation action, it is now predicted that this key coastal community will become a nonsignificant ecosystem-service generating habitat in less than 100 years (Silliman et al., 2009). Key to saving salt marsh ecosystems is recognizing a wide variety of threats and abating them through up-to-date conservation strategies (Silliman et al., 2009) and providing justification of these conservation measures by both describing and valuing all of the critical services marshes provide.

12.06.3.4 Mangroves s0035

Mangroves are coastal forests that inhabit saline tidal areas along sheltered bays, estuaries, and inlets in the tropics and subtropics throughout the world. Around 50–75 woody species are designated as 'mangrove', which is a term that describes both the ecosystem and the plant families that propagate in this saline tidal environment (Ellison and Farnsworth, 2001). In the 1970s, mangroves may have covered as much as 200 000 km^2, or 75% of the world's coastlines (Spalding et al., 1997). However, in recent decades, human development pressures have caused severe mangrove deforestation globally. p0200

At least 35% of global mangrove area has been lost over the past 2 decades, and mangroves are currently disappearing at the rate of 1–2% annually (Valiela et al., 2001; Alongi, 2002; FAO, 2007). Although many factors contribute to global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of shrimp farms (Barbier and Cox, 2003). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38% of mangrove deforestation. Forest use, mainly from industrial lumber and woodchip operations, causes 26% of mangrove loss globally. Freshwater diversion accounts for 11% of deforestation, and reclamation of land for other uses causes 5% of decline. The remaining sources of mangrove deforestation consist of herbicide impacts, agriculture, salt ponds, and other coastal developments (Valiela et al., 2001). p0205

The worldwide destruction of mangroves is of concern because they provide a number of highly valued ecosystems services (see Walters et al. (2008) for a recent review). Mangroves are directly exploited for a large variety of wood and nonwood forest products, and they serve as habitat and spawning grounds for a variety of fish and shellfish that are important for commercial, recreational, and coastal fisheries. Mangroves are considered natural barriers that protect the lives and properties of coastal communities from periodic storm events, flooding, and shoreline erosion. Mangroves also serve as barriers in the other direction; their natural filtration functions protect coral reefs, seagrass beds, and important navigation waters against siltation and pollution. Because mangroves are among the most productive and biogeochemically active ecosystems, they are important sources of global carbon sinks and biological diversity, including a number of endangered mammals, reptiles, amphibians, and birds. Finally, for many coastal communities, their traditional use of mangrove resources is often closely connected with the health and functioning of the system, and thus, this use is often intimately tied to local culture, heritage, and traditional knowledge. Some key mangrove ecosystem services, processes and functions, important controlling components, and human drivers of service loss are summarized in Table 6. p0210

Of the ecosystem services described above, three have received most attention in terms of determining their value to coastal populations. These include: (1) their use by local coastal communities for a variety of products, such as fuelwood, timber, raw materials, honey and resins, and crabs and shellfish; (2) their role as nursery and breeding habitats for offshore fisheries; and (3) their propensity to serve as natural 'coastal storm barriers' to periodic wind and wave or storm-surge events, such as tropical storms, coastal floods, typhoons, and tsunamis. Assigning a value to these three mangrove p0215 t0030 **Table 6** Ecosystem services, processes and functions, important controlling components, and human drivers of service loss for mangroves

Ecosystem services	*Ecosystem processes and functions*	*Important controlling components*	*Human drivers of service loss*
Raw materials and food	Biological productivity and diversity	Vegetation type and density, habitat quality	Mangrove disturbance, degradation, conversion; coastline disturbance; pollution; upstream soil loss; overharvesting of resources
Coastal protection	Attenuates or dissipates waves and wind energy	Wave height, wind velocity, beach slope, tide height, vegetation type and density, distance from sea edge	
Erosion control	Provides sediment stabilization and soil retention in vegetation root structure	Sea-level rise, fluvial sediment deposition, subsidence, coastal geomorphology, vegetation type and density, distance from sea edge	
Maintenance of fisheries	Provides reproductive habitat and nursery grounds, suitable living space for species	Vegetation type and density, habitat quality, water supply and quality	
Pollution control	Provides water storage and filtration, nutrient retention	Vegetation type and density, nutrient load, water supply and quality	
Carbon sequestration	Biogeochemical activity, carbon sequestration, sedimentation, biological productivity	Vegetation type and density, fluvial sediment deposition, subsidence, coastal geomorphology	

ecosystem services has been conducted for Thailand (Barbier, 2007). The estimates show that the net present value arising from the net income to local communities from collected forest products over 1996–2004 was \$484–584 ha^{-1} (in 1996 dollars). In addition, the net present value of mangroves as breeding and nursery habitat in support of offshore artisanal fisheries ranged from \$708 to \$987 ha^{-1}, and the storm protection service was \$8966–10821 ha^{-1}. Such benefits are considerable when compared to the average incomes of coastal households; a survey conducted in July 2000 of four mangrove-dependent communities in two different coastal provinces of Thailand indicates that the average household income per village ranged from \$2606 to \$6623 yr^{-1}, and the overall incidence of poverty (corresponding to an annual income of \$180 or lower) in all but three villages exceeded the average incidence rate of 8% found across all rural areas of Thailand (Sarntisart and Sathirathai, 2004). The authors also found that excluding the income from collecting mangrove forest products would have raised the incidence of poverty to 55.3% and 48.1% in two of the villages, and to 20.7% and 13.64% in the other two communities.

p0220 The Thailand example is not unusual; coastal households across the world typically display considerable direct and indirect use values for mangroves (Ruitenbeek, 1994; Bandaranayake, 1998; Barbier and Strand, 1998; Naylor and Drew, 1998; Janssen and Padilla, 1999; Rönnbäck, 1999; Badola and Hussain, 2005; Chong, 2005; Brander et al., 2006; Walton et al., 2006; Rönnbäck et al., 2007; Aburto-Oropeza et al., 2008; Walters et al., 2008). However, there is also evidence that coastal inhabitants hold important nonuse values associated with mangroves. A contingent valuation study of mangrove-dependent coastal communities in Micronesia demonstrated that the communities “place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves’ marketable products” (Naylor and Drew, 1998: 488).

Assessing the values of ecosystem services provided by mangroves is proving to be important for two land-use policy decisions concerning these systems globally. First, as we have seen, conversion of remaining mangroves to shrimp farm ponds and other commercial coastal developments continues to be a major threat to the world’s remaining mangrove areas. Second, since the December 2004 Indian Ocean tsunami, there has been considerable interest in rehabilitating and restoring mangrove ecosystems as natural barriers to future coastal storm events. Valuing the goods and services of mangrove ecosystems can therefore help address two important policy questions: Do the net economic returns to shrimp farming justify further mangrove conversion to this economic activity, and is it worth investing in mangrove replanting and ecosystem rehabilitation in abandoned shrimp farm areas? As raised later in this chapter, an important consideration is the scale at which decisions to convert mangroves to shrimp farms is made. In addition, as other chapters in this volume point out, the various ecological, economic, and social issues underlying such decisions need to be considered carefully. p0225

Table 7 illustrates how the benefits of mangrove ecosystem services help address these questions, again by using the example from Thailand. As the table indicates, the net present value of the benefits for all three mangrove ecosystems ranges from \$10158 to \$12392 ha^{-1}. These ecosystem service values clearly exceed the net economic returns to shrimp farming. In fact, the net income to local coastal communities from collected forest products and the value of habitat-fishery linkages total to \$1192–1571 ha^{-1}, which are greater than the net economic returns to shrimp farming. However, the value of the storm protection is critical to the decision as to whether or not to replant and rehabilitate mangrove ecosystems in abandoned p0230 t0035 **Table 7** Comparison of land use values per ha for mangroves in Thailand, 1996–2004 (US$)

Land use	*Net present value ha^{-1} (10–15% discount rate)*
Shrimp farming	
Net economic returns[a]	1078–1220
Mangrove ecosystem rehabilitation	
Total cost[b]	8812–9318
Ecosystem goods and services	
Net income from collected forest products[c]	484–584
Habitat-fishery linkage[d]	708–987
Storm protection service[e]	8966–10 821
Total	10 158–12 392

[a]Based on annual net average economic returns US$322 ha^{-1} for 5 years from Sathirathai and Barbier (2001), updated to 1996$.
[b]Based on costs of rehabilitating abandoned shrimp farm site, replanting mangrove forests, and maintaining and protecting mangrove seedlings. From Sathirathai and Barbier (2001), updated to 1996 US$.
[c]Based on annual average value of $101 ha^{-1} over 1996–2004 from Sathirathai and Barbier (2001), updated to 1996 US$.
[d]Based on a dynamic analysis of mangrove-fishery linkages over 1996–2004 from and assuming the estimated Thailand deforestation rate of 3.44 km^2 y^{-1} (see Barbier 2007).
[e]Based on marginal value ha^{-1} of expected damage function approach of Barbier (2007).

