Progress and Challenges in Valuing Coastal and Marine Ecosystem Services

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Introduction

Coastal and marine environments can begin up to 100 km inland, extend to the continental shelf, and include ocean systems with waters up to 50 m in depth (Millennium Ecosystem Assessment [MA] 2003). The distinct ecosystems found in these environments include estuarine and coastal wetlands such as marshes and mangroves, sand beaches and dunes, sea grass beds, and coral reefs. These ecosystems are some of the most heavily exploited globally. For example, coastal zones make up just 4 percent of the earth's total land area and 11 percent of the world's oceans, yet they contain more than a third of the world's population and account for 90 percent of marine fisheries catch (MA 2005). Coastal populations and their economic activities depend on many services provided by coastal and marine ecosystems (CMEs), such as storm buffering, fisheries production, and enhanced water quality. The location of CMEs in the land–sea interface, or seascape, also suggests a high degree of "interconnectedness" or "connectivity" across these systems and their services. This spatial continuum of heavily populated and exploited ecosystems from coastal to open water has been aptly called a "peopled seascape" (Shackeroff, Hazen, and Crowder 2009).

However, human activities are now threatening many of the world's remaining CMEs and the benefits they provide (Halpern et al. 2008; Lotze et al. 2006; MA 2005; Worm et al. 2009). Human population densities are nearly three times higher in coastal areas than in inland areas, and they are increasing exponentially (United Nations Environment Programme [UNEP] 2006). The degradation and loss of coastal and marine ecosystems are intense and increasing worldwide, with 50 percent of marshes, 35 percent of mangroves, 30 percent of coral reefs, and 29 percent of sea grasses either lost or degraded (Food and Agricultural Organization [FAO] 2007; MA 2005; Orth et al. 2006; UNEP 2006; Valiela et al. 2001; Waycott et al. 2009). Overfishing has been a persistent and growing problem in marine environments (Beddington, Agnew, and Clark 2007; FAO 2009; Jackson et al. 2001; Lotze et al. 2006; Worm et al. 2009).

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The global decline of CMEs affects at least three critical ecosystem services (Halpern et al. 2008; Lotze et al. 2006; MA 2005; Worm et al. 2009): the number of viable (noncollapsed) fisheries; the provision of nursery habitats such as oyster reefs, sea grass beds, and wetlands; and filtering and detoxification services provided by suspension feeders, submerged vegetation, and wetlands. Loss of filtering services is also linked to declining water quality and the increasing occurrence of harmful algal blooms, fish kills, shellfish and beach closures, and oxygen depletion. The decline in biodiversity and ecosystem functions in CMEs may have contributed to biological invasions and vice versa. Increasingly, the loss or change of vegetation in coastal ecosystems has affected these systems' ability to protect against shore erosion, coastal flooding, and storm events (Alongi 2008; Koch et al. 2009).

Given the nature and extent of this degradation and loss of coastal and marine environments and their ecological services, it is important to understand what is at stake in terms of the economic benefits and values of CMEs and their services. This article examines how environmental and resource economics has contributed to our knowledge of CME services, and it documents the progress achieved and the challenges remaining in valuing these services.

The article is organized as follows. The next section presents an overview of the key services and values provided by CMEs. This is followed by a discussion of the main measurement issues and challenges confronting the valuation of CME services. The valuation of key CME ecological services is proving to be important to the management of coasts and oceans. Thus the next section discusses specific cases in which the valuation of key CME services has had an important impact on policy decisions concerning coastal and marine environments, including decisions about coastal land use, the management of reef-related fisheries and dive tourism, the control of coastal pollution, and the creation of marine reserves. This is followed by discussions of two key features of CME benefits: how the spatial variability in some services can greatly influence their economic value and how CME habitats across a seascape can produce cumulative and synergistic benefits. The final section suggests some future directions for economics research on CME services.

Services and Values of Coastal and Marine Ecosystems

To classify the ecosystem services provided by various coastal and marine environments, it is helpful to adopt the broad definition of the Millennium Ecosystem Assessment (MA 2003, p. 53) that "ecosystem services are the benefits people obtain from ecosystems." Thus the term *ecosystem services* is usually interpreted to imply a variety of "benefits," which in economics would normally be classified under three different categories (Barbier 2007): (a) "goods" (e.g., harvests and other products obtained from ecosystems), (b) "services" (e.g., recreation and tourism or the benefits derived from certain ecological regulatory and habitat functions), and (c) cultural benefits (e.g., spiritual and religious beliefs, heritage values).

Regardless of how one defines and classifies ecosystem services, "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values" (Heal et al. 2005, p. 2). One pertinent feature of many CMEs is that they provide multiple benefits, or values. As Table 1 indicates, CME benefits cover a wide variety of use and nonuse values, which can have direct, indirect, and existence and bequest subcomponents.

Table 1 Various values provided by coastal and marine ecosystems

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

Sources: Adapted from Barbier (1994) and Heal et al. (2005, Table 2-1).

Direct Use Values

Typical direct use values, which consist of both consumptive and nonconsumptive uses that involve some direct physical interaction with the ecosystem and its services, include harvesting of fish and wild resources, transportation by waterways, recreation, and tourism. Some unique estuarine, coastal, and marine habitats are also important stores of genetic material and have educational and scientific research value as well. Coral reefs are particularly important in this regard because they are the most structurally complex and taxonomically diverse marine ecosystems, providing habitat for tens of thousands of associated fishes and invertebrates (Jackson et al. 2001). In developing regions, some of the more important uses of CMEs tend to involve both small-scale commercial and "informal" economic activity to support the livelihoods of local populations, for example, through fishing, hunting, fuelwood extraction, and so forth.

Indirect Use Values

As shown in Table 1, CMEs also provide many important indirect use values, such as nutrient retention and cycling, flood control, storm protection, habitat for species, and shoreline stabilization. Such values are considered to be "indirect" because they are derived mainly from the support and protection of economic activities and livelihoods that have directly measurable values (Barbier 1994). For example, in the case of mangrove ecosystems, the mangrove swamps may serve as a nursery and breeding habitat for many important fish species, some of which may migrate as adults to offshore fisheries. In addition, mangroves can reduce the economic damages inflicted by tropical storms on coastal property and communities. Finally, mangrove systems are thought to prevent coastal erosion, thus preserving valuable agricultural land and coastal properties. Such services are difficult to value, yet they stem from critical regulatory and habitat functions of CMEs, such as wave attenuation and dissipation, sediment stabilization, nutrient uptake, nursery and breeding habitat, and water regulation (Barbier 2007; Heal et al. 2005).

