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The value of estuarine and coastal ecosystem services

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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 services across a variety of ECEs, including marshes, mangroves, nearshore coral reefs, seagrass beds, and sand beaches and dunes. Where possible, we indicate estimates of the key economic values arising from these services, and discuss how the natural variability of ECEs impacts their benefits, the synergistic relationships of ECEs across seascapes, and management implications. Although reliable valuation estimates are beginning to emerge for the key services of some ECEs, such as coral reefs, salt marshes, and mangroves, many of the important benefits of seagrass beds and sand dunes and beaches have not been assessed properly. Even for coral reefs, marshes, and mangroves, important ecological services have yet to be valued reliably, such as cross-ecosystem nutrient transfer (coral reefs), erosion control (marshes), and pollution control (mangroves). An important issue for valuing certain ECE services, such as coastal protection and habitat–fishery linkages, is that the ecological functions underlying these services vary spatially and temporally. Allowing for the connectivity between ECE habitats also may have important implications for assessing the ecological functions underlying key ecosystems services, such coastal protection, control of erosion, and habitat–fishery linkages. Finally, we conclude by suggesting an action plan for protecting and/or enhancing the immediate and longer-term values of ECE services. Because the connectivity of ECEs across land–sea gradients also influences the provision of certain ecosystem services, management of the entire seascape will be necessary to preserve such synergistic effects. Other key elements of an action plan include further ecological and economic collaborative research on valuing ECE services, improving institutional and legal frameworks for management, controlling and regulating destructive economic activities, and developing ecological restoration options.

Key words: coral reef; economic value; ecosystem service; estuarine and coastal ecosystem; mangrove; salt marsh; sand beach and dune; seagrass; seascape.

Introduction

Estuarine and coastal ecosystems (ECEs) are some of the most heavily used and threatened natural systems globally (Lotze et al. 2006, Worm et al. 2006, Halpern et al. 2008). Their deterioration due to human activities is intense and increasing; 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, MEA 2005, Orth et al. 2006, UNEP 2006, FAO 2007, Waycott et al. 2009). This global decrease in ECEs is known to affect at least three critical ecosystem services (Worm et al. 2006): the number of viable (non-collapsed) 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). The loss of biodiversity, ecosystem functions, and coastal vegetation in ECEs may have contributed to biological invasions, declining water quality, and decreased coastal protection from flooding and storm events (Braatz et al. 2007, Cochard et al. 2008, Koch et al. 2009).

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Such widespread and rapid transformation of ECEs and their services suggest that it is important to understand what is at stake in terms of critical benefits and values. The purpose of this paper is to provide an overview of the main ecological services across a variety of ECEs, including marshes, mangroves, nearshore coral reefs, seagrass beds, and sand beaches and dunes. Where available, 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 in space and time results in nonlinear functions and services that greatly influence their economic value (Barbier et al. 2008, Koch et al. 2009) and some of the synergistic properties of ECEs. Because they exist 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 finish by highlighting the main management implications of this review of ECE services and their benefits, and provide an "action plan" to protect and/or enhance their immediate and longer term value to humankind.

Methods: Assessing ECE Services and Values

In identifying the ecosystem services provided by natural environments, a common practice is to adopt the broad definition of the Millennium Ecosystem Assessment (MEA 2005) that "ecosystem services are the benefits people obtain from ecosystems." Thus, the term "ecosystem services" is usually interpreted to imply the contribution of nature to a variety of "goods and services," which in economics would normally be classified under three different categories (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 and habitat functions, such as water purification, climate regulation, erosion control, and habitat provision), and (3) cultural benefits (e.g., spiritual and religious beliefs, heritage values).

However, for economists, the term "benefit" has a specific meaning. Mendelsohn and Olmstead (2009:326) summarize the standard definition as follows: "The economic benefit provided by an environmental good or service is the sum of what all members of society would be willing to pay for it." Thus, given this specific meaning, some economists argue that it is misleading to characterize all ecosystem services as "benefits." As explained by Boyd and Banzhaf (2007:619), "as end-products of nature, final ecosystem services are not benefits nor are they necessarily the final product consumed. For example, recreation is often called an ecosystem service. It is more appropriately considered a benefit produced using both ecological services and conventional goods and services." To illustrate this point, they consider recreational angling. It requires certain "ecosystem services," such as "surface waters and fish populations," but also "other goods and services including tackle, boats, time allocation, and access" (Boyd and Banzhaf 2007:619). But other economists still prefer a broader interpretation of ecosystem services, along the lines of the Millennium Ecosystem Assessment (MEA 2005), which equates ecosystem services with benefits. For example, Polasky and Segerson (2009:412) state: "We adopt a broad definition of the term ecosystem services that includes both intermediate and final services," which they justify by explaining that "supporting services, in economic terms, are akin to the infrastructure that provides the necessary conditions under which inputs can be usefully combined to provide intermediate and final goods and services of value to society." Thus, unlike Boyd and Banzhaf (2007), Polasky and Segerson (2009) consider recreation to be an ecosystem service.

Economists do agree that, in order to determine society's willingness to pay for the benefits provided by ecosystem goods and services, one needs to measure and account for their various impacts on human welfare. Or, as Freeman (2003:7) succinctly puts it: "The economic value of resource–environmental systems resides in the contributions that the ecosystem functions and services make to human well-being," and consequently, "the basis for deriving measures of the economic value of changes in resource–environmental systems is the effects of the changes on human welfare." Similarly, Bockstael et al. (2000:1385) state: "In economics, valuation concepts relate to human welfare. So the economic value of an ecosystem function or service relates only to the contribution it makes to human welfare, where human welfare is measured in terms of each individual's own assessment of his or her well-being." The key is determining how changes in ecosystem goods and services affect an individual's well-being, and then determining how much the individual is either willing to pay for changes that have a positive welfare impact, or conversely, how much the individual is willing to accept as compensation to avoid a negative effect.

In our approach to identifying the key services of estuarine and coastal ecosystem (ECEs) and their values, we adopt this consensus economic view. That is, as long as nature makes a contribution to human well-being, either entirely on its own or through joint use with other human inputs, then we can designate this contribution as an "ecosystem service." In other words, "ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations" (U.S. EPA 2009:12). In adopting this interpretation, (U.S. EPA 2009:12–13) "uses the term ecosystem service to refer broadly to both intermediate and final end services," and as a result, the report maintains that "in specific valuation contexts...it is important to identify whether the service being valued is an intermediate or a final service."

For example, following this approach, the tourism and recreation benefits that arise through interacting with an ECE can be considered the product of a "service" provided by that ecosystem. But it should be kept in mind, as pointed out by Boyd and Banzhaf (2007:619), that the role of the ECE is really to provide an "intermediate service" (along with "conventional goods and services") in the production of the final benefit of recreation and tourism. In selecting estimates of the "value" of this "intermediate" ecosystem service in producing recreational benefits, it is therefore important to consider only those valuation estimates that assess the effects of changes in the ECE habitat on the tourism and recreation benefits, but not the additional influence of any human inputs. The same approach should be taken for those "final" ecosystem services, such as coastal protection, erosion control, nutrient cycling, water purification, and carbon sequestration, which may benefit human well-being without any additional human-provided goods and services. But if "final" services do involve any human inputs, the appropriate valuation estimates should show how changes in these services affect human welfare, after controlling for the influence of these additional human-provided goods and services. Although this approach to selecting among valuation estimates of various ECE services seems straightforward, in practice there are a number of challenges to overcome. These difficulties are key to understanding an important finding of our review: Whereas considerable progress has been made in valuing a handful of ECE services, there are still a large number of these services that have either no or very unreliable valuation estimates.

The most significant problem faced in valuing ecosystem services, including those of ECEs, is that very few are marketed. Some of the products arising from ECEs, such as raw materials, food, and fish harvests, are bought and sold in markets. Given that the price and quantities of these marketed products are easy to observe, there are many value estimates of the contribution of the environmental input to this production. However, this valuation is more complicated than it appears. Market conditions and regulatory policies for the marketed output will influence the values imputed to the environment input (Freeman 2003:259–296, McConnell and Bockstael 2005, Barbier 2007). For example, one important service of many ECEs is the maintenance of fisheries through providing coastal breeding and nursery habitat. Although many fisheries are exploited for commercial harvests sold in domestic and international markets, studies have shown that the inability to control fishing access and the presence of production subsidies and other market distortions can impact harvests, the price of fish sold, and ultimately, the estimated value of ECE habitats in supporting commercial fisheries (Freeman 1991, Barbier 2007, Smith 2007).