pond areas. As shown in Table 7, storm-protection benefit makes mangrove rehabilitation an economically feasible land-use option.

p0235 To summarize, valuing the ecological services of mangroves and assessing the ecological functions underlying them is becoming increasingly important, given that irreversible conversion of mangroves for aquaculture and other coastal developments is resulting in the rapid worldwide loss of these ecosystems. The economic benefits of various mangrove services must be taken into account in land-use decisions that lead to the widespread conversion of mangroves. Although the use of mangroves from forest and nonwood products is important to the livelihoods of many coastal communities, the largest economic benefits of mangroves appear to arise from regulatory and habitat functions, such as the role of mangroves in supporting offshore fisheries and in providing protection against periodic storms. This reinforces the importance of measuring the value of such ecological services, and employing these values appropriately in coastal management and planning.

s0040 ### 12.06.3.5 Sand Beaches and Dunes

p0240 Coastal sand beaches and dunes are important but understudied arbiters of coastal ecosystem services. They form at low-lying coastal margins where sand transported by oceanic waves and wind combine with vegetation to produce dynamic geomorphic structures. Thus, sandy shore ecosystems include both marine and terrestrial components and vary, depending on sand supply, in the extent to which the beach versus the dune dominates (Short and Hesp, 1982). Sandy beaches and dunes occur at all latitudes on earth and cover roughly 34% of the world's ice-free coastlines (Hardisty, 1994). Some countries have extensive sand shorelines including 100% of the Netherlands, 60% of Australia, and 33% of the United States. For centuries, due to their unique position between ocean and land, coastal beaches and dunes have provided humans with important services such as coastal defense, subsistence fisheries, water catchment, agriculture, sand mining, and tourism (Table 8; Carter, 1990; Pye and Tsoar, 1990). Archeological and historical evidence from sand shorelines shows them to be important components of early New Zealand and Peruvian cultures (Parsons, 1968; Hesp, 2000) and of Western Europe coastal development, particularly in the Netherlands and England (van der Meulen et al., 2004). Many of the functions, and thus services, provided by sand beaches and dunes are threatened by human use and climate change (Brown and McLachlan, 2002). In particular, the removal or disruption of sand and vegetation, coupled with increased storm intensity and sea level rise, threaten critical services provided by this ecosystem, specifically those of coastal protection and coastal freshwater catchment.

Coastal protection is arguably one of the most valuable services provided by sand-shore ecosystems, especially in the face of extreme storms, tsunamis, and sea-level rise (Table 8). p0245
As waves reach the shoreline, they are attenuated by the beach slope as well as by the foredune, a structure immediately behind the beach where sand accumulates in hills or ridges parallel to the shoreline. Beaches vary in their ability to attenuate waves, depending on a continuum in their morphology (Short, 1999). At one end, dissipative beaches occur where high-wave energy coincides with ample sediment to produce broad, gentle slopes. These beaches are superior at attenuating waves and tend to have better-developed foredunes. At the other end, reflective beaches occur where lower-energy waves coincide with less sediment to produce narrower and steeper slopes. Waves break on the beach face producing high turbulence and wave energy reflected back to the ocean. Reflective beaches, because they occur where sand transport is lower, tend to have narrower foredunes when present. Thus, in terms of diffusing high wave energy characteristic of coastal storms and surges, the coastal protective services of dissipative beaches are generally better than that of reflective beaches (Short, 1999). Dissipative beaches allow high waves to break further seaward and dissipate energy over a larger area, thus decreasing wave force and erosion. In contrast, large waves hit reflective beaches considerably landward and can cause major erosion and scarping of the beach and the foredune. As a consequence of these

t0040 **Table 8** Ecosystem services, processes and functions, important controlling components, and human drivers of service loss for sand beaches and dunes

Ecosystem services	*Ecosystem processes and functions*	*Important controlling components*	*Human drivers of service loss*
Raw materials and food	Biological productivity and diversity, habitat for wild and cultivated animal and plant species	Dune and beach area, water and nutrient supply, vegetation and prey biomass and density	Loss of sand through mining, development and coastal structures (e.g., jetties), vegetation disturbance, overuse of water, pollution, biological invasion
Coastal protection	Attenuates or dissipates waves and reduces flooding from sea-level rise	Wave height, beach slope, tide height, sand supply, dune height, vegetation type and density	
Water catchment and purification	Stores and filters water through sand; raises water table	Dune area, dune height, sand and water supply	
Sand extraction	Provides sand of particular grain size, proportion of minerals	Dune and beach area, sand supply, grain size, proportion of desired minerals (e.g., silica, feldspar)	
Recreation	Provides unique and aesthetic landscapes for beach and inshore recreational activities	Dune and beach area, sand supply, grain size, wave height, wildlife, desirable shells and rocks	

differences in beach geomorphology, all things being equal, sea-level rise should be better accommodated by dissipative compared to reflective beaches and foredunes (Carter, 1991; Hesp and Short, 1999).

p0250 Foredunes can vary in height and width, and thus their ability to attenuate waves, depending on the presence of vegetation and sand supply from the beach (Hesp, 1989). Studies show that the mere presence of vegetation can produce a 30-fold decrease in sand transport over foredunes (Olson, 1958) with the type of vegetation and density playing an important AU17 role (e.g., Cowles, 1899; Buckely, 1987; Hesp, 1989). For example, tall, dense canopies such as those of beach grasses (e.g., *Ammophila*), shrubs, and trees reduce sand-laden wind very rapidly, thus causing significant sand deposition. Shorter, more compact plants such as those found for creeping species (e.g., *Cakile*) allow wind to blow over the surface, thus decreasing sand deposition. Nonetheless, if plant species identity is held constant but plant density is increased, there is a corresponding increase in foredune height (e.g., Buckley, 1987; Hesp, 1989; Kuriyama et al., 2005). Despite the role of vegetation, the ultimate shape of a foredune depends on sand supply (Psuty, 1986). Positive sand deposition from the beach onto the foredune creates shorter but wider foredunes compared to no net change in sand deposition, which produces taller and narrower foredunes. Negative sand supply produces shorter and narrower foredunes that are more susceptible to washover and erosion.

p0255 Measuring the coastal protective properties of sand shoreline systems involves understanding the relationship between beach and foredune shape and wave attenuation. Beyond examining what happens after storms, hurricanes, or tsunamis (e.g., Leatherman, 1979; Lui et al., 2005,;Ruggiero et al., 2005; Sallenger et al., 2006; Morton et al., 2007; Stockdon et al., 2007) or testing sand barriers in wave tanks (e.g., Overton et al., 1990; Larson et al., 2004), the most feasible method to estimate wave attenuation is through modeling (Ruggiero et al., 2001, 2008; Larson et al., 2004; Stockdon et al., 2006, AU18 2007;Stockdon and Thompson, 2007; Barbier et al., 2008). The models show that many factors, including the severity of the waves and storm surge, beach and dune morphology, vegetation composition and density, the tides, and the proximity of humans to the shoreline, are responsible for the degrees to which beaches and foredunes protect humans living behind them. Research on the coastal protective services of beaches and dunes is beginning to show that they can significantly reduce the impact of devastating coastal storms at a landscape scale albeit in a nonlinear manner (Barbier et al., 2008). The economic value, although not calculated previously, is likely to be substantial. For example, Liu et al. (2005) reported that, after AU19 the 2004 Indian Ocean tsunami, there was total devastation and loss of 150 lives in a resort located directly behind where a foredune once stood. The dune and its vegetation were removed to gain a better view of the ocean. It is clear that continued research is critical to understand the influence of sand shorelines, especially in tsunami-prone regions and in areas experiencing sea-level rise and heightened storm intensity due to climate change.

p0260 Another important service of coastal sand ecosystems is as water catchment (Table 8). Sand dunes are able to store significant amounts of water that can serve as aquifers for coastal populations. Along the coast, both salt water and freshwater permeate the sand but, because freshwater is less dense, it overrides and forms a lens that depresses the underlying salt water (Carter, 1990). For every meter of freshwater, however, there are roughly 40 m of salt water; thus, small changes in aquifer depth can cause salt-water intrusion making the water unsuitable for drinking. Some common ways by which water tables can be lowered, thus increasing the risk of contamination, are through using more water than is being recharged and/ or through sand removal such as with mining. For these AU20 reasons, maintaining dune aquifers requires considerable planning and management. For example, in the Meijendel dunes in the Netherlands, dune aquifers were used as a source of drinking water for centuries until the mid-1950s when salt water began to intrude into the aquifer (van der Meulen et al., 2004). At that time, water from the Rhine and Meuse rivers was piped into the dune where it was recharged and filtered for human use. The aquifer supplies enough water for 1.5 million people in surrounding cities. Because of the importance of this water source, the Meijendel dune is managed as a nature reserve that serves both drinking water and recreation needs. In 1999, the cost of management was €3.2 million ($3.8 million) yr^{-1} while the yearly income of the reserve was €84 million Euros ($99.2 million) yr^{-1}.