Nonuse Values

Many unique coastal and marine environments are considered to have substantial "nonuse values," even in developing regions. These include existence and bequest values, which refer to people benefiting from the knowledge that an ecosystem simply exists or that it will be around for future generations to enjoy. These values may be particularly important among indigenous communities in rural areas because they see their culture, heritage, and traditional knowledge closely intertwined with the surrounding environment. Even some of the poorest rural communities have expressed interest in seeing their way of life passed on to their heirs and future generations (Naylor and Drew 1998; Walters et al. 2008).

In recent years, considerable progress has been made in valuing some of these benefits of CMEs. Before reviewing this progress in more detail, it is useful to examine some of the key measurement issues and challenges confronting the valuation of CME services.

Measurement Issues and Challenges

The literature on ecological services suggests that natural ecosystems such as CMEs are assets that produce a flow of beneficial goods and services over time (Barbier 2007; Heal et al. 2005; MA 2003, 2005; Pagiola, von Ritter, and Bishop 2004; UNEP 2006). In this regard, natural ecosystems are no different from any other asset in an economy, and, in principle, the services provided by CMEs should be valued in a similar manner. That is, whether or not there exists a market for the goods and services produced by ecosystems, their social value must equal the discounted net present value of these flows of goods and services.

There are two advantages to viewing CMEs and other ecosystems as capital assets capable of producing goods and services (Barbier 2011). First, it allows application of the standard tools and analysis developed by natural resource economics for modeling these complex systems. Second, it facilitates a focus on competing uses, such as conservation versus development of a coastal landscape or marine seascape, and the need to account for the value of ecosystem services in order to make efficient choices between these uses.

However, natural ecosystems such as CMEs and their services are unique capital assets that fall in the special category of "nonrenewable resources with renewable service flows" (Just, Hueth, and Schmitz 2004, p. 603). This means that although a coastal and marine ecosystem providing a range of beneficial services, similar to those summarized in Table 1, is unlikely to increase, it can be depleted through habitat destruction, land conversion, pollution impacts, overfishing, and so forth. If the ecosystem is left relatively undisturbed, then the flow services from the ecosystem's regulatory and habitat functions are available in quantities that are not affected by the rate at which they are used.

Whereas the services from most assets in an economy are marketed, the benefits arising from many services provided by CMEs are not. This is particularly true of the indirect use values associated with the regulatory and habitat functions of CMEs, such as nutrient retention and cycling, flood control, storm protection, habitat for species, and shoreline stabilization. If the aggregate willingness to pay for these benefits is not revealed through market outcomes, then efficient management of such ecosystem services requires the use of methods that explicitly measure this social value (e.g., see Freeman 2003; Hanley and Barbier 2009; Heal et al. 2005; and Pagiola, von Ritter, and Bishop 2004).

An additional challenge to measuring the value of the ecological services of CMEs is that their beneficial flows are threatened by the continuing degradation and loss of these systems globally, as discussed in the introduction. The failure to measure explicitly the aggregate willingness to pay for otherwise nonmarketed ecological services exacerbates these issues because the benefits of these services are then "underpriced" in decisions that affect the use, exploitation, and conversion of CMEs. Population, development, and harvesting pressures in many coastal zones and oceans of the world result in increased demand from economic activities, which means that the opportunity cost of maintaining coastal and marine ecosystems is rarely zero. Unless the benefits arising from ecosystem services are explicitly measured, or "valued," then these nonmarketed flows are likely to be ignored in development decisions. Only the benefits of the marketed outputs from economic activities, such as agricultural crops, landed fish catch, urban housing, and other commercial uses of coastal land and oceans, will be taken into account. As a consequence, there will be excessive conversion and overexploitation of CMEs.

A further measurement challenge is the uncertainty about the future values of the services provided by CMEs. It is possible, for example, that the benefits of CME services may increase in the future as more scientific information becomes available about the value of some of the lesser known benefits of these systems (outlined in Table 1). The continued growth in coastal populations also suggests that scarce CME services will become even more valuable in the future. Such a scenario is familiar in environmental and resource economics: if environmental assets are depleted irreversibly through economic development, their value will rise relative to the value of other economic assets (Krutilla and Fisher 1985). Because CMEs are in fixed supply, lack close substitutes, and are difficult to restore, their beneficial services will decline as they are converted or degraded. As a result, the value of ecosystem services is likely to rise relative to other goods and services in the economy. This rising, but unknown, future scarcity value of ecosystem benefits implies an additional "user cost" to any decision that leads to irreversible conversion or depletion of CMEs today.

Uncertainty and irreversible loss are important issues to consider in valuing CME services. However, as emphasized by Heal et al. (2005), the more "fundamental challenge" in valuing these flows is that ecosystem services are largely not marketed, and unless some attempt is made to value the aggregate willingness to pay for these services, the management of natural ecosystems and their services will not be efficient. Thus the remainder of this article focuses on how valuation of the ecological services provided by CMEs can be used in policy decisions about conservation versus development.

Case Studies: Impact of Valuation of CME Services on Policy Decisions

Valuation studies of key CME ecological services have become increasingly important in the management of coasts and oceans. Various methods have been used to estimate the value of the ecological services associated with CMEs. A complete review of all the methods and their real-world applications is beyond the scope of this article.[1] Thus this section highlights selected case studies in which the valuation of CME services influenced important policy decisions concerning the management of coastal and marine environments.

[1]For examples of such reviews applied to various CMEs, see Barbier et al. (2011), Brander et al. (2006), and Heal et al. (2005).