However, the majority of other key ECE services do not lead to marketed outputs. These include many services arising from ecosystem processes and functions that benefit human beings largely without any additional input from them, such as coastal protection, nutrient cycling, erosion control, water purification, and carbon sequestration. In recent years, substantial progress has been made by economists working with ecologists and other natural scientists in applying environmental valuation methodologies to assess the welfare contribution of these services. The various nonmarket valuation methods employed for ecosystem services are essentially the standard techniques that are available to economists. For example, Freeman (2003), Pagiola et al. (2004), NRC (2005), Barbier (2007), U.S. EPA (2009), Mendelsohn and Olmstead (2009), and Hanley and Barbier (2009) discuss how these standard valuation methods are best applied to ecosystem services, emphasizing in particular both the advantages and the shortcomings of the different methods and their application. However, what makes applying these methods especially difficult is that they require three important, and interrelated, steps (Barbier 1994, 2007, Freeman 2003, NRC 2005, Polasky and Segerson 2009).

The first step involves determining how best to characterize the change in ecosystem structure, functions, and processes that gives rise to the change in the ecosystem service. For instance, the change could be in the spatial area or quality of a particular type of ECE habitat, such as a mangrove forest, marsh vegetation, or sand dune extent. It could also be a change in a key population, such as fish or main predator. Alternatively, the change could be due to variation in the flow of water, energy or nutrients through the system, such as the variability in tidal surges due to coastal storm events or the influx of organic waste from pollution upstream from ECEs.

The second step requires tracing how the changes in ecosystem structure, functions, and processes influence the quantities and qualities of ecosystem service flows to people. Underlying each ecosystem service is a range of important energy flow, biogeochemical and biotic processes and functions. For example, water purification by seagrass beds is linked to the ecological processes of nutrient uptake and suspended particle deposition (Rybicki 1997, Koch et al. 2006). However, the key ecological process and functions that generate an ecosystem service are, in turn, controlled by certain abiotic and biotic components that are unique to each ecosystem's structure. The various controlling components that may affect nutrient uptake and particle deposition by seagrasses include seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, and light availability. Only when these first two steps are completed is it possible to conduct the final step, which involves using existing economic valuation method to assess the changes in human well-being that result from the change in ecosystem services.

As summarized by NRC (2005:2) this three-step approach implies that "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." This approach is summarized in Fig. 1. Human drivers of ecosystem change affect important ecosystem processes and functions and their controlling components. Assessing this change is fundamental yet difficult. However, "making the translation from ecosystem structure and function to ecosystem goods and services (i.e., the ecological production) is even more difficult" and "probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function" (NRC 2005:2–3). Similarly, Polasky and Segerson (2009:422) maintain that "among the more practical difficulties that arise in either predicting changes in service flows or estimating the associated value of ecosystem services" include the "lack of multiproduct, ecological production functions to quantitatively map ecosystem structure and function to a flow of services that can then be valued."

We find that, for many key ECE services, the integration of the "ecological production function" with the "economic valuation function" is incomplete. In many instances, how to go about making this linkage is poorly understood. However, for a handful of services, considerable progress has been made in estimating how the structure and functions of ECEs generate economic benefits. Thus, the main purpose of our review is to illustrate the current state of identifying, assessing, and valuing the key ecosystem services of ECEs, which is motivated by an important question: What is the current state of progress in integrating knowledge about the "ecological production function" underlying each important ECE service with economic methods to value changes in this service in terms of impacts on human welfare? To answer this important question, we adopt the following approach.

First, for each of five critical ECEs, coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes, we identified the main ecosystem services associated with each habitat. Second, we provided an overview of the "ecological production function" underlying each service by assessing current knowledge of the important ecosystem processes, functions, and controlling components that are vital to this service. Third, where possible, we cited estimates of economic values arising from each service, and identified those services where there is no reliable estimate of an economic value. Fourth, we discussed briefly the main human drivers of ecosystem change that are affecting each ECE habitat. Finally, the results of our review are summarized in a table for each ECE. This facilitates comparison across all five habitats and also illustrates the important "gaps" in the current state of valuing some key ECE services. To keep the summary table short, we selected only one valuation estimate as a representative example. In some cases, it may be the only valuation estimate of a particular ecosystem service; in others, we have tried to choose one of the best examples from recent studies.

Note that our purpose in reviewing valuation estimates of ECE services is, first, to determine which services have at least one or more reliable estimate and which do not, and, second, to identify future areas of ecological and economic research to further progress in valuing ECE services. We do not attempt to quantify the total number of valuation studies for each ECE service, nor do we analyze in detail the various valuation methods used in assessing an ecosystem service. Instead, we selected those examples of valuation studies that conform to the standard and appropriate techniques that are recommended for application to various ecosystem services, as discussed in Freeman (2003), Pagiola et al. (2004), NRC (2005), Barbier (2007), Hanley and Barbier (2009), U.S. EPA (2009), and Mendelsohn and Olmstead (2009). The interested reader should consult these references for a comprehensive discussion of economic nonmarket valuation methods and their suitable application to ecosystem services.

Because our aim is to assess the extent to which reliable valuation estimates exist for each identified ECE service, we have reported each estimate as it appears in the original valuation study. This is for two principal reasons. First, many of the studies are for specific ECE habitats in distinct locations at different time periods, such as the recreation value of several coral reef marine parks in the Seychelles (Mathieu et al. 2003), the value of increased offshore fishery production from mangrove habitat in Thailand (Barbier 2007), or the benefits of beach restoration in the U.S. states of Maine and New Hampshire (Huang et al. 2007). Each study also uses specific measures and units of value appropriate for the relevant study. For example, in the Seychelles study, the value estimate was expressed in terms of the average consumer surplus per tourist for a single year, the Maine and New Hampshire study estimated each household's willingness to pay for an erosion control program to preserve five miles of beach, and the Thailand study calculated the capitalized value per hectare of mangrove in terms of offshore fishery production. Although it is possible to make assumptions to transform the valuation estimate of each study into the same physical units (e.g., per hectare), temporal period (e.g., capitalized or annual value), or currency (e.g., US$), we do not think such a transformation is warranted for the purposes of this study.

Second, we do not alter the original valuation estimates into a common unit of measure (such as US$$\cdot$ha$^{-1}\cdot$yr^{-1} in 2010 prices) because of the concern that such standardizing of values will be misused or misinterpreted. For example, one might be tempted to "add up" all the ecosystem service values and come up with a "total value" of a particular ECE habitat, such as a salt marsh. Or, one might take the estimate for a specific location, such as the recreation value of several

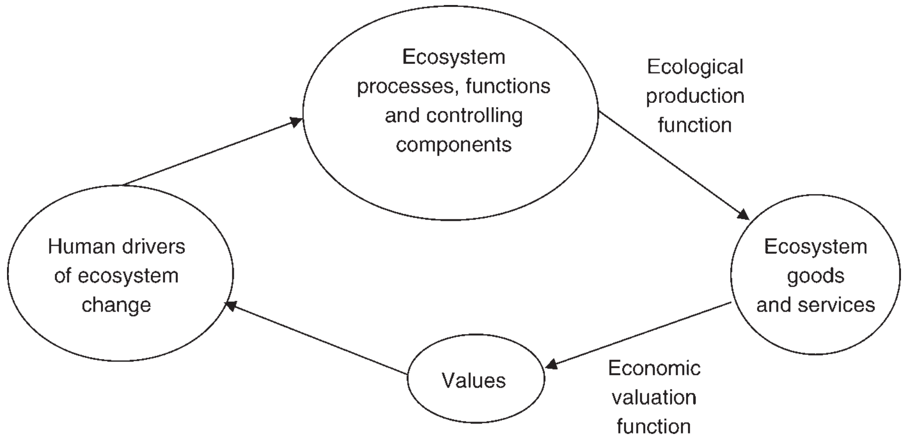

FIG. 1. Key interrelated steps in the valuation of ecosystem goods and services. This figure is adapted from NRC (2005: Fig. 1-3).

coral reef marine parks in the Seychelles (Mathieu et al. 2003), and "scale it up" by all the total hectares of coral reefs in the Indian Ocean or even the world to come up with a regional or global value of the recreational value of coral reefs. As argued by Bockstael et al. (2000:1396), when "the original studies valued small changes in specific and localized components of individual ecosystems . . . it is incorrect to extrapolate the value estimates obtained in any of these studies to a much larger scale, let alone to suppose that the extrapolated estimates could then be added together."

Finally, because our efforts here focus on identifying individual ECE services and any reliable estimates that value changes in these specific services, we do not emphasize valuation studies that estimate the value of entire ecosystems to human beings or assessing broader values, such as many nonuse existence and bequest values, that relate to the protection of ecosystems. However, we do recognize that such values are an important motivation for the willingness to pay by many members of society to protect ecosystems, including ECEs.