p0265 Sandy beaches and dunes provide a valuable extractive resource, and also support hunting and agricultural activities (**Table 8**). For example, the mining of sand has occurred for centuries for multiple uses, including extraction of minerals such as silica and feldspar for glass and ceramic production, infill for development, amendments for agriculture, and base material for construction products. Although sand is a valuable resource, its extraction through mining can have obvious negative and conflicting effects on the other services mentioned in this section, especially that of coastal protection and aquifers. In addition, beaches and dunes are habitat for fish, shellfish, birds, rodents, and ungulates, which have been captured or cultivated for food since humans first colonized the coast. Baeyens and Martînez (2004) described the use of animals by humans in European dunes in some detail. These systems have undergone a transition from subsistence hunting to grazing to protection as a response to overexploitation and overgrazing. For example, starting in the Middle Ages, rabbits, waterfowl, and deer were encouraged in dunes through predator control and the addition of food. As these species began to dominate, they caused declines in vegetation and other animal species. Now, especially in the European dunes, protection and restoration of dune wildlife and habitat has become a priority. In other regions of the world, dunes have been used for agricultural purposes (Pye and Tsoar, 1990). For centuries, a simple form of agriculture known as mawasi has been used to grow vegetables and fruits in the moist interdune depressions in the Gaza and Sinai.

p0270 Finally, beaches and dunes also supply important recreational benefits (**Table 8**). Boating, fishing, swimming, scuba diving, walking, beachcombing, and sunbathing are among the numerous recreational and scenic opportunities that are provided by beach and dune access. In the US alone, 70% of the population visits the beach on vacation and 85% of total tourism dollars come from beach visits (Houston, 2008). However, overuse of dune habitat due to beach recreation can cause significant damages. The impacts to beach and dune function have been mostly in the form of changes in sand stabilization and distribution. Trampling of native vegetation by pedestrians or vehicles can destabilize sand and result in the loss of foredunes and thus coastal protection. Therefore, as with all coastal systems, reducing the damages caused by overuse of certain services such as the recreation and tourism benefits provided by beaches and dunes requires thoughtful management and planning (e.g., Heslenfeld et al., 2004; Moreno-Casasola, 2004).

12.06.4 Nonlinearity of ECE Services and Values s0045

p0275 Many of the functions and services described above are likely to show nonlinear characteristics across different spatial or temporal scales (Barbier et al., 2008; Koch et al., 2009). Because the functional relationships inherent in many of these ecological processes are understudied, and there is so little corresponding economic information on the value of important services, estimations of how the value of an ecosystem service varies across an ecological landscape or time frame are rare. However, for a handful of key services, researchers have begun to explore how ecological functions vary spatially or temporally and thus influence the economic benefits that they provide (Peterson and Turner, 1994; Petersen et al., 2003; Rountree and Able, 2007; Aburto-Oropeza et al., 2008; Aguilar-Perera et al., 2008; Barbier et al., 2008; Meynecke et al., 2008; Koch et al., 2009). As the example below illustrates, this type of information can be critical to valuing a suite of services that can have conflicting uses. AU21,22 AU23

p0280 As described in this chapter, coastal interface systems, including nearshore coral reefs, seagrass beds, salt marshes, mangroves, and sand dunes, can provide protection against wind and wave damage caused by coastal storm and surge events, although the magnitude of protection will vary among these ECEs (**Tables 3–6** and **8**). However, for all these coastal habitats, nonlinear landscape relationships have been identified between habitat area and measurements of the ecosystem function of wave attenuation (Barbier et al., 2008; Koch et al., 2009). For example, salt marshes and mangroves show a pattern of decreased wave attenuation with increased distance from the shoreline. In the case of nearshore coral reefs and seagrasses, wave attenuation is a function of the water depth above the reef or grass bed, and these relationships are also nonlinear. Also, sand beaches and dunes attenuate waves in a nonlinear manner with exponentially increasing protection at higher elevations or where dune vegetation cover is greatest.

p0285 Moreover, coral reefs, seagrass beds, mangroves, and salt marshes strongly influence the abundance, growth, and structure of resident and neighboring marine fisheries by providing nursery, breeding, and other habitat functions for commercially important fish and invertebrate species that spend at least part of their life cycles in ECEs. Evidence of this coastal habitat-fishery linkage increasingly indicates that the value of this service is nonlinear because the quality of the habitat is greater at the seaward edge or 'fringe' of the coastal habitat than further inland (Peterson and Turner, 1994; Manson et al., 2005; Aburto-Oropeza et al., 2008; Aguilar-Perera et al., 2008). For example, Peterson and Turner (1994) found that densities of most fish and crustaceans were highest in salt marshes in Louisiana within 3 m of the water's edge compared to the interior marshes. In the Gulf of California, Mexico, the mangrove fringe with a width of 5–10 m has the most influence on the productivity of nearshore fisheries, with a median value of $37 500 ha^{-1}. Fishery landings also increased positively with the length of the mangrove fringe in a given location (Aburto-Oropeza et al., 2008).

p0290 To illustrate how nonlinearity can affect the value of an ecosystem service for an entire coastal landscape, we return to the Thailand mangrove example of Section 12.06.3.4. Recall that this study found that three ecosystem services – coastal protection, wood-product collection, and habitat support for offshore fisheries – have a combined value ranging from $10 158 to $12 392 ha^{-1} in net present value terms, and that the highest value of the mangrove is its storm protection service, which yields a net present value of $8966–$10821 ha^{-1} (see Table 7). But what if these values for mangroves were used to inform a land-use decision, in particular, that of converting an entire mangrove ecosystem to shrimp aquaculture? Deciding how much of a mangrove forest coastline to convert to shrimp aquaculture may depend critically on whether or not all the mangroves are equally beneficial in terms of coastal storm protection (Barbier et al., 2008). Suppose that it is assumed initially that the annual per hectare values for the various ecosystem benefits are uniform, and thus vary linearly across a 10 km^2 mangrove landscape. Following this assumption, a mangrove area of 10 km^2 would have an annual storm protection value of 1000 times the $1879 ha^{-1} point estimate, which yields an annual total benefit estimate of nearly $1.9 million. Barbier et al. (2008) employed this assumption to compare the net present value (10% discount rate and 20-year horizon) of shrimp farming to the three mangrove services – coastal protection, wood-product collection, and habitat support for offshore fisheries – for the entire mangrove area. Figure 1 shows the comparison of benefits, when they are all 'scaled' uniformly across the 10-km^2 landscape.

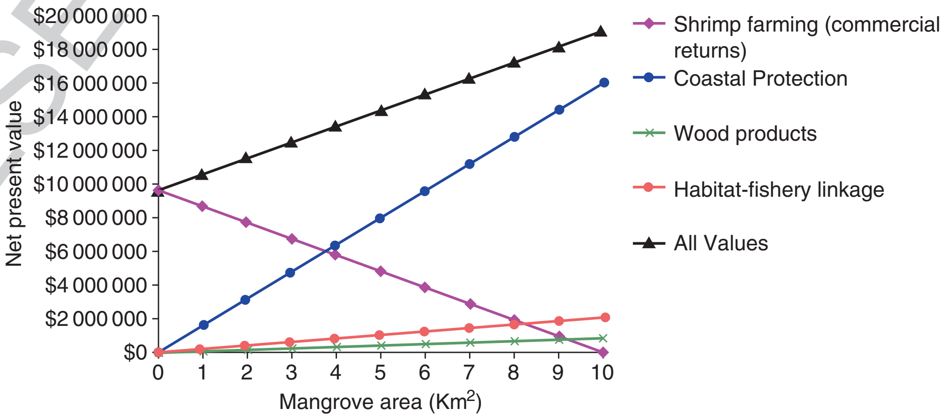
p0295 Figure 1 also aggregates all four values to test whether an integrated land-use option involving some conversion and some preservation yields the highest total value. When all values are linear, the outcome is a typical all-or-none scenario; either the aggregate values will favor complete conversion or they will favor preservation of the entire habitat. Because the ecosystem service values are large and increase linearly with mangrove area, the preservation option is preferred. The aggregate value of the mangrove system is at its highest ($18.98 million) when it is completely preserved and any conversion to shrimp farming would lead to less aggregate value compared to full preservation. Therefore, any land-use strategy that considers all the values of the ecosystem would favor mangrove preservation and no shrimp-farm conversion.

p0300 However, not all mangroves along a coastline are equally effective in storm protection and thus this value is unlikely to be uniform across all mangroves. The reason is that the storm protection service provided by mangroves depends on their ability to attenuate waves. Ecological and hydrological field studies suggest that mangroves are unlikely to stop storm waves that are greater than 6 m in height (Forbes and Broadhead, 2007; Wolanski, 2007; Alongi, 2008; Cochard et al., 2008). On the other hand, where mangroves are effective as natural barriers against storms that generate waves less than 6 m in height, the wave height of a storm decreases quadratically for each 100 m that a mangrove forest extends out to sea (Mazda et al., 1997; Barbier et al., 2008). In other words, wave attenuation is greatest for the first 100 m of mangroves but declines as more mangroves are added to the seaward edge.