Aquaculture versus Mangrove Ecosystems in Thailand

In Thailand, aquaculture expansion has been associated with mangrove ecosystem destruction. Since 1961, the country has lost from 1,500 to 2,000 km^2 of coastal mangroves, or about 50 to 60 percent of the original area (FAO 2007). Between 1975 and 1996, 50 to 65 percent of Thailand's mangrove deforestation was due to shrimp farm conversion alone (Aksornkoae and Tokrisna 2004). Such widespread loss has focused attention on the role of the two principal services provided by mangrove ecosystems: (a) nursery and breeding habitats for offshore fisheries, and (b) natural barriers to periodic coastal storm events, such as windstorms, tsunamis, storm surges, and typhoons. Mangroves provide a third service to the many coastal communities that exploit them directly for a variety of products, such as fuelwood, timber, raw materials, honey and resins, and crabs and shellfish. Various studies have suggested that in the case of Thailand, the value of these three types of services (i.e., benefits) from mangroves is significant (Barbier 2003, 2007; Sathirathai and Barbier 2001).

For example, Barbier (2007) compares the value in U.S. dollars (1996) of the discounted net returns per hectare to shrimp farming, the costs of mangrove rehabilitation, and the value of mangrove services. Although the commercial profitability of shrimp aquaculture in Thailand provides a substantial incentive for private landowners to invest in such operations, many of the conventional inputs used in shrimp pond operations are subsidized, and below border-equivalent prices, thus artificially increasing the private returns to shrimp farming. Once the estimated subsidies are removed, the discounted net economic returns to shrimp farming over a five-year production period amount to $1,078 to $1,220 per hectare (Barbier 2007). After the five-year period of their productive life, shrimp pond yields decline rapidly and are often abandoned. Across Thailand, those areas with abandoned shrimp ponds degenerate rapidly into wasteland because the soil becomes very acidic, compacted, and too poor in quality to be used for any other productive use such as agriculture. Rehabilitation of the abandoned shrimp farm site requires reestablishing tidal flows, treating and detoxifying the soil, replanting mangrove forests, and maintaining and protecting mangrove seedlings for several years. These restoration costs are considerable, $8,812 to $9,318 per hectare in net present value terms (Barbier 2007). Thus converting mangroves to establish shrimp farms is almost an irreversible land use, and without considerable additional investment in restoration, these areas do not regenerate into mangrove forests.

In contrast, with a combined value ranging from $10,158 to $12,392 per hectare in net present value terms, the three mangrove ecosystem services—coastal protection, wood product collection, and habitat support for offshore fisheries—far exceed the net economic returns to shrimp farming (Barbier 2007). The highest net present value is the storm protection service of mangroves, which is $8,966 to $10,821 per hectare. The net income to local coastal communities from collected forest products and the value of habitat–fishery linkages together total $1,192 to $1,571 per hectare, which is greater than the net economic returns to shrimp farming. However, it is the value of the storm protection service that is critical to the decision concerning whether or not to replant and rehabilitate mangrove ecosystems in abandoned pond areas because it is the storm protection benefit that makes mangrove rehabilitation an efficient land use option.

The Thailand case study demonstrates the importance of valuing ecological services in coastal land use decisions. The irreversible conversion of mangroves for aquaculture results in the loss of ecological services that generate significantly large economic benefits. This loss of benefits should be taken into account in land use decisions that will lead to the widespread conversion of mangroves, but they are often ignored. The most frequently neglected ecological services are those that arise from regulatory and habitat functions, such as coastal storm protection and habitat–fishery linkages, which, in the case of Thailand, appear to provide the largest economic benefits from mangroves.

Storm Protection Value of Mangroves in India

Since the 2004 Indian Ocean tsunami, there has been an increased interest throughout Asia in the storm protection value of mangroves in diminishing the economic damages or deaths associated with major storm events. For example, two studies of the 1999 cyclone that struck Orissa, India, found that 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). Statistical analysis indicates that there would 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 (Das and Vincent 2009). Losses incurred per household were greatest (US$154) in a village that was protected by an embankment but had no mangroves, whereas losses per household were significantly lower (US$33) in a village that was protected only by mangrove forests (Badola and Hussain 2005).

Direct and Indirect Use Values of Mangroves to Coastal Communities

A number of studies have estimated the various direct and indirect use values of mangroves to local coastal communities (Barbier and Strand 1998; Brander, Florax, and Vermaat 2006; Chong 2005; Hammitt, Liu, and Liu 2001; Naylor and Drew 1998; Othman, Bennett, and Blamey 2004; Rönnbäck, Crona, and Ingwall 2007; Ruitenbeek 1994; Walters et al. 2008; Walton et al. 2006). As in the case of Thailand, these studies reveal that significant nonmarket benefits to local communities from habitat–fishery linkages and collection or harvesting of mangrove products are threatened by the continuing destruction of local mangrove ecosystems. There is also evidence that coastal communities, including those in poor economies, hold important nonuse values associated with mangroves. For example, a contingent valuation study of mangrove-dependent coastal communities in Micronesia demonstrated that the communities "place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves' marketable products" (Naylor and Drew 1998, p. 488).

Valuation of Coral Reef Benefits

Coral reefs support near-shore fisheries harvested by coastal communities, supply the global aquarium trade, and provide valuable shoreline protection (Cesar and van Beukering 2004; Chong 2005; Rodwell et al. 2003; Wilkinson et al. 1999; Zeller, Booth, and Pauly, 2007). Coral reefs also provide recreational benefits, including ecotourism, diving and snorkeling, and sport fishing (Brander et al. 2007; Cesar and van Beukering 2004; Mathieu, Langford, and Kenyon 2003; Tapsuwan and Asafu-Adjaye 2008). However, unless they are managed carefully for these multiple benefits, coral reefs can easily become overexploited and damaged.

Reef-related fishery benefits in the U.S. state of Hawaii are estimated to be $1.3 million per year (Cesar and van Beukering 2004). Between 1982 and 2002, small-scale, predominantly coral reef, fisheries contributed a total of $54.7 million to the economies of America

Samoa and the Commonwealth of the Northern Mariana Islands (Zeller, Booth, and Pauly 2007). Small fishes, invertebrates, and corals are also collected from reefs for the live animal aquarium trade. Global aquarium trade has grown rapidly in recent decades, from \$20 to \$40 million per year in 1985 (Wood 1985) to an estimated \$90 to \$300 million per year^{-} in 2002 (Sadovy, Vincent, and Sale 2002). However, reliable estimates of values for the sustainable production of coral reef fish for local consumption and the aquarium trade are rare. White, Vogt, and Arin (2000) estimate that the potential annual revenue from sustainable fish production in the Philippines could be \$15,000 to \$45,000 per square kilometer of healthy coral reef for local consumption and \$5,000 to \$10,000 per square kilometer for live fish export. Zhang and Smith (2011) estimate the maximum sustainable yield to the Gulf of Mexico reef fishery (mainly grouper and snapper species, amberjack, and tilefish) to be approximately 2.86 million pounds per month, although the reefs in the Gulf are generally exposed limestone or sandstone rather than coral.