For example, Fig. 2 is a more detailed version of Fig. 1, emphasizing the economic valuation component of the latter diagram. As indicated in Fig. 2, there are a number of different ways in which humans benefit from, or value, ecosystem goods and services. The first distinction is between the "use values" as opposed to "nonuse values" arising from these goods and services. Typically, use values involve some human "interaction" with the environment, whereas nonuse values do not, as they represent an individual valuing the pure "existence" of a natural habitat or ecosystem or wanting to "bequest" it to future generations. Direct-use values refer to both consumptive and nonconsumptive uses that involve some form of direct physical interaction with environmental goods and services, such as recreational activities, resource harvesting, drinking clean water, breathing unpolluted air, and so forth. Indirect-use values refer to those ecosystem services whose values can only be measured indirectly, since they are derived from supporting and protecting activities that have directly measurable values.

As is apparent from Tables 1–5, the individual ECE services that we identified and discuss contribute to consumptive direct-use values (e.g., raw materials and food), nonconsumptive direct-use values (e.g., tourism, recreation, education, and research), and indirect-use values (e.g., coastal protection, erosion control, water catchment and purification, maintenance of beneficial species, and carbon sequestration). When it comes to valuing whether or not to create national parks from ECEs, or to protect entire ecosystems, assessing non-users' willingness to pay is also important. For example, Bateman and Langford (1997) assess the nonuse values of households across Great Britain for preserving the Norfolk and Suffolk Broads coastal wetlands in the United Kingdom from salt water intrusion. Even poor coastal communities in Malaysia, Micronesia, and Sri Lanka show considerable existence and other nonuse values for mangroves that can justify the creation of national parks and other protection measures (Naylor and Drew 1998, Othman et al. 2004, Wattage and Mardle 2008). As our review highlights how ECEs globally are endangered by a wide range of human drivers of change, it will be important that future studies assess all the use and nonuse values that arise from ecosystem goods and services to determine whether it is worth preserving or restoring critical ECEs.

REVIEWS

RESULTS: THE KEY SERVICES AND VALUES OF ECEs

In the following sections, we provide an overview of the results of our review of the main ecological services for five ECEs, arranged in order of most to least submerged: coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes. To give an indication of the "ecological production function" underlying the ecological services generated by each

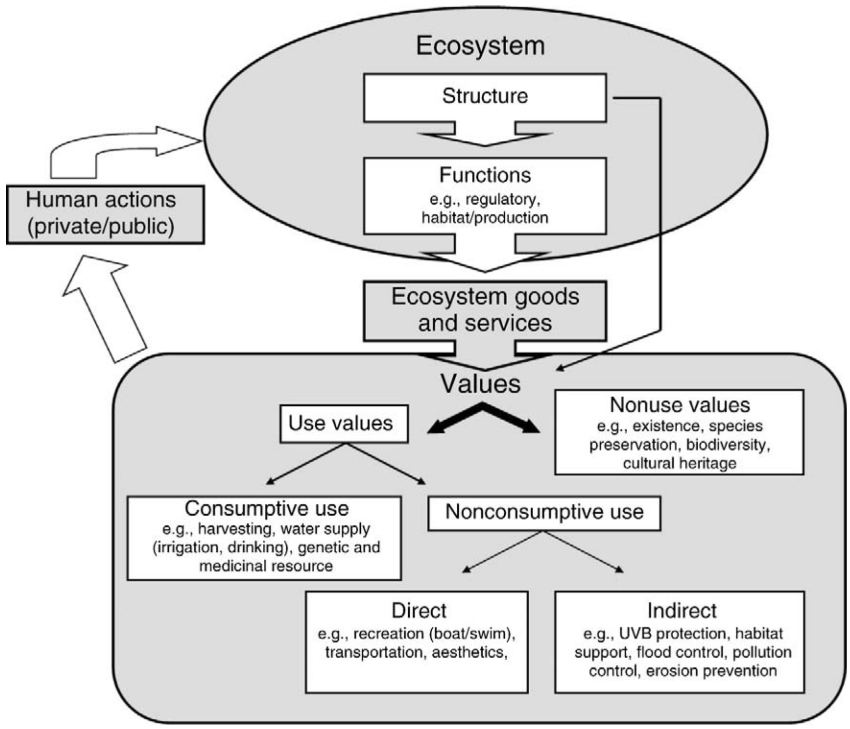

FIG. 2. Economic valuation of ecosystem goods and services. UVB is ultraviolet-B radiation from sunlight, which can cause skin cancer. This figure is adapted from NRC (2005: Fig. 4-1).

ECE (see Fig. 1), we outline briefly its key ecological structure, processes and functions, and identify the main controlling abiotic and biotic components. When available, we cite estimates of economic values from these services. The results give an indication as to the level of progress in valuing key ECE services and, equally important, where more integrated work on ecological and economic assessment of ecosystem services needs to be done.

Coral reefs

Coral reefs are structurally complex limestone habitats that form in shallow coastal waters of the tropics. Reefs can form nearshore and extend hundreds of kilometers in shallow offshore environments. Coral reefs are created by sedentary cnidarians (corals) that accrete calcium carbonate and feed on both zooplankton and maintain a mutualistic symbiosis with photosynthetic dinoflagellates. 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. In addition, coralline algae play an important role in stabilizing and cementing the coral reef structure. The community composition of reefs depends on global, regional, and local factors, which interact to produce the wide variety of coral reefs present on earth (Connell et al. 1997, Glynn 1997, Pandolfi 2002, Hughes et al. 2005).

As outlined in Table 1, coral reefs provide a number of ecosystem services to humans including raw materials, coastal protection, maintenance of fisheries, nutrient cycling, and tourism, recreation, education, and research. The table indicates representative examples of the values of some of these services, where they are available.

Historically, live reefs have served as a source of lime, which is an essential material in the manufacturing of mortar and cement and road building, and is 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. As there are no examples of such coral mining being conducted sustainably, we have not included any value estimates in Table 1.

An important ecosystem service provided by coral reefs is coastal protection or the buffering of shorelines from severe weather, thus protecting coastal human populations, property, and economic activities. As indicated in Table 1, this service is directly related to the economic processes and functions of attenuating or dissipating waves and facilitating beach and shoreline retention. 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. Despite the importance of this coastal protection service, very few economic studies have estimated a value for it. Those

TABLE 1. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for nearshore coral reefs.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials	generates biological productivity and diversity	reef size and depth, coral type, habitat quality	estimates unavailable	climate change, blast or cyanide fishing, lime mining, eutrophication, sedimentation, coastal development, dredging, pollution, biological invasion
Coastal protection	attenuates and/or dissipates waves, sediment retention	wave height and length, water depth above reef crest, reef length and distance from shore, coral species, wind climate	US\$174·ha^{-1}·yr^{-1} for Indian Ocean based on impacts from 1998 bleaching event on property values (Wilkinson et al. 1999)	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	coral species and density, habitat quality, food sources, hydrodynamic conditions	US\$15–45 000·km^{-2}·yr^{-1} in sustainable fishing for local consumption and \$5–10 000·km^{-2}·yr^{-1} for live-fish export, the Philippines (White et al. 2000)	
Nutrient cycling	provides biogeochemical activity, sedimentation, biological productivity	coral species and density, sediment deposition, subsidence, coastal geomorphology	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic reefscapes, suitable habitat for diverse fauna and flora	lagoon size, beach area, wave height, habitat quality, coral species and density, diversity	US\$88 000 total consumer surplus for 40 000 tourists to marine parks, Seychelles (Mathieu et al. 2003) and meta-analysis of recreational values (Brander et al. 2007)	

studies that do exist tend to use benefit transfer and replacement cost methods of valuation in an ad hoc manner, which undermine the reliability of the value estimates (see Chong 2005 and Barbier 2007 for further discussion). However, the widespread reef destruction caused by catastrophic events and global change, such as hurricanes, typhoons, and coral bleaching, gives some indication of the value of the lost storm protection services. For example, as a result of the 1998 bleaching event in the Indian Ocean, the expected loss in property values from declining reef protection was estimated to be US\$174·ha^{-1}·yr^{-1} (hereafter all values in US\$, unless otherwise stated; Wilkinson et al. 1999).

Coral reefs also serve to maintain fisheries through the enhancement of ecologically and economically important species by providing shelter space and substrate for smaller organisms, and food sources 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 parrot fishes), and large invertebrates (e.g., giant clams, conch, lobsters, and crabs). The commercial value of these fisheries can be significant for some economies. For example, fish harvested from Hawaiian coral reefs are estimated to contribute \$1.3 million yearly to the Hawaiian economy (Cesar and van Beukering 2004). From 1982 to 2002, small-scale, predominantly coral reef, fisheries contributed \$54.7 million to the economies of America Samoa and the Commonwealth of the Northern Mariana Islands (Zeller et al. 2007).