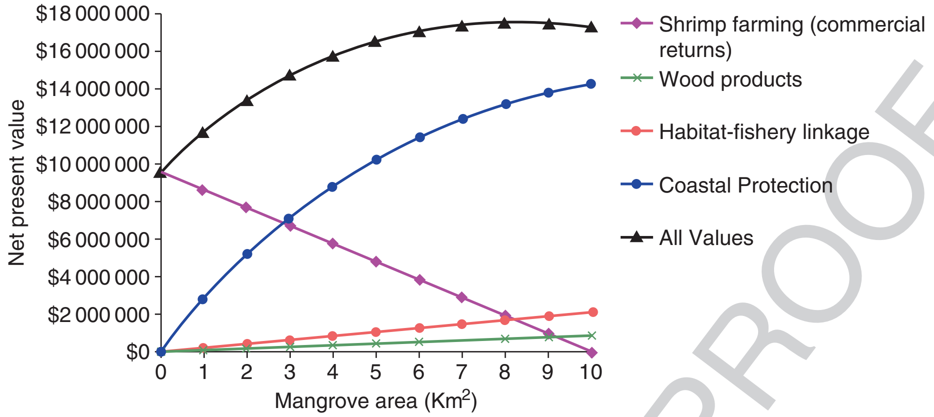
p0305 Barbier et al. (2008) employed the nonlinear wave attenuation function for mangroves based on the field study by Mazda et al. (1997) to revise the estimate of storm protection service value for the Thailand case study (Figure 2).

p0310 The storm-protection service of mangroves still dominates all values, but small losses in mangroves will not cause the economic benefits of storm buffering by mangroves to fall precipitously. The consequence is that the aggregate value across all uses of the mangroves, including shrimp farming and ecosystem values, is at its highest ($17.5 million) when up to 2 km^2 of mangroves are allowed to be converted to shrimp aquaculture and the remainder of the ecosystem is preserved. Thus, taking into account the nonlinear relationship AU24 in storm protection, it is possible to make a more economically feasible land-use decision than if a linear, or all-or-none, strategy was employed.

p0315 Given the potential generality of nonlinear ecosystem functions and services, it is clear that future coastal planning and management will need to incorporate this aspect into their decision-making process. Although these nonlinearities increase the complexity of management decisions, they may, as we have seen in the Thailand mangrove example, also help resolve conflicting ECE uses while providing the best economic value for stakeholders. For that reason, we envision this aspect of ecosystem services to become more important in the future.



AU30 f0005 **Figure 1** Comparison of shrimp farming to various mangrove services at coastal landscape level (10 km^2), Thailand (net present value, 10% discount rate, 1996$), assuming linear scaling of wave attenuation function. From Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J.H., Granek, E., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo G.M., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321–323.



f0010 **Figure 2** Comparison of shrimp farming to various mangrove services at coastal landscape level (10 km^2), Thailand (net present value, 10% discount rate, 1996$), incorporating nonlinear scaling of wave attenuation function. From Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J.H., Granek, E., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo G.M., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321–323.

s0050 ## 12.06.5 Synergistic Characteristics of ECE Services and Values

p0320 One unique feature of ECEs is that they occur at the interface between the coast, land, and watersheds, which also make them especially valuable. The location of ECE in the land-sea interface suggests a high degree of interconnectedness or connectivity across these systems, leading to the linked provision of one or multiple services by more than one ECE.

p0325 As Moberg and Rönnbäck (2003) described for tropical regions, numerous physical and biogeochemical interactions have been identified among mangroves, seagrass beds, and coral reefs that effectively create interconnected systems, or a single 'seascape'. By dissipating the force of currents and waves, coral reefs are instrumental for the evolution of lagoons and sheltered bays that are suitable environments for seagrass beds and mangroves. In turn, the control of sedimentation of nutrients and pollutants by mangroves and seagrasses creates the coastal water conditions that favor the AU25 growth of coral reefs. This synergistic relationship between coral reefs, seagrasses, mangroves, and even sand dunes suggests that the presence of these interlinked habitats in a seascape may considerably enhance the ecosystem service provided by one single habitat.

p0330 For example, Alongi (2008) suggested that the extent to which mangroves offer protection against catastrophic storm events, such as tsunamis, may depend not only on the relevant features and conditions within the mangrove ecosystem, such as width of forest, slope of forest floor, forest density, tree diameter and height, proportion of above-ground biomass in the roots, soil texture, and forest location (open coast vs. lagoon), but also on the presence of foreshore habitats, such as coral reefs, seagrass beds, and dunes. Similar cumulative effects of wave attenuation are noted for seascapes containing coral reefs, seagrasses, and marshes (Koch et al., 2009). As we have seen in Section 12.06.3, each ECE habitat has considerable ability to attenuate waves, and thus, the presence of foreshore habitats, such as coral reefs and seagrasses, can reduce significantly the wave energy reaching the seaward edge of mangroves, salt marshes, sand beaches, or dunes. For instance, evidence from the Seychelles documents how rising coral reef mortality and deterioration have increased significantly the wave energy reaching shores that are normally protected from erosion and storm surges by these reefs (Sheppard et al., 2005). In the Caribbean, mangroves appear not only to protect shorelines from coastal storms but may also to enhance the recovery of coral reef fish populations from disturbances due to hurricanes and other violent storms (Mumby and Hastings, 2008).

ECE systems are also linked biologically. Many fish and shellfish species utilize mangroves and seagrass beds as nursery grounds, and eventually migrate to coral reefs as adults, only to return to the mangroves and seagrasses to spawn (Nagelkerken et al., 2002; Mumby et al., 2004; Rountree and Able, 2007; Meynecke et al., 2008). In addition, the high biological productivity of mangroves, marshes, and seagrasses also produces significant amounts of organic matter that is used directly or indirectly by marine fishes, shrimps, crabs, and other species (Chong, 2007). The consequence is that interconnected seascapes contribute significantly to supporting fisheries via a number of ecosystem functions, including nursery and breeding habitat, trophic interactions, and predator-free habitat. p0335

For example, studies in the Caribbean show that the presence of mangroves and seagrasses enhances considerably the biomass of coral reef fish communities (Nagelkerken et al., 2002; Mumby et al., 2004; Mumby, 2006). In Malaysia, it is estimated that mangrove forests sustain more than half of the annual offshore fish landings, much of which are from reef fisheries (Chong, 2007). In Puerto Rico, maps show fish distributions to be controlled by the spatial arrangement of mangroves, seagrasses, and coral reefs, and the relative value of these habitats as nurseries (Aguilar-Perera and Appeldoorn, 2008). Stratification of environmental conditions along a marsh-habitat gradient, stretching from intertidal vegetated salt marshes, to subtidal marsh creeks, to marsh-bay fringe, and then to open water channels, indicates large spatial and temporal variability in fish migration, nursery habitats, and food webs (Rountree and Able, 2007). Finally, indices representing the connectivity of mangroves, salt marshes, and channels explained 30–70% of the catch-per-unit effort p0340 harvesting yields for commercially caught species in Queensland, Australia (Meynecke et al., 2008).

p0345 In sum, allowing for the connectivity of ECE habitats may have important implications for assessing the ecological functions underling key ecosystems services, such as coastal protection, control of erosion, and habitat-fishery linkages. Only recently have studies of ECEs begun to assess the cumulative implications for these services, or to model this connectivity (for one example, see Sanchirico and Mumby, 2009). As we shall discuss further in the next section, allowing for such synergistic characteristics of ECEs has important management implications.

s0055 12.06.6 Management Implications of Ecosystem Services and Values

p0350 Given the rate and scale at which ECEs are disappearing worldwide, assessing that the ecological services of these systems is critically important to their management. Our review of five ECEs, that is, near-shore coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes, reveals that many of the important benefits of these habitats are undervalued or even ignored in many coastal development decisions. This reinforces the importance of measuring the value of such ecological services, and employing these values appropriately in coastal management and planning. For example, the case study of converting mangroves to shrimp ponds in Thailand outlined in Section 12.06.4 illustrates how assessing the values of mangrove ecosystem services can help address two important land-use policy decisions concerning mangroves globally. First, to what extent should the remaining mangroves be conserved as opposed to conversion into shrimp aquaculture and other commercial coastal development? Second, do the ecosystem benefits of rehabilitating and restoring mangroves in deforested areas justify the costs of investing in restoration efforts? Both questions are also critical to the other ECEs and their ecosystem services, as all coastal interface habitats are facing increasing pressure for conversion to other economic activities, while at the same time, in many coastal areas where ECEs have been degraded or lost, there is often keen interest in restoring these habitats.

p0355 Maintaining ECEs for their multiple and synergistic ecosystem services invariably involves managing coastal landscapes across different spatial and temporal scales. As this review has indicated, for a number of key ecosystem services, such as coastal protection and habitat-fishery linkages, there is now sufficient evidence to suggest that these services are not uniform across a coastal seascape. Incorporating nonlinear and synergistic characteristics of ECEs into management scenarios is likely to result in the most ecologically and economically sustainable management plan possible (Granek et al., 2010). For example, in Section 12.06.4, we discussed how nonlinearity could affect the value of an ecosystem service for an entire coastal landscape by again using the example of Thailand mangroves that could be converted to shrimp farming. When it is assumed that the value of the key service – coastal protection against tropical storms – is uniform across the entire mangrove landscape, then the management decision is simple: either the aggregate values favor complete conversion or they favor preserving the entire landscape. In contrast, however, when coastal protection is assumed to vary nonlinearly across the landscape, for example, because wave attenuation is greatest for mangroves along the seaward edge but declines for mangroves further inward, then small losses in mangroves will not cause the economic benefits of storm buffering by mangroves to fall precipitously. It is then possible that a small part of the mangroves further inward could be converted to shrimp aquaculture, while the remainder of the mangrove would be preserved. Although this case study was a highly simplified example of how an ecological function, and thus the ecosystem service it supports, varies nonlinearly across a coastal landscape, it nonetheless shows how such nonlinear relationships can have important implications for management at the landscape scale for all ECEs.