The tourism activities associated with coral reefs can be substantial. However, a recent meta-analysis of studies estimating the recreational value of reefs found substantial bias and variation if key socioeconomic characteristics and site choices are not taken into account (Brander, Van Beukering, and Cesar 2007). If such biases are controlled, more reliable estimates can be made. For example, Mathieu, Langford, and Kenyon (2003) conducted a stated preference study to determine tourists' willingness to pay (WTP) to visit marine national parks in the Seychelles that contain coral reefs. They found that the average consumer surplus per tourist was \$2.20, resulting in a total estimated consumer surplus of \$88,000 for the 40,000 tourists visiting the Seychelles in 1997. The authors found that country of origin, reasons for visiting the parks, and expectations concerning the visit were more important in determining WTP responses than sociodemographic factors such as age, sex, education, and income. In addition, respondents indicated a much higher WTP for certain parks (e.g., Curieuse and Ile de Coco) than for others (Baie Terney). Tapsuwan and Asafu-Adjaye (2008) estimated the economic value of scuba diving in the Similan Island coral reefs in Thailand, controlling for divers' 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 that the value of the consumer surplus per person per dive trip was \$3,233.

An important ecosystem service provided by coral reefs is buffering shorelines from severe weather, thus protecting coastal human populations, property, and economic activities. Unfortunately, existing economic estimates of this value tend to be unreliable because they tend to use benefit transfer and replacement cost methods of valuation in an ad hoc manner (Chong 2005). However, isolated events indicate that the value of storm protection services could be substantial. For example, as a result of the 1998 bleaching event in the Indian Ocean, the expected annual loss in property values from declining reef protection is estimated to be \$174 per hectare (Wilkinson et al. 1999).

Valuation of Benefits from Saltwater Marshes

Saltwater marshes provide a wide range of ecosystem services, including recreational benefits, storm and shoreline protection, support for offshore fisheries, and absorption of pollutants (see Brander, Florax, and Vermaat 2006 for a review). Early studies focused on the coastal habitat–fishery linkages of salt marshes in the southeastern United States (Bell 1997; Ellis and Fisher 1987; Freeman 1991; Lynne, Conroy, and Prochaska 1981; Swallow 1994). For example, for the east and west coasts of Florida, the capitalized value of an acre of salt marsh in terms of recreational fishing is estimated to be $6,471 and $981, respectively (Bell 1997), and 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 per acre (Freeman 1991). However, for the estuarine peat-bog wetlands on the Pamlico Sound, North Carolina, the support for an offshore shrimp fishery provided by the wetland did not prove to be sufficiently valuable to justify reversing development decisions that could eliminate that service (Swallow 1994). The study found that the loss of normal quality wetlands reduced shrimp fishery values by $277 per square kilometer, but that protection of these wetlands could not be justified on such grounds alone because the economic value of increased shrimp production would be less than the value of coastal agricultural development.

However, various studies have also shown that the values estimated for improved coastal and marine habitat quality for fisheries, whether through protection of the nursery and breeding habitat service of coastal wetlands or reduction of nutrient pollution, are influenced by the market conditions and regulatory policies that affect harvesting decisions in the fishery, in particular whether it operates under conditions of open access or optimal management (Barbier and Strand 1998; Barbier, Strand, and Sathirathai 2002; Freeman 1991; Smith 2007). Thus one should be cautious about basing coastal development decisions solely on single services such as habitat–fishery linkages when open access conditions may distort the impact of conserving coastal wetlands on increased fishery production and returns.

In addition, as with other CMEs, salt marshes provide benefits other than fish habitat and nurseries, and it may be important to consider these benefits when deciding whether to convert or conserve these coastal wetlands. For example, an analysis of the economic damages associated with thirty-four major U.S. hurricanes since 1980 found that coastal wetlands explained 60 percent of the variation in relative damages inflicted on coastal communities (Costanza et al. 2008). More specifically, the additional storm protection value per unit area of coastal wetlands from a specific hurricane ranges from a minimum of US$23 per hectare for Hurricane Bill to a maximum of US$463,730 per hectare for Hurricane Opal, with a median value of just under US$5,000 per hectare. Another study found that in southern Louisiana, the treatment of wastewater predominantly by marsh swamps achieved capitalized cost savings of $785 to $15,000 per acre relative to conventional municipal treatment (Breaux, Farber, and Day 1995).

Salt marshes also support a variety of recreational and other uses. For example, estimates from land sales and leases for marshes in England suggest prices in the range of £150 to £493 per acre for bird shooting and wildfowling (King and Lester 1995). Respondents were willing to pay £31.60 per person to create otter habitat and £1.20 to protect birds in the Severn Estuary Wetlands bordering England and Wales (Birol and Cox 2007). In the Ribble estuary on England's west coast, annual net income from grazing in a salt marsh nature reserve is estimated at £15.27 per hectare (King and Lester 1995).

Valuation of the Impacts of Coastal Pollution and Degradation on CMEs

Valuation studies have also indicated how CMEs are vulnerable to problems of coastal pollution and degradation (Gren, Elogsson, and Jannke 1997; Kahn and Kemp 1985; Knowler and Barbier 2005; Knowler, Barbier, and Strand 2001; Lipton 2004; Smith 2007; Smith and Crowder 2005; Stephenson and Shabman 1998). For example, Smith and Crowder (2005) showed how nitrogen loading and the subsequent hypoxia (low dissolved oxygen) in a North Carolina estuary affect a blue crab fishery both directly and via its prey. They found that a 30 percent reduction in nitrogen loading in the estuary could increase net present value fishery rents by US$2.56 million. But Smith (2007) showed that the benefits from improving environmental quality in the estuary are small relative to the benefits from improving management in the open access fishery. Coastal pollution in the Chesapeake Bay has been shown to impact aquatic habitats and water quality, thus reducing the value of recreational and fishing benefits and residential housing (Kahn and Kemp 1985; Leggett and Bockstael 2000; Lipton 2004). The result has been a range of initiatives to control water pollution (Boesch and Goldman 2009; Stephenson and Shabman 1998). For example, cap-and-trade allowances for point source pollution have been implemented in Virginia and Pennsylvania, and in Maryland, a US$2.50 per household per month fee is levied to finance a US$460 million per year enhanced nutrient removal system for publicly owned wastewater treatment systems.