Additional fishery harvests consist of the live-animal aquarium trade, based on corals, small fishes, and invertebrates collected from reefs. The aquarium trade has substantially expanded in the past 20 years, listed in 1985 as making \$20–40 million/yr as a world market (Wood 1985) and expanding to an estimated \$90–300 million/yr in 2002 (Sadovy and Vincent 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 byproducts.

Reliable values for the sustainable production of coral reef fish for local consumption and the aquarium trade are rare. White et al. (2000) provide some estimates for the Philippines. The potential annual revenue for sustainable fish production could be $15–45 000/km^2 of healthy coral reef for local consumption and $5–10 000/km^2 for live fish export. Zhang and Smith (*in press*) estimate the maximum sustainable yield to the Gulf of Mexico reef fishery (mainly grouper and snapper species, amberjack, and tilefish) to be ~1.30 million kg/month (~2.86 million pounds/month). Though the reefs in the Gulf of Mexico are generally exposed limestone or sandstone and not coral, the habitats are similar in their structural complexity, which is an important factor in protecting young fish and smaller species from predation.

Coral reef ecosystems also perform important services by cycling organic and inorganic nutrients. Despite housing a great deal of inorganic carbon in the limestone skeleton that makes up the structure of the reef, coral reefs may actually be a net source of atmospheric carbon dioxide (Kawahata et al. 1997). Reefs do, however, contribute significantly to the global calcium carbonate ($CaCO_3$) budget, estimated as 26% of coastal marine $CaCO_3$ and 11% of the total $CaCO_3$ precipitation (Hallock 1997, Gattuso et al. 1998). Reefs additionally transfer excess nitrogen production from cyanobacteria and benthic microbes on the reef to the pelagic (water column) environment (Moberg and Folke 1999). Though poorly quantified, the sequestering of $CaCO_3$ to form the foundation or habitat of the reef is the primary reason for such high abundance and diversity of organisms. Unfortunately, as indicated in Table 1, there are no reliable estimates of the economic value of the nutrient cycling and transfer services of coral reefs.

Coral reefs and associated placid lagoons are also economically valuable for the tourism and recreational activities they support. Resorts depend on the aesthetically turquoise lagoons, white sandy beaches, and underwater opportunities on the reef to attract 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 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/year (van Beukering and Cesar 2004). Revenues from coral reef tourism in the Pulau Payar Marine Park, Malaysia, are estimated at $390 000/year (Yeo 2002), and coral reef diving earns gross revenue of $10 500–45 540/year in the Bohol Marine Triangle, the Philippines (Samonte-Tan et al. 2007).

However, estimates of the recreational value of individual reefs should be interpreted with caution as a recent review of such studies found substantial bias in the estimates of individual recreation values (Brander et al. 2007). Reliable estimates can be made if such biases are controlled. For example, Mathieu et al. (2003) found that the average consumer surplus per tourist visiting the marine national parks in the Seychelles is $2.20, giving a total consumer surplus estimate of $88 000 for the 40 000 tourists to the coral reefs in 1997. Tapsuwan and Asafu-Adjaye (2008) were able to estimate the economic value of scuba diving in the Similan Island coral reefs in Thailand, controlling for diver's attitude toward the quality of the dive site, frequency of dive trips, and socioeconomic characteristics, including whether divers were Thai or foreign. The authors estimated a consumer surplus value of $3233 per person per dive trip.

In addition to tourism and recreation, reefs also provide substantial services through research opportunities for scientists, work that is essential to basic and applied science (Greenstein and Pandolfi 2008). There are no reliable estimates of this value for coral reefs. As a rough indication of this value, expenditures for field work, primary data gathering, boat/vessel rental, supplies, and diving equipment amount to $32–111·ha^{-1}·yr^{-1} in Bohol Marine Triangle, the Philippines (Samonte-Tan et al. 2007).

Despite the numerous economic benefits coral reefs provide, reef ecosystems are under threat of irrevocable decline worldwide from a suite of anthropogenic stressors. Localized stressors (i.e., within reefs or archipelagos) include overfishing, dynamite or cyanide fishing, pollution, mining, eutrophication, coastal development, dredging, sedimentation, and biological invasion (e.g., Hoegh-Guldberg 1999, Gardner et al. 2003, Bellwood et al. 2004, Hoegh-Guldberg et al. 2007). A variety of reef ecosystem services may be affected by coral degradation. For example, areas in Sumatra where dynamite fishing had occurred suffered 70% greater wave heights than undisturbed areas during the 2004 Indian Ocean Tsunami (Fernando et al. 2005). Blast fishing can also have negative effects on local economies by reducing the amount of available reef for tourisms; in Indonesia, blast fishing led to the loss of a reef that was valued at $306 800/km^2 (Pet-Soede et al. 1999). Overfishing has important cascading consequences on both reef ecosystem function and sustainable production by inducing phase shifts (Mumby et al. 2006, 2007). Overharvesting by the aquarium industry has also been documented on local levels (Lubbock and Polunin 1975, Warren-Rhodes et al. 2004). Moreover, eutrophication-induced algal blooms led to millions of dollars of lost tourism revenue in Hawaii (van Beukering and Cesar 2004).

Global-scale climate change is also threatening reefs through coral bleaching, disease, and ocean acidification, leading to both reef destruction and structural degradation (Graham et al. 2007, Hoegh-Guldberg et al. 2007, Carpenter et al. 2008). Several important reef ecosystem services are likely to be affected. Though the economic impacts of climate change on fisheries remain somewhat unclear, the benthic composition of reefs is likely to shift, thus affecting overall fish productivity and

TABLE 2. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for seagrasses.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	estimates unavailable	eutrophication, overharvesting, coastal development, vegetation disturbance, dredging, aquaculture, climate change, sea level rise
Coastal protection	attenuates and/or dissipates waves	wave height and length, water depth above canopy, seagrass bed size and distance from shore, wind climate, beach slope, seagrass species and density, reproductive stage	estimates unavailable	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, subsidence, tidal stage, wave climate, coastal geomorphology, seagrass species and density	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, light availability	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	seagrass species and density, habitat quality, food sources, hydrodynamic conditions	loss of 12 700 ha of seagrasses in Australia; associated with lost fishery production of AU$235 000 (McArthur and Boland 2006)	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	seagrass species and density, water depth, light availability, burial rates, biomass export	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic submerged vegetated landscape, suitable habitat for diverse flora and fauna	biological productivity, storm events, habitat quality, seagrass species and density, diversity	estimates unavailable	

harvests, as well as the availability of the most valued fishes collected in the aquarium trade (Pratchett et al. 2008). Reductions in tourism due to recent climate change-driven coral bleaching events are estimated in the billions (Wilkinson et al. 1999, Pratchett et al. 2008). The overall estimated economic damages from lost fisheries production, tourism and recreation, coastal protection, and other ecosystem services from the 1998 Indian Ocean coral bleaching event have ranged from $706 million to $8.2 billion (Wilkinson et al. 1999).

Seagrass beds

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, cobble) and grow to depths where ~11% of surface light reaches the bottom (Duarte 1991). Seagrasses 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, Waycott et al. 2009), seagrasses receive little attention when compared to other ECEs (Duarte et al. 2008).

As indicated in Table 2, seagrass beds provide a wide range of ecosystem services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research, yet reliable estimates of the economic values of most of these services are lacking.

Although in the past seagrasses were highly valued as raw materials and food, modern direct uses of seagrasses are rather limited. For example, seagrasses are still harvested in Tanzania, Portugal, and Australia, where they are used as fertilizer (Hemminga and Duarte 2000, de la Torre-Castro and Rönnbäck 2004). In the Chesapeake Bay, USA, seagrass by-catch or beach-cast is used to keep crabs moist during transport. In East Africa, some species are served as salad, while others are used in potions and rituals (de la Torre-Castro and Rönnbäck 2004). In the Solomon Islands, roots of the seagrass *Enhalus acoroides* are sometimes used as food, while leaf fibers are used to make necklaces and to provide spiritual benefits such as a gift to a newborn child, for fishing luck, and to remove an aphrodisiac spell (Lauer and Aswani 2010). However, currently there are no reliable estimates of the values of these food and raw material uses of harvested seagrasses.