Because the connectivity of ECE across land–sea gradients has implications for the provision of certain ecosystem services, management of the entire seascape will be necessary to preserve such synergistic effects. For example, Mumby (2006) argued that the life cycle migration of fish between mangroves, sea grass beds, and coral reefs has important implications for the management of ECE habitats in the Caribbean. He recommended that management planning should focus on connected corridors of these habitats and emphasized four key priorities: (1) the relative importance of mangrove nursery sites; (2) the connectivity of individual reefs to mangrove nurseries; (3) areas of nursery habitat that have an unusually large importance to specific reefs; and (4) priority sites for mangrove restoration projects. Similarly, Meynecke et al. (2008) emphasized that to improve marine-protected areas, it is important to understand the role of connectivity in the life history of fishes that likely utilize different ECEs. p0360

In sum, the more we learn about ECEs and their ecosystem services, it is apparent that ignoring these benefits is detrimental to coastal management and planning. In addition, more attention needs to be paid to how these services vary across seascapes, as these considerations clearly matter to managing estuarine, coastal, and inshore marine environments (Granek et al., 2010). Coasts and small islands may comprise just 4% of the Earth's total land area, but, as this review has shown, the ECEs that dominate these geographic areas provide some of the most important global benefits for humankind. p0365

References AU26

Aburto-Oropeza, O., Ezcurra, E., Danemann, G., Valdez, V., Murray, J., Sala, E., 2008. Mangroves in the Gulf of California increase fishery yields. Proceedings of the National Academy of Sciences of the United States of America 105, 10456–10459. bib0005

Aguilar-Perera, A., Appeldoorn, R.S., 2008. Spatial distribution of marine fishes along a cross-shelf gradient containing a continuum of mangrove–seagrass–coral reefs off southwestern Puerto Rico. Estuarine, Coastal and Shelf Science 76, 378–394. bib0010

Alongi, D.M., 2002. Present state and future of the world's mangrove forests. Environmental Conservation 29, 331–349. bib0015

Alongi, D.M., 2008. Mangrove forests: resilience, protection from tsunamis, and responses to global climate change. Estuarine, Coastal and Shelf Science 76, 1–13. bib0020

Badola, R., Hussain, S.A., 2005. Valuing ecosystems functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. Environmental Conservation 32, 85–92. bib0025

Baeyens, G., Martînez, M.L., 2004. Animal life on coastal dunes: from exploitation and prosecution to protection and monitoring. In: Martinez, M.L., Psuty, N.P. (Eds.), Coastal Dunes: Ecology and Conservation. Springer, Heidelberg, Germany, pp. 279–296. bib0030

Bandaranayake, W.M., 1998. Traditional and medicinal uses of mangroves. Mangroves and Salt Marsh 2, 133–148. bib0035