Nonmarket Values of Marine Reserves and Protected Areas

There has also been interest in assessing the wider nonmarket values of marine reserves and protected areas, especially because the costs of closing fisheries are often substantial. Some reserves may create an important "insurance value" for commercial fisheries through increasing the stability of underlying marine ecosystems (Clark 1996; Grafton and Kompas 2005; Grafton, Kompas, and Van Ha 2009; Greenville and MacAulay 2007; Sumaila 2002). For example, analysis of the Northeast Atlantic cod fishery shows that creation of marine reserves can protect the discounted economic rent of the fishery if the habitat faces a "shock" caused by something such as climate change or a pollution event (Grafton, Kompas, and Van Ha 2009; Sumaila 2002). As discussed earlier, the economic value of dive tourism in marine reserves can be substantial (Mathieu, Langford, and Kenyon 2003; Tapsuwan and Asafu-Adjaye 2008). Even in temperate zones, a marine reserve may reduce profits for a commercial fishery while increasing an important ecotourism benefit such as seal watching (Boncoeur, Ifremer, and Ifremer 2002). Similarly, a survey of day visitors to sandy beaches or adjacent rocky habitats in Orange County, California, estimated that the benefits of more effective enforcement of marine-protected areas to avoid coastal ecosystem decay were $6 per family visit, or $3.6 to $4.8 million per coastline (Hall, Hall, and Murray 2002).

Spatial Variability of CME Services

Studies of CMEs are increasingly finding that the ecological functions underlying some services vary spatially or temporally, thus influencing the economic benefits they provide (Aburto-Oropeza et al. 2008; Barbier et al. 2008; Koch et al. 2009; Peterson and Turner 1994).

For example, it has been found that wave attenuation by coral reefs, sea grass beds, salt marshes, mangroves, and sand dunes provides protection against wind and wave damage from coastal storm and surge events, but that the magnitude of protection varies spatially across these habitats (Barbier et al. 2008; Koch et al. 2009). For salt marshes and mangroves, wave attenuation diminishes as the habitat moves further inland (i.e., its distance from the shoreline increases). In the case of near-shore coral reefs and sea grasses, wave attenuation is a function of the water depth above the reef or grass bed, which also varies spatially. Wave attenuation by sand beaches and dunes displays exponentially increasing protection at higher elevations or where dune vegetation cover is greatest.

There is also evidence that the nursery and breeding habitat function of coastal wetlands is greater at the seaward edge, or "fringe," of the coastal habitat than further inland. For example, Peterson and Turner (1994) compared "fringe" and interior marshes in Louisiana and found that the densities of most fish and crustaceans were highest in salt marshes within 3 m of the water's edge. They also found that in the Gulf of California, Mexico, the mangrove fringe with a width of 5 to 10 m had the most influence on the productivity of near-shore fisheries, with a median value of $37,500 per hectare. Fishery landings have also been found to increase with the length of the mangrove fringe in a given location (Aburto-Oropeza et al. 2008).

Economic studies are beginning to take into account the spatial variability of these key CME services. For example, Barbier et al. (2008) showed that how much of a mangrove forest coastline should be converted to shrimp aquaculture may depend critically on the spatial variability of coastal storm protection across the mangrove landscape. Mangroves are unlikely to stop storm waves that are greater than 6 m in height, and where mangroves do attenuate waves, the wave height of a storm decreases quadratically for each 100 m that a mangrove forest extends out to sea (Alongi 2008; Barbier et al. 2008; Mazda et al. 1997). When Barbier et al (2008) used this nonlinear wave attenuation function to reestimate the value of the storm protection service for the Thailand case study, they found that the storm protection service of mangroves still dominates all values, but that small losses in mangroves will not cause a precipitous fall in the economic benefits of storm buffering by mangroves. This means that the aggregate value across all uses of the mangroves, shrimp farming, and ecosystem values is at its highest ($17.5 million) when up to 2 km^2 of mangroves are allowed to be converted to shrimp aquaculture and the remainder of the ecosystem is preserved. Thus, accounting for the nonlinear relationship between storm protection and mangroves changes the economically efficient coastal land use decision.

Since the seminal contribution to the bioeconomic literature by Clark (1976), economists have sought to develop spatial models of marine ecosystem management in a variety of contexts, including determining optimal harvesting and marine reserve establishment (Brown and Roughgarden 1997; Costello and Polasky 2008; Sanchirico 2005; Sanchirico and Wilen 1999, 2002, 2005; Smith, Sanchirico, and Wilen 2009; Smith and Wilen 2003). As noted by Grafton, Kompas, and Schneider (2005), including spatial variability in such models has two key implications. First, fishers tend to reallocate their effort to areas that generate higher relative rents, resulting in the creation of spatial "economic gradients" that might differ from the "biological gradients" generated by larval export and adult migration. Second, including the impact of spatial variation on net revenues may be particularly important in determining the location of marine reserves. Ignoring this spatial effect tends to result in predicted rises in fish stocks that are greater than the actual increases associated with establishing a reserve. For example, in the California sea urchin fishery, the response of divers to profits earned in different locations was found to be an important determinant of whether reserve creation leads to higher net benefits (Smith and Wilen 2003).

Synergies across Seascapes

As noted in the introduction, one unique feature of CMEs is that they occur at the interface between watersheds, the coast, and the oceans. The location of CMEs in the land–sea interface, or seascape, leads to a high degree of interconnectedness or connectivity across these systems and their services, which may considerably enhance the ecosystem service provided by one single habitat. For example, many fish and shellfish species use mangroves and sea grass beds as nursery grounds, and they eventually migrate to coral reefs as adults, only to return to the mangroves and sea grasses to spawn (Moberg and Rönnbäck 2003; Mumby 2006; Mumby et al. 2004). Similar synergies exist for salt marshes, sea grass beds, and near-shore marine systems in estuaries (Rountree and Able 2007; Shackeroff, Hazen, and Crowder 2009).