Coastal protection and erosion control are often listed as important ecosystem services provided by seagrasses (Hemminga and Duarte 2000, Spalding et al. 2003, Koch et al. 2009). Seagrasses can attenuate waves and, as a result, smaller waves reach the adjacent shoreline (Fonseca and Cahalan 1992, Koch 1996, Prager and Halley 1999). Coastal protection is 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 deeper waters, their contribution to wave attenuation and coastal protection is more limited. Sediment stabilization by seagrass roots and rhizomes, as well as by their beach-casted debris is important for controlling coastal erosion (Hemminga and Nieuwenhuize 1990). The benefits seagrasses provide in terms of coastal protection and erosion control via sediment stabilization and wave attenuation are yet to be valued satisfactorily.

Water purification, or the increase in water clarity, by seagrasses occurs via two processes: nutrient uptake and suspended particle deposition. Seagrasses not only remove nutrients from the sediments and water column (Lee and Dunton 1999), but also their leaves are colonized by algae (epiphytes), which further remove nutrients from the water column (Cornelisen and Thomas 2006). 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 (Romero et al. 2006). In addition to reducing nutrients, seagrass beds also decrease the concentration of suspended particles (e.g., sediment and microalgae) from the water (Gacia et al. 1999). Leaves in the water column provide an obstruction to water flow and, as a result, currents and waves are reduced within seagrass canopies causing particles to be deposited (Koch et al. 2006). This water purification effect can be quite dramatic with clearer water in vegetated areas compared to those without vegetation (Rybicki 1997). No reliable economic estimates exist for the water purification service provided by seagrass beds.

Seagrasses also generate value as habitat for ecologically and economically important species such as scallops, shrimp, crabs, and juvenile fish. Seagrasses protect these species from predators and provide food in the form of leaves, detritus, and epiphytes. The market value of the potential shrimp yield in seagrass beds in Western Australia is estimated to be between \$684 and \$2511$\cdot$ha$^{-1}\cdot$yr^{-1} (Watson et al. 1993). In Bohol Marine Triangle, the Philippines, the annual net revenue from gleaning mollusks and echinoderms (e.g., starfish, sea urchins, sea cucumbers, etc.) from seagrass beds at low tide ranges from \$12–120/ha and from fishing \$8–84/ha (Samonte-Tan et al. 2007). The fish, shrimp, and crab yield in southern Australia is valued at US\$1436$\cdot$ha$^{-1}\cdot$yr^{-1} (McArthur and Boland 2006). Based on the latter estimate, a loss of 2700 ha of seagrass beds results in lost fishery production of AU\$235 000 (Table 2).

Seagrasses are involved in carbon sequestration by using 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 sequestration capacity, although this effect has yet to be valued.

Anthropogenic influences such as eutrophication, overharvesting, sediment runoff, algal blooms, commercial fisheries and aquaculture practices, vegetation disturbance, global warming, and sea level rise are among the causes for the decline of seagrasses worldwide (Orth et al. 2006, Waycott et al. 2009). With the disappearance of seagrasses, valuable ecosystem services are also lost (McArthur and Boland 2006). Yet, as very few of these benefits have been estimated reliably (see Table 2), we have only historical and anecdotal evidence of the likely economic impacts. For example, the disappearance of most seagrasses in Long Island, USA, in the 1930s due to wasting disease led to the collapse of the scallop industry (Orth et al. 2006).

Salt marshes

Salt marshes are intertidal grasslands that form in low-energy, wave-protected shorelines along continental margins. Extensive salt marshes (>2 km in width) establish and grow both behind barrier-island systems and along the wave-protected shorelines of bays and estuaries. Salt marshes are characterized by sharp zonation of plants and low species diversity, but extremely high primary and secondary production. 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 (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 2002, Silliman and Bortolus 2003, Silliman et al. 2005, Henry and Jefferies 2009).

Among coastal ecosystems, salt marshes provide a high number of valuable benefits to humans, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research. Some of these important values have been estimated (Table 3).

For over 8000 years, humans have relied on salt marshes for direct provisioning of raw materials and food (Davy et al. 2009). Although harvesting of marsh grasses and use of salt marshes as pasture lands has decreased today, these services are still important locally in both developed and developing areas of the world (Bromberg-Gedan et al. 2009). For example, in the Ribble estuary on England's west coast, annual net income from grazing in a salt marsh nature reserve is: £15.27$\cdot$ha$^{-1}\cdot$yr^{-1} (King and Lester 1995).

For thousands of years, salt marshes have provided coastal protection from waves and storm surge, as well as from coastal erosion, for humans (Davy et al. 2009). By stabilizing sediment, increasing the intertidal height, and providing baffling vertical structures (grass), salt marshes reduce impacts of incoming waves by reducing their velocity, height, and duration (Morgan et al. 2009; Bromberg-Gedan et al., *in press*). Marshes are also likely to reduce storm surge duration and height by providing extra water uptake and holding capacity in comparison to the sediments of unvegetated mudflats. This storm protection value can be substantial, as a study of the protection against hurricanes by coastal wetlands along the U.S. Atlantic and Gulf coasts reveals (Table 3; Costanza et al. 2008). However, there are no reliable estimates of the economic value of salt marshes in controlling coastal erosion.

Salt marshes act as natural filters that purify water entering the estuary (Mitsch and Gosselink 2008). As water (e.g., from rivers, terrestrial runoff, groundwater, or rain) passes through marshes, it slows due to the baffling and friction effect of upright grasses (Morgan et al. 2009). Suspended sediments are then deposited on the marsh surface, facilitating nutrient uptake by salt marsh grasses. This water filtration service benefits human health, but also adjacent ecosystems, such as seagrasses, which may be degraded by nutrients and pollutants. In southern Louisiana, USA, treatment of wastewater by predominantly marsh swamps achieved capitalized cost savings of \$785 to \$15 000/acre (1 acre = 0.4 ha) compared to conventional municipal treatment (Breaux et al. 1995).

Salt marsh ecosystems also serve to maintain fisheries by boosting the production of economically and ecologically important fishery species, such as shrimp, oysters, clams, and fishes (Boesch and Turner 1984, MacKenzie and Dionne 2008). 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. 2000). Because of their complex and tightly packed plant structure, marshes provide habitat that is mostly inaccessible to large fishes, thus providing protection and shelter for the increased growth and survival of young fishes, shrimp, and shellfish (Boesch and Turner 1984). For example, the capitalized value of an acre of salt marsh in terms of recreational fishing is estimated to be \$6471 and \$981 for the east and west coasts of Florida, USA, respectively (Bell 1997). The contribution of an additional acre of salt marsh to the value of the Gulf Coast blue crab fishery ranges from \$0.19 to \$1.89/acre (Freeman 1991).

As one of the most productive ecosystems in the world (up to 3900 g C$\cdot$m$^{-2}\cdot$yr^{-1}), salt marshes sequester millions of tons of carbon annually (Mitsch and Gosselink 2008). Because of the anoxic nature of the marsh soils (as in most wetlands), carbon sequestered by salt marsh plants during photosynthesis is often shifted 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 peat (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. However, to our knowledge, there is no valuation estimation of this carbon sequestration service. Based on an estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C/ha by Chmura et al. (2003), and employing the 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg converted to \$2000, we calculated a value of \$30.50$\cdot$ha$^{-1}\cdot$yr^{-1} as an approximate indicator of this benefit, but this is likely to vary greatly depending on latitude, as warmer marshes do not accumulate peat like their colder counterparts.

Salt marshes provide important habitat for many other beneficial species, which are important for tourism, recreation, education, and research. For example, estimates from land sales and leases for marshes in England suggest prices in the range of £150–493/acre for bird shooting and wildfowling (King and Lester 1995). Respondents were willing to pay £31.60/person to create otter habitat and £1.20 to

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TABLE 3. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for salt marshes.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality, inundation depth, habitat quality, healthy predator populations	£15.27· ha^{-1}·yr^{-1} net income from livestock grazing, UK (King and Lester 1995)	marsh reclamation, vegetation disturbance, climate change, sea level rise, pollution, altered hydrological regimes, biological invasion
Coastal protection	attenuates and/or dissipates waves	tidal height, wave height and length, water depth in or above canopy, marsh area and width, wind climate, marsh species and density, local geomorphology	US\$8236·ha^{-1}·yr^{-1} in reduced hurricane damages, USA (Costanza et al. 2008)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, coastal geomorphology, subsidence, fluvial sediment deposition and load, marsh grass species and density, distance from sea edge	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	marsh grass species and density, marsh quality and area, nutrient and sediment load, water supply and quality, healthy predator populations	US\$785–15 000/acre capitalized cost savings over traditional waste treatment, USA (Breaux et al. 1995)†	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	marsh grass species and density, marsh quality and area, primary productivity, healthy predator populations	US\$6471/acre and \$981/acre capitalized value for recreational fishing for the east and west coasts, respectively, of Florida, USA (Bell 1997) and \$0.19–1.89/acre marginal value product in Gulf Coast blue crab fishery, USA (Freeman 1991)†	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	marsh grass species and density, sediment type, primary productivity, healthy predator populations	US\$30.50·ha^{-1}·yr^{-1}‡	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	marsh grass species and density, habitat quality and area, prey species availability, healthy predator populations	£31.60/person for otter habitat creation and £1.20/person for protecting birds, UK (Birl and Cox 2007)	

† One acre = 0.4 ha.
‡ Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C·ha^{-1}·yr^{-1} and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US\$2000.

protect birds in the Severn Estuary Wetlands bordering England and Wales (Birol and Cox 2007).