bib0040 Barbier, E.B., 1994. Valuing environmental functions: tropical wetlands. Land Economics 70, 155–173.
bib0045 Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. Economic Policy 22, 177–229.
bib0050 Barbier, E.B., Cox, M., 2003. Does economic development lead to mangrove loss? A cross-country analysis. Contemporary Economic Policy 21, 418–432.
bib0055 Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J.H., Granek, E., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo G.M., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321–323.
bib0060 Barbier, E.B., Strand, I., 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. Environmental and Resource Economics 12, 151–166.
bib0065 Beer, S., Koch, E.W., 1996. Photosynthesis of seagrasses vs. marine macroalgae in globally changing CO_2 environments. Marine Ecology Progress Series 141, 199–204.
bib0070 Bellwood, D.R., Hughes, T.P., Connolly, S.R., Tanner, J., 2005. Environmental and geometric constraints on Indo-Pacific coral reef biodiversity. Ecology Letters 8, 643–651.
bib0075 Bellwood, D.R., Hughes, T.P., Folke, C., Nystrom, M., 2004. Confronting the coral reef crisis. Nature 429, 827–833.
bib0080 Berkes, F., 1999. Sacred Ecology: Traditional Ecological Knowledge and Resource Management. Taylor and Francis, Philadelphia, PA, 209 pp.
bib0085 Bertness, M.D., 1984. Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. Ecology 66, 1042–1055.
bib0090 Bertness, M.D., 1991. Zonation of *Spartina* spp. in New England salt marshes. Ecology 72, 138–148.
AU27 bib0095 Bertness, M.D., 2006. Atlantic Shoreline Ecology: A Natural History. Princeton University Press.
bib0100 Boesch, D.F., Turner, R.E., 1984 Dependency of fishery species on salt marshes: the role of food and refuge. Estuaries 7, 460–468.
bib0105 Braatz, S., Fortuna, S., Broadhead, J., Leslie, R., 2007. Coastal Protection in the Aftermath of the Indian Ocean Tsunami. What Role for Forests and Trees? FAO, Bangkok, pp. 157–179.
bib0110 Brander, L.M., Florax, R.J.G.M., Vermaat, J.E., 2006. The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. Environmental and Resource Economics 33, 223–250.
bib0115 Brander, L.M., Van Beukering, P., Cesar, H.S.J., 2007. The recreational value of coral reefs: a meta-analysis. Ecological Economics 63, 209–218.
bib0120 Broomberg, K., Silliman, B.R., 2009. Patterns of salt marsh loss within coastal regions of North America: pre-settlement to present. In: Silliman, B.R., Grosholz, T., Bertness, M.D. (Eds.), Human Impacts on Salt Marshes: A Global Perspective. University of California Press.
bib0125 Bromberg-Gedan, K., Silliman, B.R., Bertness, M.D., 2009. Centuries of human-driven change in salt marsh ecosystems. Annual Reviews of Marine Science 1, 117–141.
bib0130 Brown, A.C., McLachlan, A., 2002. Sandy shore ecosystems and the threats facing them: some predictions for the year 2025. Environmental Conservation 29, 62–77.
bib0135 Buckley, R., 1987. The effect of sparse vegetation on the transport of dune sand by wind. Nature 325, 426–428.
bib0140 Carté, B.K., 1996. Biomedical potential of marine natural products. Bioscience 46, 271–286.
bib0145 Carter, R.W.G., 1990. Coastal Environments: An Introduction to the Physical, Ecological and Cultural Systems of Coastlines. Academic Press, London, 617 pp.
bib0150 Carter, R.W.G., 1991. Near-future sea level impacts on coastal dune landscapes. Landscape Ecology 6, 29–39.
bib0155 Cesar, H., Burke, L., Pet-Soede, L., 2003. The Economics of Worldwide Coral Reef Degradation. Cesar Environmental Economics Consulting, Arnhem, Netherlands.
bib0160 Cesar, H.S.J., van Beukering, P.J.H., 2004. Economic valuation of the coral reefs of Hawai'i. Pacific Science 58, 231–242.
bib0165 Chong, J., 2005. Protective Values of Mangrove and Coral Ecosystems: A Review of Methods and Evidence. IUCN, Gland, Switzerland.
bib0170 Chong, V.C., 2007. Mangrove-fishery linkages – the Malaysian perspective. Bulletin of Marine Science 80, 755–772.
bib0175 Cochard, R., Ranamukhaarachchi, S.L., Shivakotib, G.P., Shipin, O.V., Edwards, P.J., Seeland, K.T., 2008. The 2004 tsunami in Aceh and Southern Thailand: a review on coastal ecosystems, wave hazards and vulnerability. Perspectives in Plant Ecology, Evolution and Systematics 10, 3–40.
bib0180 Connell, J., Hughes, T., Wallace, C., 1997. A 30-year study of coral abundance, recruitment, and disturbance at several scales in space and time. Ecological Monographs 67, 461–488.
bib0185 Costanza, R., Pérez-Maqueo, O., Martinez, M.L., Sutton, P., Anderson, S.J., Mulder, K., 2008. The value of coastal wetlands for hurricane protection. AMBIO 37, 241–248.
bib0190 Cowles, H.C., 1899. The ecological relations of the vegetation on the sand dunes of Lake Michigan. Botanical Gazette 27, 167–202.
bib0195 Daily, G., 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC.
Davy, A., Figueroa, E., Bakker, J., 2009. Human modification European salt marshes. In: Silliman, B.R., Grosholz, T., Bertness, M.D. (Eds.), Human Impacts on Salt Marshes: A Global Perspective. University of California Press. bib0200
de la Torre-Castro, M., Rönnbäck, P., 2004. Links between humans and seagrasses – an example from tropical East Africa. Ocean and Coastal Management 47, 361–387. bib0205
Duarte, C.M., 2002. The future of seagrass meadows. Environmental Conservation 29, 192–206. bib0210
Duarte, C.M., Dennison, W.C., Orth, R.J., Carruthers, T.J.B., 2008. The charisma of coastal ecosystems: addressing the imbalance. Estuaries and Coasts 31, 233–238. bib0215
Duarte, C.M., Middleburg, J.J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2, 1–8. bib0220
Dulvy, N.K., Stanwellsmith, D., Darwall, W.R.T., Horrill, C.J., 1995. Coral mining at Mafia-island, Tanzania – a management dilemma. AMBIO 24, 358–365. bib0225
Ellison, AM., Farnsworth, E.J., 2001. Mangrove communities. In: Bertness, M.D., Gaines, S.D., Hay, M.E. (Eds.), Marine Community Ecology. Sinauer, Sunderland, MA, pp. 423–442. bib0230
FAO (Food and Agricultural Organization) of the United Nations. 2007. The world's mangroves 1980–2005. FAO Forestry Paper 153. FAO, Rome. bib0235
Felger, R.S., Moser, M.B., Moser, E.W., 1980. Seagrasses in Seri Indian culture. In: Phillips, R.C., McRoy, C.P. (Eds.), Handbook of Seagrass Biology: An Ecosystem Perspective. Garland STPM Press, New York, NY, pp. 260–276. bib0240
Fernando, H.J.S., McCulley, J.L., Mendis, S.G., Perera, K., 2005. Coral poaching worsens tsunami destruction in Sri Lanka. EOS, Transactions American Geophysical Union 86 (33), 301. bib0245
Fonseca, M.S., Cahalan, J.A., 1992. A preliminary evaluation of wave attenuation for four species of seagrass. Estuarine, Coastal and Shelf Science 35, 565–576. bib0250
Forbes, K., Broadhead, J., 2007. The Role Of Coastal Forests in the Mitigation of Tsunami Impacts. RAP Publication 2007/1, Food and Agricultural Organization of the United Nations, Regional Office for Asia and the Pacific, Bangkok. bib0255
Gardner, T.A., Cote, I.M., Gill, J.A., Grant, A., Watkinson, A.R., 2003. Long-term region-wide declines in Caribbean corals. Science 301, 958–960. bib0260
Gattuso, J.P., Frankignoulle, M., Wollast, R., 1998. Carbon and carbonate metabolism in coastal aquatic ecosystems. Annual Review of Ecology and Systematics 29, 405–434. bib0265
Glynn, P.W., 1976. Some physical and biological determinants of coral community structure in the eastern Pacific. Ecological Monographs 46, 431–456. bib0270
Glynn, P.W., 1997. Bioerosion and coral-reef growth: a dynamic balance. In: Birkeland, C. (Ed.), Life and Death of Coral Reefs. Chapman and Hall, New York, NY, pp. 68–95. bib0275
Graham, N.A.J., Wilson, S.K., Jennings, S., Polunin, N.V.C., Robinson, J., Bijoux, J.P., Daw, T.M., 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. Conservation Biology 21, 1291–1300. bib0280
Granek, E.F., Polasky, S., Kappel, C.V., Reed, D.J., Stoms, D.M., Koch, E.W., Kennedy, C.J., Cramer, L.A., Hacker, S.D., Barbier, E.B., Aswani, S., Ruckelshaus, M., Perillo, G.M., Silliman, B.R., Muthiga, N., Bael, D., Wolanski, E., 2010. Ecosystem services as a common language for coastal ecosystem-based management. Conservation Biology 24 (1), 207–216. bib0285
Greenstein, B.J., Pandolfi, J.M., (2008). Escaping the heat: range shifts of reef coral taxa in coastal Western Australia. Global Change Biology 14, 513–528. bib0290
Hacker, S.D., Bertness, M.D., 1995. Morphological and physiological consequences of a positive plant interaction. Ecology 76, 2165–2175. bib0295
Hallock, P., 1997. Reefs and reef limestones in earth history. In: Birkeland, C. (Ed.), Life and Death of Coral Reefs. Chapman and Hall, New York, NY, pp. 13–42. bib0300
Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R.S., Watson, R., 2008. A global map of human impacts on marine ecosystems. Science 319, 948–952. bib0305
Hardisty, J. 1994. Beach and nearshore sediment transport. In: Pye, K. (Ed.), Sediment Transport and Depositional Processes. Blackwell, London, pp. 216–255. bib0310
Heal, G.M., Barbier, E.B., Boyle, K.J., Covich, A.P., Gloss, S.P., Hershner, C.H., Hoehn, J.P., Pringle, C.M., Polasky, S., Segerson, K., Shrader-Frechette, K., 2005. Valuing Ecosystem Services: Toward Better Environmental Decision Making. The National Academies Press , Washington, DC. bib0315
Hemminga, M.A., Duarte, C.M., 2000. Seagrass Ecology. Cambridge University Press, Cambridge, 298 pp. bib0320
Hemminga, M.A., Nieuwenhuize, J., 1990. Seagrass wrack-induced dune formation on a tropical coast (Banc d'Arguin, Mauritania). Estuarine, Coastal and Shelf Science 31, 499–502. bib0325
Henry, H., Jefferies, R., 2009 Opportunistic herbivores, migratory connectivity, and catastrophic shifts in Arctic coastal ecosystems. In: Silliman, B.R., Grosholz, T., Bertness, M.D. (Eds.),. Human Impacts on Salt Marshes: A Global Perspective. University of California Press. bib0330 AU28
Heslenfeld, P., Jungerius, P.D., Klijn, J.A., 2004. European coastal dunes: ecological values, threats, opportunities, and policy development. In: Martinez, M.L., Psuty, N.P. (Eds.), Coastal Dunes: Ecology and Conservation. Springer, Heidelberg, Germany, pp. 335–351. bib0335

bib0340 Hesp, P.A., 1989. A review of biological and geomorphological processes involved in the initiation and development of incipient foredunes. Proceedings of the Royal Society of Edinburgh 96B, 181–201.

bib0345 Hesp, P.A. 2000. Coastal Sand Dunes. Form and Function. CDVN Technical Manual 4. Massey University, Rotorua Printers, NZ, 28 pp.

bib0350 Hesp, P.A., Short, A.D., 1999. Barrier morphodynamics. In: Short, A.D. (Ed.), Handbook of Beach and Shoreface Morphodynamics. Wiley, Chichester, pp. 307–333.

bib0355 Houston, J.R., 2008. The economic value of beaches – a 2008 update. Shore and Beach 76, 22–26.

bib0360 Janssen, R., Padilla, J.E., 1999. Preservation or conservation? Valuation and evaluation of a mangrove forest in the Philippines. Environmental and Resource Economics 14, 297–331.

bib0365 Hoegh-Guldberg, O., 1999. Climate change, coral bleaching and the future of the world's coral reefs. Marine and Freshwater Research 50, 839–866.

bib0370 Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E., Harvell, C.D., Sale, P.F., Edwards, A.J., Caldeira, K., Knowlton, N., Eakin, C.M., Iglesias-Prieto, R., Muthiga, N., Bradbury, R.H., Dubi, A., Hatziolos, M.E., 2007. Coral reefs under rapid climate change and ocean acidification. Science 318, 1737–1742.

bib0375 Jain, R., 2009. Marine life: New hope for cancer drugs. Indian Journal of Cancer 46, 243–244.

bib0380 Koch, E.W., Ackerman, J., van Keulen, M., Verduin, J., 2006. Fluid dynamics in seagrass ecology: from molecules to ecosystems. In: Larkum, A.W.D., Orth, R.J., Duarte, C.M. (Eds.), Seagrasses: Biology, Ecology and Conservation. Springer, Heidelberg, Germany, pp. 193–225.

bib0385 Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M.E., Hacker, S.D., Granek, E.F., Primavera, J.H., Muthiga, N., Polasky, S., Halpern, B.S., Kennedy, C.J., Kappel, C.V., Wolanski, E., 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Frontiers in Ecology and the Environment 7, 29–37.

bib0390 Kuriyama, Y., Mochizuki, N., Nakashima, T., 2005. Influence of vegetation on Aeolian sand transport rate from a backshore to a foredune at Hasaki, Japan. Sedimentology 52, 1123–1132.