There are two ways in which current economic studies of CME services are incorporating such synergies. The first approach is to assess the multiple benefits arising from entire interconnected habitats, such as estuaries. The 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–sea grass–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 on Long Island, New York. The tidal mudflats, salt marshes, and sea grass (eelgrass) beds of the estuary support the shellfish and demersal (seabed) fisheries. In addition, bird watching and waterfowl hunting are popular activities. The authors incorporate production function methods to simulate the biological and food web interactions of the ecosystems in order 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 mudflats, $338 for salt marsh, and $1,065 for sea grass across the estuary system. Using these estimates, the authors calculate the asset value of protecting existing habitats to be $12,412 per acre for sea grass, $4,291 for salt marsh, and $786 for mudflats; in comparison, the asset value of restored habitats is $9,996 per acre for sea grass, $3,454 for marsh, and $626 for mudflats.

Sanchirico and Mumby (2009) developed an integrated seascape model to illustrate that the presence of mangroves and sea grasses considerably enhances 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. The results of this study support the development of "ecosystem-based" fishery management and the design of integrated coastal-marine reserves that emphasize four key priorities: (a) the relative importance of mangrove nursery sites, (b) the connectivity of individual reefs to mangrove nurseries, (c) areas of nursery habitat that have an unusually large importance to specific reefs, and (d) priority sites for mangrove restoration projects (Mumby 2006).

Economic studies of large marine ecosystems have extended simple single-species or predator–prey harvesting models to consider multispecies relationships and their impacts on harvesting the commercially valuable species from the ecosystem (Finnoff and Tschirhart 2003a, 2003b; Flaaten and Stollery 1996; Sanchirico, Smith, and Lipton 2008; Wacker 1999). These studies find that the interactions between noncommercial species in marine ecosystems and the harvesting of commercial fish species are highly important. For example, the results for a marine ecosystem in the East Bering Sea off Alaska indicate that predator–prey relations between Orcas (killer whales) and Stellar sea lions are intricately linked to the stock density of pollack, the sea lions' commercially harvested prey (Finnoff and Tschirhart 2003a, 2003b). Orcas also feed on otter, which in turn prey on sea urchins. This complex food web implies that changes in any one population will impact all other species. Such a result suggests that the noncommercial species have value because they support the fisheries indirectly via the effect of total diversity on the productivity of commercial fisheries and overall ecosystem stability. Similarly, Sanchirico, Smith, and Lipton (2008) demonstrate that if total allowable catch levels for Chesapeake Bay fisheries accounted for species interdependencies and other environmental factors, fishing revenues would increase and their variability could be reduced.

Larval dispersal and fish migration between reserve and fishery areas may be important factors in determining whether marine reserves enhance the productivity of neighboring fishery areas (Grafton, Kompas, and Schneider 2005; Grafton, Quentin, and Van Ha 2009; Rodwell et al. 2002; Sanchirico 2005; Sanchirico and Wilen 2002; Smith, Zhang, and Coleman 2008). In tropical systems, the quality of critical habitat, such as coral reefs, is also relevant. Evidence from Kenya indicates that coral reefs may also be critical to larval dispersal to fishing areas, which could influence the effectiveness of marine reserves and closed fishing grounds in inducing stock recovery and thus eventual reopening to fishing (Rodwell et al. 2003).

Conclusions and Directions for Future Research

CMEs are disappearing at a rapid rate and on an alarming scale worldwide. Yet many of the benefits of CME habitats are undervalued or even ignored in coastal and marine development decisions. It is critically important to measure the value of the ecological services provided by these systems so we can better understand what is at stake if these CME habitats are lost and to incorporate these values into coastal and marine management and planning. This article has sought to contribute to the economics literature concerning ecological services by reviewing the current state of knowledge about CME services, documenting the progress made as well as the challenges remaining in the economic valuation of these services, and demonstrating how valuing the ecological services provided by CMEs can influence policy decisions concerning the management of coastal and marine environments.

Maintaining CMEs for their multiple and synergistic ecosystem services invariably involves managing seascapes across different spatial scales. As this article has indicated, for a number of key ecosystem services, such as coastal protection, productivity of marine systems, and habitat–fishery linkages, there is now sufficient evidence to suggest that these services are not uniform across a seascape. Incorporating spatial and synergistic characteristics of CMEs into management scenarios is likely to result in the most ecologically and economically sustainable management plan possible. Because the connectivity of CMEs across land–sea gradients has implications for the provision of certain ecosystem services, management of the entire seascape may be necessary to preserve such synergistic effects.

As scientific knowledge concerning how CMEs provide services has evolved in recent decades, so too has the economic valuation of these services. This evolution is reflected, for example, in the economic analysis of habitat–fishery linkages, where early efforts focused on including wetland habitat, such as marshes and mangroves, as an environmental input into fisheries production but examined only the impacts on current harvest, prices, and consumer and producer surpluses (Bell 1997; Ellis and Fisher 1987; Freeman 1991; Lynne, Conroy, and Prochaska 1981; Sathirathai and Barbier 2001). These studies demonstrated that, in cases where biological growth of fisheries is affected, it is necessary to incorporate the dynamic impacts of habitat–fishery linkages on future stocks and market outcomes. Initial efforts in this area focused solely on the long-run equilibrium impacts of changes in coastal habitat (Barbier 2003; Barbier and Strand 1998; Barbier, Strand, and Sathirathai 2002; Swallow 1994) or water quality (Kahn and Kemp; Smith and Crowder 2005). More recent valuations have examined the impacts on intertemporal changes in rents and market outcomes (Barbier 2007; Smith 2007).

Future research is likely to focus on estimating habitat–fishery linkages for the multiple benefits arising from entire interconnected habitats, such as estuaries and mangrove–sea grass–reef systems (Johnston et al. 2002; Sanchirico and Mumby 2009); accounting for the spatial variability of habitat–fishery linkages whether for coastal wetlands or pollution (Aburto-Oropeza et al. 2008; Peterson and Turner 1994; Smith 2007); and estimating habitat–fishery linkages as joint products with other services provided by interconnected seascapes (Moberg and Rönnbäck 2003; Shackeroff, Hazen, and Crowder 2009).