Current human threats to salt marshes include biological invasions, eutrophication, climate change and sea level rise, increasing air and sea surface temperatures, increasing CO_2 concentrations, altered hydrologic regimes, marsh reclamation, vegetation disturbance, and pollution (Silliman et al. 2009). As indicated in Table 3, a growing number of valuable marsh services are lost with the destruction of this habitat. Approximately 50% of the original salt marsh ecosystems have been degraded or lost globally, and in some areas, such as the West Coast of the USA, the loss is >90% (Bromberg and Silliman 2009, Bromberg-Gedan et al. 2009). This is likely to be exacerbated by the recent Gulf of Mexico oil spill in 2010.

TABLE 4. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for mangroves.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	US\$484–585·ha^{-1}·yr^{-1} capitalized value of collected products, Thailand (Barbier 2007)	mangrove disturbance, degradation, conversion; coastline disturbance; pollution; upstream soil loss; overharvesting of resources
Coastal protection	attenuates and/or dissipates waves and wind energy	tidal height, wave height and length, wind velocity, beach slope, tide height, vegetation type and density, distance from sea edge	US\$8966–10 821/ha capitalized value for storm protection, Thailand (Barbier 2007)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, fluvial sediment deposition, subsidence, coastal geomorphology, vegetation type and density, distance from sea edge	US\$3679·ha^{-1}·yr^{-1} annualized replacement cost, Thailand (Sathirathai and Barbier 2001)	
Water purification	provides nutrient and pollution uptake, as well as particle retention and deposition	mangrove root length and density, mangrove quality and area	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	mangrove species and density, habitat quality and area, primary productivity	US\$708-\$987/ha capitalized value of increased offshore fishery production, Thailand (Barbier 2007)	
Carbon sequestration	generates biological productivity, biogeochemical activity, sedimentation	vegetation type and density, fluvial sediment deposition, subsidence, coastal geomorphology	US\$30.50·ha^{-1}·yr^{-1}†	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	mangrove species and density, habitat quality and area, prey species availability, healthy predator populations	estimates unavailable	

† Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C·ha^{-1}·yr^{-1}and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US\$2000.

Mangroves

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 (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). But since then, at least 35% of global mangrove area has been lost, and mangroves are currently disappearing at the rate of 1–2% annually (Valiela et al. 2001, Alongi 2002, FAO 2007).

The worldwide destruction of mangroves is of concern because they provide a number of highly valued ecosystems services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research (Table 4). 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 (Walters et al. 2008).

Of the ecosystem services listed, 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 fuel wood, 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 ecosystem services has been conducted for Thailand by Barbier (2007), who compared the net economic returns per hectare to shrimp farming, the costs of mangrove rehabilitation, and the value of mangrove services. All land uses were assumed to be instigated over a nine-year period (1996 to 2004), and the net present value (NPV) of each land use or ecosystem service was estimated in 1996 US$ per hectare. The NPV arising from the net income to local communities from collected forest and other products and shellfish was $484 to $584/ha. In addition, the NPV of mangroves as breeding and nursery habitat in support of offshore artisanal fisheries ranged from $708 to $987/ha, and the storm protection service was $8966 to $10 821/ha.

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, 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.

The Thailand example is not unusual; coastal households across the world typically benefit from the mangrove services, indicated in Table 4 (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, Lange and Jiddawi 2009, Nfotabong Atheull et al. 2009). Mangroves also provide important cultural benefits to coastal inhabitants. A study in Micronesia finds 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).

Since the 2004 Indian Ocean Tsunami, there has been considerable global interest in one particular service of mangroves: their role as natural barriers that protect the lives and properties of coastal communities from periodic storm events and flooding. Eco-hydrological evidence indicates that this protection service is based on the ability of mangroves to attenuate waves and thus reduce storm surges (Mazda et al. 1997, 2006, Massel et al. 1999, Wolanski 2007, Barbier et al. 2008, Koch et al. 2009). Comprehensive reviews of all the field assessments in the aftermath of the Indian Ocean Tsunami suggest that some areas were more protected by the presence of healthy mangroves, provided that the tidal wave was not too extreme in magnitude (Montgomery 2006, Braatz et al. 2007, Forbes and Broadhead 2007, Alongi 2008, Cochard et al. 2008). For other major storm events, there is more economic evidence of the protective role of mangroves. For example, during the 1999 cyclone that struck Orissa, India, mangroves significantly reduced the number of deaths as well as damages to property, livestock, agriculture, fisheries, and other assets (Badola and Hussain 2005, Das and Vincent 2009). Das and Vincent estimated that there could have been 1.72 additional deaths per village within 10 km of the coast if the mangrove width along shorelines had been reduced to zero. Losses incurred per household were greatest ($154) in a village that was protected by an embankment but had no mangroves compared to losses per household ($33) in a village protected only by mangrove forests (Badola and Hussain 2005).

The ability of mangroves to stabilize sediment and retain soil in their root structure reduces shoreline erosion and deterioration (Daehler and Strong 1996, Sathirathai and Barbier 2001, Thampanya et al. 2006, Wolanski 2007). But despite the importance of this erosion control service, very few economic studies have been conducted to value it. Existing studies tend to use the replacement cost methods of valuation, due to lack of data, which can undermine the reliability of the value estimates (Chong 2005, Barbier 2007). In Thailand, the annualized replacement cost of using artificial barriers instead of mangroves is estimated to be $\$3679 \cdot ha^{-1} \cdot yr^{-1}$ (Sathirathai and Barbier 2001).

Mangroves also serve as barriers in the other direction; their water purification functions protect coral reefs, seagrass beds, and important navigation waters against siltation and pollution (Wolanski 2007). In southern China, field experiments have been conducted to determine the feasibility of using mangrove wetlands for wastewater treatment (Chen et al. 2009). Mangrove roots may also serve as a sensitive bio-indicator for metal pollution in estuarine systems (MacFarlane et al. 2003). The economic value of the pollution control service of mangroves has not been reliably estimated, however.

Because mangroves are among the most productive and biogeochemically active ecosystems, they are important sources of global carbon sequestration. To date, the value of mangroves as a carbon sink has not been estimated. Based on an estimate of permanent carbon sequestration by all mangroves globally (Chmura et al. 2003), following the same approach described above for salt marshes (see *Salt marshes*), we calculate a value of $\$30.50 \cdot ha^{-1} \cdot yr^{-1}$ as an approximate indicator of this benefit for mangroves.

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%. 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). The extensive and rapid loss of mangroves globally reinforces the importance of measuring the value of such ecological services, and employing these values appropriately in coastal management and planning.

Sand beaches and dunes

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 vs. 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).

For centuries, due to their unique position between ocean and land, coastal beaches and dunes have provided humans with important services such as raw materials, coastal protection, erosion control, water catchment and purification, maintenance of wildlife, carbon sequestration, and tourism, recreation, education, and research (Table 5; Carter 1990, Pye and Tsoar 1990). However, very few of these services have been valued, with the exception of erosion control and recreation and tourism (Table 5).

Beaches and dunes provide raw materials in the form of sand that has been mined for centuries for multiple uses, including extraction of minerals such 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 effects, especially on coastal protection and aquifers.

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. As waves reach the shoreline they are attenuated by the beach slope and, at high tide, also 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 (Carter 1991, Hesp and Short 1999, Short 1999). 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; Hacker et al., *in press*). Measuring the coastal protective properties of sand shoreline systems involves understanding the relationship between beach and foredune shape and wave attenuation, especially in the aftermath of storms, hurricanes, or tsunamis (Leatherman 1979, Lui et al. 2005, Sallenger et al. 2006, Morton et al. 2007, Stockdon et al. 2007, Ruggiero et al. 2010). The economic value, although not calculated previously, is likely to be substantial. For example, Liu et al. (2005) report that, after the 2004 Indian Ocean Tsunami, there was total devastation and loss of 150 lives in a resort located directly behind where a foredune was removed to improve the scenic view of the beach and ocean.