bib0395 Larson, M., Erickson, L., Hanson, H., 2004. An analytical model to predict dune erosion due to wave impact. Coastal Engineering 51, 675–696.

bib0400 Leatherman, S.P., 1979. Beach and dune interactions during storm conditions. Quarterly Journal of Engineering Geology and Hydrogeology 12, 281–290.

bib0405 Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809.

bib0410 Loya, S., Tal, R., Hizi, A., Issacs, S., Kashman, Y., Loya, Y., 1993. Hexaprenoid hydroquinones, novel inhibitors of the reverse-transcriptase of human-immunodeficiency-virus type-1. Journal of Natural Products 56, 2120–2125.

bib0415 Lubbock, H.R., Polunin, N.V.C., 1975. Conservation and the tropical marine aquarium trade. Environmental Conservation 2, 229–232.

bib0420 Lui, P.L.-F., Lynett, P., Fernando, H., Jaffe, B.E., Fritz, F., Higman, B., Morton, R., Goff J., Synolakis, C., 2005. Observations by the international tsunami survey team in Sri Lanka. Science 308, 1595.

bib0425 MacKenzie, R.A., Dionne, M., 2008. Habitat heterogeneity: importance of salt marsh pools and high marsh surfaces to fish production in two Gulf of Maine salt marshes. Marine Ecology Progress Series 368, 217–230.

bib0430 Manson, F.J., Loneragan, N.R., Skilleter, G.A., Phinn, S.R., 2005. An evaluation of the evidence for linkages between mangroves and fisheries: a synthesis of the literature and identification of research directions. In: Gibson, R.N., Atkinson, R.J.A., Gordon, J.D.M. (Eds.), Oceanography and Marine Biology: An Annual Review. CRC Press, Boca Raton, FL, vol. 43, pp. 483–513.

bib0435 Mathews, E., 1983. Global vegetation and land-use: new high-resolution databases for climate studies. Journal of Climatology and Applied Meteorology 22, 474–487.

bib0440 Mayor, J., Hicks, C., 2009. Potential impacts of elevated CO_2 on plant interactions, sustained growth, and carbon cycling in salt marsh ecosystems. In: Silliman, B.R., Grosholz, T., Bertness, M.D. (Eds.), Human Impacts on Salt Marshes: A Global Perspective. University of California Press.

bib0445 Mazda, Y., Magi, M., Kogo, M., Hong, P.N., 1997. Mangroves as a coastal protection from waves in the Tong King Delta, Vietnam. Mangroves and Salt Marshes 1, 127–135.

bib0450 McArthur, L.C., Boland, J.W., 2006. The economic contribution of seagrass to secondary production in South Australia. Ecological Modeling 196, 163–172.

bib0455 Meynecke, J.-O., Lee, S.Y., Duke, N.C., 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. Biological Conservation 141, 981–996.

AU29 bib0460 Millennium Ecosystem Assessment, 2005. Coastal systems. In: Ecosystems and Human Well-Being: Current State and Trends. Island Press, Washington, DC.chapter 19.

bib0465 Mitsch, W.J., Gosselink, J.G., 2008. Wetlands. Van Nostrand Reinhold, New York, NY.

bib0470 Moberg, F., Folke, C., 1999. Ecological goods and services of coral reef ecosystems. Ecological Economics 29, 215–233.

Moberg, F., Rönnbäck, P., 2003. Ecosystem services of the tropical seascape: interactions, substitutions and restoration. Ocean and Coastal Management 46, 27–46. bib0475

Moreno-Casasola, P., 2004. A case study of conservation and management of tropical sand dune systems: La Mancha-El Llano. In: Martinez, M.L., Psuty, N.P. (Eds.), Coastal Dunes: Ecology and Conservation. Springer, Heidelberg, Germany, pp. 319–333. bib0480

Morgan, P.A., Burdick, D.M., Short, F.T., 2009. The functions and values of fringing slat marshes in Northern New England, USA. Estuaries and Coasts 32, 483–495. bib0485

Morton, R.A., Gelfenbaum, G., Jaffe, B.E., 2007. Physical criteria for distinguishing sandy tsunami and storm deposits using modern examples, Sedimentary Geology doi:10.1016/j.sedgeo.2007.01.003. bib0490

Mumby, P.J., 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. Biological Conservation 128, 215–222. bib0495

Mumby, P.J., Dahlgren, C.P., Harborne, A.R., Kappel, C.V., Micheli, F., Brumbaugh, D.R., Holmes, K.E., Mendes, J.M., Broad, K., Sanchirico, J.N., Buch, K., Box, S., Stoffle, R.W., Gill, A.B., 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311, 98–101. bib0500

Mumby, P.J., Edwards, A.J., Arias-Gonzalez, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H., Wabnitz, C.C.C., Llewellyn, G., 2004. Mangroves enhance the biomass of reef fisheries in the Caribbean. Nature 427, 533–536. bib0505

Mumby, P.J., Harborne, A.R., Williams, J., Kappel, C.V., Brumbaugh, D.R., Micheli, F., Holmes, K.E., Dahlgren, C.P., Paris, C.B., Blackwell, P.G., 2007. Trophic cascade facilitates coral recruitment in a marine reserve. Proceedings of the National Academy of Sciences of the United States of America 104, 8362–8367. bib0510

Mumby, P.J., Hastings, A., 2008. The impact of ecosystem connectivity on coral reef resilience. Journal of Applied Ecology 45, 854–862 bib0515

Nagelkerken, I., Roberts, C.M., van der Velde, G., Dorenbosch, M., van Riel, M.C., Cocheret de la Morinière, E., Nienhuis, P.H., 2002. How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. Marine Ecology Progress Series 244, 299–305. bib0520

Naylor, R., Drew, M., 1998. Valuing mangrove resources in Kosrae, Micronesia. Environment and Development Economics 3, 471–490. bib0525

Odum, E.P., Smalley, A.E., 1959. Comparison of population energy flow of a herbivorous and a deposit-feeding invertebrate in a salt marsh ecosystem. Proceedings of the National Academy of Sciences of the United State of America 45, 617–622. bib0530

Olson, J.S., 1958. Lake Michigan dune development. 1. Wind–velocity profiles. Journal of Geology 66, 254–263. bib0535

Orth, R.J., Carruthers, T.J.B, Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L., Jr., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M., Williams, S.L., 2006. A global crisis for seagrass ecosystems. BioScience 56, 987–996. bib0540

Overton, M.F., Fisher, J.S., Stone, A.L., 1990. Large scale laboratory tests of dune erosion. Proceedings of the 22nd Coastal Engineering Conference, ASCE 1990. ASCE pp. 2471–2479. bib0545

Pandolfi, J.M., 2002. Coral community dynamics at multiple scales. Coral Reefs 21, 12–23. bib0550

Parsons, J.R., 1968. The archeological significance of mahameaes cultivation on the coast of Peru. American Antiquity 33, 80–85. bib0555

Paulay, G., Meyer, C., 2002. Diversification in the tropical Pacific: comparisons between marine and terrestrial systems and the importance of founder speciation. Integrative and Comparative Biology 42, 922–934. bib0560

Petersen, J.E., Kemp, W.M., Bartleson, R., Boynton, W.R., Chen, C.-C., Cornwell, J.C., Gardner, R.H., Hinkle, D.C., Houde, E.D., Malone, T.C., Mowitt, W.R., Murray, L., Sanford, L.P., Stevenson, J.C., Sundberg, K.L., Suttles, S.E., 2008. Multiscale experiments in coastal ecology: improving realism and advancing theory. BioScience 53, 1181–1197. bib0565

Pet-Soede, C., Cesar, H.S.J., Pet, J.S., 1999. An economic analysis of blast fishing on Indonesian coral reefs. Environmental Conservation 26, 83–93. bib0570

Peterson, C.H., Lubchenco, J., 1997. Marine ecosystem services. In: Daily, G.C. (Ed.), Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC, pp. 177–194. bib0575

Peterson, G.W., Turner, R.E., 1994. The value of salt marsh edge versus interior as habitat for fish and decapods crustaceans in a Louisiana tidal marsh. Estuaries 17, 235–262. bib0580

Porter, J.W., 1972. Predation by Acanthaster and its effect on coral species diversity. American Naturalist 106, 487-492. bib0585

Pratchett, M.S., Munday, M.S., Wilson, S.K., Graham, N.A.J., Cinner, J.E., Bellwood, D.R., Jones, G.P., Polunin, N.V.C., McClanahan, T.R., 2008. Effects of climate-induced coral bleaching on coral-reef fishes: ecological and economica consequences. Oceanography and Marine Biology 46, 251–296. bib0590

Psuty, N.P., 1986. A dune/beach interaction model and dune management. Thalassas 4, 11–15. bib0595