As we learn more about other key CME services and the ecological functions underlying them, we should expect economic analysis and valuation of these benefits to evolve as well. Thus this field will likely continue to provide fertile ground for environmental and resource economics research for some years to come.

References

Aburto-Oropeza, O., E. Ezcurra, G. Danemann, V. Valdez, J. Murray, and E. Sala. 2008. Mangroves in the Gulf of California increase fishery yields. *Proceedings of the National Academy of Sciences* 105: 10456–59.

Aksornkoae, Sanit, and Ruangrai Tokrisna. 2004. Overview of shrimp farming and mangrove loss in Thailand. In *Shrimp farming and mangrove loss in Thailand*, ed. Edward B. Barbier and Suthawan Sathirathai. London: Edward Elgar.

Alongi, Daniel M. 2008. Mangrove forests: Resilience, protection from tsunamis and responses to global climate change. *Estuarine, Coastal and Shelf Science* 76 (1): 1–13.

Badola, Ruchi, and S. A. Hussain. 2005. Valuing ecosystems functions: An empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. *Environmental Conservation* 32 (1): 85–92.

Barbier, Edward B. 1994. Valuing environmental functions: tropical wetlands. *Land Economics* 70 (2): 155–73.

———. 2003. Habitat-fishery linkages and mangrove loss in Thailand. *Contemporary Economic Policy* 21 (1): 59–77.

———. 2007. Valuing ecosystem services as productive inputs. *Economic Policy* 22 (49): 177–229.

———. 2011. *Capitalizing on nature: Ecosystems as natural assets*. Cambridge: Cambridge University Press.

Barbier, Edward B., Sally D. Hacker, Chris Kennedy, Evamaria W. Koch, Adrian C. Stier, and Brian R. Silliman. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81 (2): 169–93.

Barbier, Edward B., Evamaria W. Koch, Brian R. Silliman, Sally D. Hacker, Eric Wolanski, Jurgenne H. Primavera, Elise F. Granek, Stephen Polasky, Shankar Aswani, Lori A. Cramer, David M. Stoms, Chris J. Kennedy, David Bael, Carrie V. Kappel, Gerardo M. Perillo, and Denise J. 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. *Science* 319 (5861): 321–23.

Barbier, Edward B., and Ivar Strand. 1998. Valuing mangrove-fishery linkages: A case study of Campeche, Mexico. *Environmental and Resource Economics* 12 (2): 151–66.

Barbier, Edward B., Ivar Strand, and Suthawan Sathirathai. 2002. Do open access conditions affect the valuation of an externality? Estimating the welfare effects of mangrove-fishery linkages in Thailand. *Environmental and Resource Economics* 21 (4): 343–67.

Beddington, John R., D. J. Agnew, and Colin W. Clark. 2007. Current problems in the management of marine fisheries. *Science* 316 (5832): 1713–16.

Bell, Frederick W. 1997. The economic value of saltwater marsh supporting marine recreational fishing in the Southeastern United States. *Ecological Economics* 21 (3): 243–54.

Birol, K., and V. Cox. 2007. Using choice experiments to design wetland management programmes: The case of the Severn Estuary Wetland, UK. *Journal of Environmental Planning and Management* 50 (3): 363–80.

Boesch, Donald F., and Erica B. Goldman. 2009. Chesapeake Bay, USA. In *Ecosystem-based management for the oceans*, ed. Karen McLeod and Heather Leslie, 268–93. Washington, DC: Island Press.

Boncoeur, J., F. Alban, O. G. Ifremer, and O. T. Ifremer. 2002. Fish, fishers, seals and tourists: Economic consequences of creating a marine reserve in a multi-species, multi-activity context. *Natural Resource Modeling* 15 (4): 387–411.

Brander, Luke M., J. G. Raymond, M. Florax, and Jan E. Vermaat. 2006. The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature. *Environmental and Resource Economics* 33 (2): 223–50.

Brander, Luke M., Peter J. H. Van Beukering, and Herman S. J. Cesar. 2007. The recreational value of coral reefs: A meta-analysis. *Ecological Economics* 63 (1): 209–18.

Breaux, A., S. Farber, and J. Day. 1995. Using natural coastal wetlands systems for wastewater treatment: an economic benefit analysis. *Journal of Environmental Management* 44 (3): 285–91.

Brown, Gardner, Jr., and J. Roughgarden. 1997. A metapopu1ation model with a common pool. *Ecological Economics* 22 (1): 65–71.

Cesar, Herman S. J., and Peter J. H. van Beukering. 2004. Economic valuation of the coral reefs of Hawai'i. *Pacific Science* 58 (2): 231–42.

Chong, Jo. 2005. *Protective values of mangrove and coral ecosystems: A review of methods and evidence.* Gland, Switzerland: IUCN.

Clark, Colin W. 1976. *Mathematical bioeconomics.* New York: John Wiley.

———. 1996. Marine reserves and precautionary management of fisheries. *Ecological Applications* 6 (2): 369–70.

Costanza, R., O. Pérez-Maqueo, M. L. Martinez, P. Sutton, S. J. Anderson, and K. Mulder. 2008. The value of coastal wetlands for hurricane protection. *Ambio* 37 (4): 241–48.

Costello, Christopher, and Stephen Polasky. 2008. Optimal harvesting of stochastic spatial resources. *Journal of Environmental Economics and Management* 56 (1): 1–18.

Das, Saudamini, and Jeffrey R. Vincent. 2009. Mangroves protected villages and reduced death toll during Indian super cyclone. *Proceedings of the National Academy of Sciences* 106 (40): 7357–60.

Ellis, Greg M., and Anthony C. Fisher. 1987. Valuing the environment as input. *Journal of Environmental Management* 25 (2): 149–56.

Finnoff, David, and John T. Tschirhart. 2003a. Protecting an endangered species while harvesting its prey in a general equilibrium ecosystem model. *Land Economics* 79 (2): 60–80.

———. 2003b. Harvesting in an eight-species ecosystem. *Journal of Environmental Economics and Management* 45 (3): 589–611.

Flaaten, Olli, and K. Stollery. 1996. The economic costs of biological predation: Theory and application to the case of the North Atlantic minke whale's (*Balaenoptera Acutorostrata*) consumption of fish. *Environmental and Resource Economics* 8 (1): 75–95.