Beaches and sand dunes provide sediment stabilization and soil retention in vegetation root structure, thus controlling coastal erosion and protecting recreational beaches, tourist-related business, ocean front properties, land for aquaculture and agriculture, and wildlife habitat. Although this service has not been valued directly, there have been a growing number of studies that value the benefits gained from erosion control programs that either preserve or "nourish" existing beaches and dunes (Landry et al. 2003, Kriesel and Landry 2004, Huang et al. 2007, Whitehead et al. 2008, Morgan and Hamilton 2010). Such programs often substitute for property owners building their own erosion protection structures, such as seawalls and groins, which can inadvertently accelerate the degradation of the coastal environment (Landry et al. 2003, Kriesel and Landry 2004). However, erosion control programs can also have negative effects on the surrounding environment, including affecting recreational beach use and views, displacing coastal erosion elsewhere, and disturbing wildlife habitat. For example, in the U.S. states of New Hampshire and Maine, a coastal erosion program that preserves five miles of beach is estimated to have net benefits, adjusted for the costs associated with the risk of injury to swimmers from the control measures, disturbance to wildlife habitat, and deterioration of water quality, of $4.45/household (Huang et al. 2007). Landry et al. (2003) find that a one-meter increase in beach width, or equivalently, the prevention of one meter of beach erosion, increased oceanfront and inlet-front property values by $233 on Tybee Island in the U.S. state of Georgia.

Another important service of coastal sand ecosystems is water catchment. Sand dunes are able to store significant amounts of water that can serve as aquifers for coastal populations (Carter 1990). For example, in the Meijendel dunes in The Netherlands, dune aquifers have been used as a source of drinking water for centuries (van der Meulen et al. 2004). The aquifer still supplies enough water for 1.5 million people in

TABLE 5. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for sand beaches and dunes.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials	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)	estimates unavailable for sustainable extraction	loss of sand through mining, development and coastal structures (e.g., jetties), vegetation disturbance, overuse of water, pollution, biological invasion
Coastal protection	attenuates and/or dissipates waves and reduces flooding and spray from sea	wave height and length, beach slope, tidal height, dune height, vegetation type and density, sand supply	estimates unavailable	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, subsidence, tidal stage, wave climate, coastal geomorphology, beach grass species and density	US$4.45/household for an erosion control program to preserve 8 km of beach, for Maine and New Hampshire beaches, USA (Huang et al. 2007)	
Water catchment and purification	stores and filters water through sand; raises water table	dune area, dune height, sand and water supply	estimates unavailable	
Maintenance of wildlife	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	estimates unavailable	
Carbon sequestration	generates biological productivity, biogeochemical activity	vegetative type and density, fluvial sediment deposition, subsidence, coastal geomorphology	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic landscapes, suitable habitat for diverse fauna and flora	dune and beach area, sand supply, wave height, grain size, habitat quality, wildlife species, density and diversity, desirable shells and rocks	US$166/trip or $1574 per visiting household per year for North Carolina beaches, USA (Landry and Liu 2009)	

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 $3.8 million/year, while the yearly income of the reserve was $99.2 million/year.

Coastal dunes can provide maintenance of wildlife in the form of habitat for fish, shellfish, birds, rodents, and ungulates, which have been captured or cultivated for food since humans first colonized the coast (Carter 1990, Pye and Tsoar 1990). In Europe, protection and restoration of dune wildlife and habitat has become a priority (Baeyens and Martínez 2004). In other regions of the world, dunes have been used for agricultural purposes (Pye and Tsoar 1990). However, there are no reliable estimates on the value of beaches and dunes as a source of habitat for wildlife.

Dunes that encourage vegetation growth and productivity will also assist in carbon sequestration, although this process is likely to vary with the type of vegetation, sediment deposition and subsidence, and coastal geomorphology. There are currently no estimates of the value of this service provided by dunes, however.

Beaches and dunes also supply important recreational benefits. 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 USA alone, 70% of the population visits the beach on vacation, and 85% of total tourism dollars come from beach visits (Houston 2008). An analysis of North Carolina beaches shows that implementation of a beach replenishment policy to improve beach width by an average of 100 feet would increase the average number of trips by visitors in the subsequent year from 11 to 14, with beach-goers willing to pay \$166/trip or \$1574 per visiting household per year (Landry and Liu 2009). Another study of North Carolina beaches found that widening beach width increases the consumer surplus of visitors by \$7/trip (Whitehead et al. 2008). However, overuse of dune habitat due to beach recreation can also 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).

Many of the services provided by sand beaches and dunes are threatened by human use, species invasions, and climate change (Brown and McLachlan 2002, Zarnetske et al. 2010; Hacker et al., *in press*). 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 (Ruggiero et al. 2010) and coastal freshwater catchment. The fact that no reliable estimates of these services are currently available is worrisome.

Discussion: Issues for Future Research

Our review of economic values of key ecosystem services for five estuarine and coastal ecosystems (coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes) reveals that progress has been made in estimating these benefits for some systems and services, but much work remains. For example, reliable valuation estimates are beginning to emerge for the key services of some ECEs, such as coral reefs, salt marshes, and mangroves, but many of the important benefits of seagrass beds and sand dunes and beaches have not been assessed properly. Even for coral reefs, marshes, and mangroves, important ecological services have yet to be valued reliably, such as cross-ecosystem nutrient transfer (coral reefs), erosion control (marshes), and pollution control (mangroves). Although more studies valuing ECE services have been conducted recently, our review shows that the number of reliable estimates is still relatively small.

Measurement issues, data availability, and other limitations continue to prevent the application of standard valuation methods to many ecosystem services. In circumstances where an ecological service is unique to a specific ecosystem and is difficult to value, often the cost of replacing the service or treating the damages arising from the loss of the service is used as a valuation approach. Such methods have been employed frequently to measure coastal protection, erosion control, and water purification services by ECEs (Ellis and Fisher 1987, Chong 2005, Barbier 2007). However, economists recommend that the replacement cost approach should be used with caution because, first, one is essentially estimating a benefit (e.g., storm protection) by a cost (e.g., the costs of constructing seawalls, groins, and other structures), and second, the human-built alternative is rarely the most cost-effective means of providing the service (Ellis and Fisher 1987, Barbier 1994, 2007, Freeman 2003, NRC 2005).

As summarized in our tables, ECE habitats tend to generate multiple ecosystem services. These typically range from tourism and recreation benefits to coastal protection, erosion control, nutrient cycling, water purification, and carbon sequestration to food and raw-material products. Where studies are aware of such multiple benefits, the current approach is still to value each service as if it is independent, as was done for coastal protection, habitat–fishery linkages, and raw materials for mangroves in Thailand (Barbier 2007). However, as our tables indicate, similar ecological processes and functions, as well as controlling components, may influence more than one ecosystem service. Such ecological interactions are bound to affect the value of multiple services arising from a single habitat, which is an important direction for future research in valuing ECE services.

For a growing number of services, there is evidence that 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 and Appeldoorn 2008, Barbier et al. 2008, Meynecke et al. 2008, Koch et al. 2009). For example, wave attenuation by coral reefs, seagrass beds, salt marshes, mangroves, and sand dunes provides protection against wind and wave damage caused by coastal storm and surge events, but the magnitude of protection will vary spatially across the extent of these habitats (Barbier et al. 2008, Koch et al. 2009). In particular, 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.

Valuation of coastal habitat support for offshore fisheries increasingly indicates that the value of this service varies spatially because the quality of the habitat is greater at the seaward edge or "fringe" of the coastal ecosystem than further inland (Peterson and Turner 1994, Manson et al. 2005, Aburto-Oropeza et al. 2008, Aguilar-Perera and Appeldoorn 2008). In the case of mangroves and salt marshes, the evidence suggests that both storm protection and habitat–fishery linkage benefits tend to decline with the distance inshore from the seaward edge of most coastal wetland habitats, such as mangroves and salt marshes. 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. Fishery landings also increased positively with the length of the mangrove fringe in a given location (Aburto-Oropeza et al. 2008). The tendency for these services to vary unidirectionally across such coastal landscapes has implications for modeling the provision of these services and valuing their benefits (Barbier 2008).

Coastal protection can also vary if damaging storm events occur when plant biomass and/or density are low (Koch et al. 2009). This is particularly important in temperate regions, where seasonal fluctuations of biomass may differ from the seasonal occurrence of storms. For example, along the U.S. Atlantic coast, the biomass of seagrass peaks in the summer (April–June), yet decreases in the fall (July–September) when storm events usually strike. In tropical areas, vegetation in coastal systems, such as mangroves but also seagrasses, has relatively constant biomass throughout the year, so the coastal protection service is relatively unaffected by seasonal or temporal variability.