Pye, K., Tsoar, H., 1990. Aeolian Sand and Sand Dunes. Unwin Hyman, London, 396 pp. bib0600

bib0605 Rönnbäck, P., 1999. The ecological basis for economic value of seafood production supported by mangrove ecosystems. Ecological Economics 29, 235–252.
bib0610 Ruggiero, P., 2008. Impacts of climate change on coastal erosion and flood probability in the US Pacific Northwest. Proceedings of Solutions to Coastal Disasters 2008. Oahu Pearl City, HI, USA.
bib0615 Rönnbäck, P., Crona, B., Ingwall, L., 2007. The return of ecosystem goods and services in replanted mangrove forests – perspectives from local communities in Gazi Bay, Kenya. Environmental Conservation 34, 313–324.
bib0620 Ruggiero, P., Kaminsky, G.M., Gelfenbaum, G., Voigt, B., 2005. Seasonal to interannual morphodynamics along a high-energy dissipative littoral cell. Journal of Coastal Research 21, 553–578.
bib0625 Ruggiero, P., Komar, P.D., McDougal, W.G., Marra, J.J., Beach, R.A., 2001. Wave runup, extreme water levels, and the erosion of properties backing beaches. Journal of Coastal Research 17, 407–419.
bib0630 Ruitenbeek, H.J., 1994. Modeling economy–ecology linkages in mangroves: economic evidence for promoting conservation in Bintuni Bay, Indonesia. Ecological Economics 10, 233–247.
bib0635 Rybicki, N.B., 1997. Observations of tidal flux between submersed aquatic plant stand and the adjacent channel in the Potomac River near Washington, DC. Limnology and Oceanography 42, 307–317.
bib0640 Sadovy, Y.J., Vincent, A.C.J., Sale, P.F., 2002. Ecological issues and the trades in live reef fishes. In: Sale, P.F. (ed.), Coral Reef Fishes: Dynamics and Diversity in a Complex Ecosystem. Academic Press, San Diego, CA, pp. 391–420.
bib0645 Sallenger, A.H., Stockdon, H.F., Fauver, L., Hansen, M., Thompson, D., Wright, C.W., Lillycrop, J., 2006. Hurricanes 2004: an overview of their characteristics and coastal change. Estuaries and Coasts 29, 880–888.
bib0650 Sanchirico, J.N., Mumby, P., 2009. Mapping ecosystem functions to the valuation of ecosystem services: implications of species-habitat associations for coastal land-use decisions. Theoretical Ecology 2, 67–77.
bib0655 Sarntisart, I., Sathirathai, S., 2004. Mangrove dependency, income distribution and conservation. In: Barbier, E.B., Sathirathai, S. (Eds.), Shrimp Farming and Mangrove Loss in Thailand. Edward Elgar, London, pp. 96–114.
bib0660 Sathirathai, S., Barbier, E.B., 2001. Valuing mangrove conservation, Southern Thailand. Contemporary Economic Policy 19, 109–122.
bib0665 Sheppard, C., Dixon, D.J., Gourlay, M., Sheppard, A., Payet, R., 2005. Coral mortality increases wave energy reaching shores protected by reef flats: examples from the Seychelles. Estuarine, Coastal and Shelf Science 64, 223–234.
bib0670 Short, A.D., 1999. Wave-dominated beaches. In: Short, A.D. (Ed.), Handbook of Beach and Shoreface Morphodynamics. Wiley, Chichester, pp.173–203.
bib0675 Short, A.D., Hesp, P.A., 1982. Wave, beach and dune interactions in southeast Australia. Marine Geology 48, 259–284.
bib0680 Silliman, B.R., Grosholz, T., Bertness, M.D., 2009. Salt marshes under global siege. In: Silliman, B.R., Grosholz, T., Bertness, M.D. (Eds.), Human Impacts on Salt Marshes: A Global Perspective. University of California Press.
bib0685 Silliman, B.R., Bertness, M.D., 2002. A trophic cascade regulates salt marsh primary production. Proceedings of the National Academy of Sciences of the United States of America 99, 10500–10505.
bib0690 Silliman, B.R., van de Koppel, J., Bertness, M.D., Stanton, L., Mendelsohn, I., 2005 Drought, snails, and large-scale die-off of southern U.S. salt marshes. Science 310, 1803–1806.
bib0695 Smith, S.V. 1978. Coral-reef area and contributions of reefs to processes and resources of world's oceans. Nature 273, 225–226.
bib0700 Spalding, M.D., Blasco, F., Field, C.D., 1997. World Mangrove Atlas. International Society for Mangrove Ecosystems, Okinawa, Japan, 178 pp.
bib0705 Spalding, M.D., Taylor, M., Ravilious, C., Short, F., Green, E., 2003. The distribution and status of seagrasses. In: Green, E.P., Short, F.T. (Eds.), World Atlas of Seagrasses. University of California Press, Berkeley, LA, pp. 5–26.
Stockdon, H.F., Holman, R.A., Howd, P.A., Sallenger, A.H., 2006. Empirical parameterization of setup, swash, and runup. Coastal Engineering 53, 573–588. bib0710
Stockdon, H.F., Sallenger, H.A., Holman, R.A., Howd, P.A., 2007. A simple model for the spatially-variable coastal response to hurricanes. Marine Geology 238, 1–20. bib0715
Stockdon, H.F., Thompson, D.M., 2007. Vulnerability of National Park Service beaches to inundation during a direct hurricane landfall: Fire Island National Seashore. *USGS Open-File Report 2007–1389*. USGS, St. Petersburg, FL, 8 pp. bib0720
UNEP (United Nations Environment Programme), 2006. Marine and Coastal Ecosystems and Human Wellbeing: A Synthesis Report Based on the Findings of the Millennium Ecosystem Assessment. UNEP, Nairobi, 64 pp. bib0725
Valiela, I., Bowen, J.L., York, J.K., 2001. Mangrove forests: one of the world's threatened major tropical environments. BioScience 51, 807–815. bib0730
Van Der Meulen, F., Bakker, T.W.M., Houston, J.A., 2004. The costs of our coasts: examples of dynamic dune management from Western Europe. In: Martinez, M.L., Psuty, N.P. (Eds.), Coastal Dunes: Ecology and Conservation. Springer, Heidelberg, Germany, pp. 259–277. bib0735
van Beukering, P.J.H., Cesar, H.S.J., 2004. Ecological economic modeling of coral reefs: evaluating tourist overuse at Hanauma Bay and algae blooms at the Kihei coast, Hawai'i. Pacific Science 58, 243–260. bib0740
Walters, B.B., Rönnbäck, P., Kovacs, J.M., Crona, B., Hussain, S.A., Badola, R., Primavera, J.H., Barbier, E.B., Dahdouh-Guebas, F., 2008. Ethnobiology, socio-economics and management of mangrove forests: a review. Aquatic Botany 89, 220–236. bib0745
Walton, M.E.M, Samonte-Tan, G.P.B., Primavera, J.H., Edwards-Jones, G., Le Vay, L., 2006. Are mangroves worth replanting? The direct economic benefits of a community-based reforestation project. Environmental Conservation 33, 335–343. bib0750
Warren-Rhodes, K., Sadovy, Y., Cesar, H., 2004. Marine ecosystem appropriation in the Indo-Pacific: a case study of the live reef fish food trade. AMBIO 33, 116–119. bib0755
Watson, R.A., Coles, R.G., Long, W.L.L., 1993. Simulation estimates of annual yield and landed value for commercial Panaeid prawns from a tropical seagrass habitat, Northern Queensland, Australia. Australian Journal of Marine and Freshwater Research 44, 211–219. bib0760
Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Jr., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences 106, 12377–12381. bib0765
Wilkinson, C., Linden, O., Cesar, H., Hodgson, G., Rubens, J., Strong, A.E., 1999. Ecological and socioeconomic impacts of 1998 coral mortality in the Indian Ocean: An ENSO impact and a warning of future change? AMBIO 28, 188–196. bib0770
Wolanski, E., 2007. Estuarine Ecohydrology. Elsevier, Amsterdam, 157 pp. bib0775
Wood, E.M., 1985. Exploitation of Coral Reef Fishes for the Aquarium Fish Trade. Marine Conservation Society, Rosson-Wye. bib0780
Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314, 787–790. bib0785
Wyllie-Echeverria, S., Cox, P.A., 1999. The seagrass (*Zostera marina*) industry of Nova Scotia (1907–1960). Economic Botany 53, 419–426. bib0790
Yeo, B. 2002. Valuing a marine park in Malaysia. In: Pearce, D., Pearce, C., Palmer, C. (Eds.), Valuing the Environment in Developing Countries: Case Studies. Edward Elgar Publishing, Cheltenham, pp. 311–326. bib0795
Zimmerman, R.J., Minello, T.J., Rozas, L.P., 2000. Salt marsh linkages to productivity of penaeid shrimps and blue crabs in the Northern Gulf of Mexico. In: Weinstein, M.P., Kreeger, D.A. (Eds.), Concepts and Controversies in Tidal Marsh Ecology. Kluwer, Dordrecht. bib0800