The value of some ECE services can also vary spatially (i.e., distance from the shoreline) and temporally (i.e., seasonality). This is of particular importance for recreational and property-related benefits (Coombes et al. 2010, Morgan and Hamilton 2010). A study of home values near Pensacola Beach, Florida, found that Gulf-front property owners were willing to pay an annual tax of \$5807 for a five-year beach nourishment project that would improve access and shoreline views; however, the tax payment declines to \$2770 for a property in the next block, \$2540 for a property two blocks away, and \$1684 for a property three blocks away (Morgan and Hamilton 2010). Models of beach visitors in East Anglia, UK, reveal that seasonal differences are important. For example, school holidays and temperatures have the greatest influence on visitor numbers, and the visitors' propensity to visit the coast increases rapidly at temperatures exceeding 15°C (Coombes et al. 2010). Spatial characteristics that were also associated with more visitors included wide and sandy beaches, beach cleanliness, the presence of a nature reserve, pier, or an urban area behind the beach, and close proximity of an entrance point, car park, and toilet facilities.

Another 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 ECEs 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.

As Moberg and Rönnbäck (2003) describe 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, nutrients, and pollutants by mangroves and seagrasses create the coastal water conditions that favor the 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.

For example, Alongi (2008) suggests 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 aboveground 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 can be seen from Tables 1–5, 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, and sand beaches and 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 enhance the recovery of coral reef fish populations from disturbances due to hurricanes and other violent storms (Mumby and Hastings 2008).

ECE habitats 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 (Layman and Silliman 2002, 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 produce 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.

For example, studies in the Caribbean show that the presence of mangroves and seagrasses enhance 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% to 70% of the catch-per-unit effort harvesting yields for commercially caught species in Queensland, Australia (Meynecke et al. 2008).

There are two ways in which current economic studies of ECE services are incorporating such synergies. One approach is to assess the multiple benefits arising from entire interconnected habitats, such as estuaries. A second method is to allow for the biological connectivity of habitats, food webs, and migration and life-cycle patterns across specific seascapes, such as mangrove–seagrass–reef systems and large marine systems.

For example, Johnston et al. (2002) estimate the benefits arising from a wide range of ecosystem services provided by the Peconic Estuary in Long Island, New York, USA. The tidal mudflats, salt marshes, and seagrass (eelgrass) beds of the estuary support the shellfish and demersal fisheries. In addition, bird-watching and waterfowl hunting are popular activities. Incorporating production function methods, the authors simulate the biological and food web interactions of the ecosystems to assess the marginal value per acre in terms of gains in commercial value for fish and shellfish, bird-watching, and waterfowl hunting. The aggregate annual benefits are estimated to be \$67 per acre for intertidal mud flats, \$338 for salt marsh, and \$1065 for seagrass across the estuary system. Using these estimates, the authors calculate that the asset value per acre of protecting existing habits to be \$12 412 per acre for seagrass, \$4291 for salt marsh, and \$786 for mudflats; in comparison, the asset value of restored habitats is \$9996 per acre for seagrass, \$3454 for marsh, and \$626 for mudflats.

Sanchirico and Mumby (2009) developed an integrated seascape model to illustrate how the presence of mangroves and seagrasses enhance considerably the biomass of coral reef fish communities. A key finding is that mangroves become more important as nursery habitat when excessive fishing effort levels are applied to the reef, because the mangroves can directly offset the negative impacts of fishing effort. Such results support the development of "ecosystem-based" fishery management and the design of integrated coastal-marine reserves that emphasize the importance of conserving and restoring coastal mangroves as nursery sites for reef fisheries (Mumby 2006).

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. This is one important area for future direction of research into ECE services that requires close collaboration between economists, ecologists and other environmental scientists.

Conclusion: Toward a Management Action Plan

Given the rate and scale at which ECEs are disappearing worldwide, assessing and valuing the ecological services of these systems are critically important for improving their management and for designing better policies. Certainly, the various economic values of ECEs should be incorporated into policy decisions that are currently determining the major human drivers of ecological change, such as ecosystem conversion and degradation, resource overexploitation, pollution, and water diversion. As indicated in Figs. 1 and 2, valuation of ECE services is a key step in demonstrating how these human drivers of change alter ecosystem structure and functions, and thus the ecological production of important ecosystem goods and services that benefit human beings.

Yet, as this review has shown, many ECE values are non-marketed. If the aggregate willingness to pay for these benefits is not revealed through market outcomes, then efficient management of such ecosystem services requires explicit methods to measure this social value. Thus, it should not be surprising that the failure to consider the values provided by key ECE services in current policy and management decisions is a major reason for the widespread disappearance of many of these ecosystems and habitats across the globe. Improving the assessment and valuation of ECE services should therefore be a top policy priority for any global management plan for these ecosystems (Granek et al. 2010).

Such a priority is urgent. Our review of five ECEs (i.e., nearshore coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes) reveals that many of the important benefits of these habitats have not been estimated reliably, and even for those services that have been valued, only a few dependable studies have been conducted. Without more efforts to value the key services of ECEs, and to employ these values appropriately in coastal management and planning, slowing the worldwide degradation of coastal and estuarine landscapes will be difficult. Assessing the values of ECE services is critical, 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.

Our review also points to other important policy challenges for improving global management of ECEs. For example, there is now sufficient evidence to suggest that some services, such as coastal protection and habitat–fishery linkages, are not uniform across a coastal seascape. Maintaining ECEs for their multiple and synergistic ecosystem services will also invariably involve managing coastal landscapes across different spatial and temporal scales. 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). How an ecological function, and thus the ecosystem service it supports, varies nonlinearly across a coastal landscape can have important implications for management at the landscape scale for all ECEs (Koch et al. 2009).

Because the connectivity of ECEs across land–sea gradients also influences the provision of certain ecosystem services, management of the entire seascape will be necessary to preserve such synergistic effects. For example, Mumby (2006) argues that the management of ECE habitats in the Caribbean should take into account the life cycle migration of fish between mangroves, seagrass beds, and coral reefs. He recommends that management planning should focus on connected corridors of these habitats and emphasize 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) emphasize 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.

Given the perilous state of many ECEs globally and their critically important benefits, there is clearly a need for a global action plan for protecting and/or enhancing the immediate and longer term values of important ECE services. Such a plan should contain the following features.

First, more interdisciplinary studies involving economists, ecologists, and environmental scientists are required to assess the values of the various ECE services identified in this review for coral reefs, seagrasses, salt marshes, mangroves, and sand beaches and dunes (Tables 1–5). A key priority is to value those services identified in this review for which estimates are currently unavailable or unreliable. Although we know less about the economic benefits of seagrasses and sand beaches and dunes compared to the other ECEs, the number of reliable estimates of almost all services remains woefully inadequate.

Second, destruction of these five critical ECEs for coastal economic development can no longer be viewed as "costless" by those responsible for managing and approving such developments. In particular, the widespread global practice of giving away mangroves, salt marshes, and other ECEs as "free land" for coastal aquaculture, agricultural, and residential development needs to be halted. Especially destructive economic activities, such as dynamite fishing of coral reefs, clearcutting mangroves for wood chips or shrimp farming, mining of sand dunes, extracting seagrasses for shellfish beds, and using salt marshes for landfills, should be banned and the bans enforced. Coastal pollution from aquaculture, tourism activities and infrastructure, agriculture, urban areas and industry need to be monitored, regulated, and where appropriate, taxed.

Third, in many developing countries, the current legal framework and formal institutional structures of ECEs and resource management do not allow local coastal communities any legal rights to establish and enforce control over the ECE goods and services on which the livelihoods of these communities depend. Establishing an improved institutional framework does not necessarily require transferring full ownership of ECE resources to local communities, but could involve co-management by governments and local communities that would allow, for example, the participation of the communities in decisions concerning the long-term management, development and utilization of these resources.

Finally, where appropriate, ecological restoration of key ECEs should be encouraged. However, ecological restoration of these systems is difficult and costly, and requires the right incentives. For example, in Thailand, the full costs of replanting and restoring mangroves in abandoned shrimp ponds is estimated to be around \$9318/ha, which nearly accounts for the entire capitalized value of the restored services of \$12 392/ha (Barbier 2007). This suggests that investors in shrimp farms and other coastal developments that cause widespread mangrove destruction should have the legal requirement to replant mangroves and finance the costs, rather than leaving mangrove restoration solely to governments and local communities. It should be recognized, however, that ex post ecological restoration is no panacea for failed conservation. Such investments are not only costly but risky, and in many cases fall short of recovering the full suite of ecosystem services (Palmer and Filoso 2009). For example, as discussed in the previous section, the Johnston et al. (2002) study of the Peconic Estuary of Long Island found that the asset value of restored salt marsh and seagrass and tidal mudflats in terms of nursery habitat and recreational services were much lower than for conserving the original habitats.

In sum, the more we learn about ECEs and their 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.

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