Food and Agricultural Organization (FAO) of the United Nations. 2007. *The world's mangroves 1980–2005.* FAO Forestry Paper 153. Rome: FAO.

———. 2009. *The State of World Fisheries and Aquaculture 2008.* Rome: FAO .

Freeman, A. Myrick III. 1991. Valuing environmental resources under alternative management regimes. *Ecological Economics* 3 (3): 247–56.

———. 2003. *The measurement of environmental and resource values: Theory and methods.* 2nd ed. Washington, DC : Resources for the Future.

Grafton, R. Quentin, and Tom Kompas. 2005. Uncertainty and the active adaptive management of marine reserves. *Marine Policy* 29 (5): 471–79.

Grafton, R. Quentin, Tom Kompas, and Viktoria Schneider. 2005. The bioeconomics of marine reserves: A selected review with policy implications. *Journal of Bioeconomics* 7 (2): 161–78.

Grafton, R. Quentin, Tom Kompas, and Pham Van Ha. 2009. Cod today and none tomorrow: The economic value of a marine reserve. *Land Economics* 85 (3): 454–69.

Greenville, Jared, and T. Gordon MacAulay. 2007. Untangling the benefits of protected areas in fisheries. *Marine Resource Economics* 22 (3): 267–85.

Gren, Inge-Marie, K. Elofsson, and P. Jannke. 1997. Cost-effective nutrient reductions to the Baltic Sea. *Environmental and Resource Economics* 10 (4): 341–62.

Hall, Darwin C., Jane V. Hall, and S. N. Murray. 2002. Contingent valuation of marine protected areas: Southern California rocky intertidal ecosystems. *Natural Resource Modeling* 15 (3): 335–68.

Heal, Geoffrey M., Edward B. Barbier, Kevin J. Boyle, Alan P. Covich, Stephen P. Gloss, Carl H. Hershner, John P. Hoehn, Catherine M. Pringle, Stephen Polasky, Kathleen Segerson, and Kirstin Shrader-Frechette. 2005. *Valuing ecosystem services: Toward better environmental decision making.* Washington, DC: National Academies Press.

Halpern, Benjamin S., Shaun Walbridge, Kimberly A. Selkoe, Carrie V. Kappel, Fiorenza Micheli, Caterina D.' Agrosa, John F. Bruno, Kenneth S. Casey, Colin Ebert, Helen E. Fox, Rod Fujita, Dennis Heinemann, Hunter S. Lenihan, Elizabeth M. P. Madin, Matthew T. Perry, Elizabeth R. Selig, Mark Spalding, Robert Steneck, and Reg Watson. 2008. A global map of human impact on marine ecosystems. *Science* 319 (5865): 948–52.

Hammitt, James K., Jin-Tan Liu, and Jin-Long Liu. 2001. Contingent valuation of a Taiwanese wetland. *Environment and Development Economics* 6 (2): 259–68.

Hanley, Nick, and Edward B. Barbier. 2009. *Pricing nature: Cost-benefit analysis and environmental policy-making*. London: Edward Elgar.

Jackson, Jeremy B. C., Michael X. Kirby, Wolfgang H. Berger, Karen A. Bjørndal, Louis W. Botsford, Bruce J. Bourque, Roger H. Bradbury, Richard Cooke, Jon Erlandson, James A. Estes, Terence P. Hughes, Susan Kidwell, Carine B. Lange, Hunter S. Lenihan, John M. Pandolfi, Charles H. Peterson, Robert S. Steneck, Mia J. Tegner, and Robert R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293 (5530): 629–38.

Johnston, Robert J., Thomas A. Grigalunas, James J. Opaluch, Marisa Mazzotta, and Jerry Diamantedes. 2002. Valuing estuarine resource services using economic and ecological models: The Peconic Estuary System. *Coastal Management* 30 (1): 47–65.

Just, Richard E., Darrell L. Hueth, and Andrew Schmitz. 2004. *The welfare economics of public policy: A practical approach to project and policy evaluation.* Cheltenham, UK: Edward Elgar.

Kahn, James R., and W. Michael Kemp. 1985. Economic losses associated with the degradation of an ecosystem: The case of submerged aquatic vegetation in the Chesapeake Bay. *Journal of Environmental Economics and Management* 12 (3): 246–63.

King, S. E., and J. N. Lester. 1995. The value of salt marsh as a sea defence. *Marine Pollution Bulletin* 30 (3): 180–89.

Knowler, Duncan, and Edward B. Barbier. 2005. Managing the Black Sea anchovy fishery with nutrient enrichment and a biological invader. *Marine Resource Economics* 20 (3): 263–85.

Knowler, Duncan, Edward B. Barbier, and Ivar Strand. 2001. An open-access model of fisheries and nutrient enrichment in the Black Sea. *Marine Resource Economics* 16 (3): 195–217.

Koch, Evamaria W., Edward B. Barbier, Brian R. Silliman, Denise J. Reed, Gerardo M. E. Perillo, Sally D. Hacker, Elise F. Granek, Jurgenne H. Primavera, Nyawira Muthiga, Stephen Polasky, Benjamin S. Halpern, Chris J. Kennedy, Carrie V. Kappel, and Eric Wolanski. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment* 7 (1): 29–37.

Krutilla, J. V., and A. C. Fisher. 1985. *The economics of natural environments: Studies in the valuation of commodity and amenity resources.* Washington, DC: Resources for the Future.

Leggett, Christopher G., and Nancy E. Bockstael. 2000. Evidence of the effects of water quality on residential land prices. *Journal of Environmental Economics and Management* 39: 121–44.

Lipton, Douglas W. 2004. The value of improved water quality to Chesapeake Bay boaters. *Marine Resource Economics* 19 (2): 265–70.

Lotze, Heike K., Hunter S. Lenihan, Bruce J. Bourque, Roger H. Bradbury, Richard G. Cooke, Matthew C. Kay, Susan M. Kidwell, Michael X. Kirby, Charles H. Peterson, and Jeremy B. C. Jackson. 2006. Depletion, degradation and recovery potential of estuaries and coastal seas. *Science* 312 (5781): 1806–9.

Lynne, Gary D., Patricia Conroy, and F. J. Prochaska. 1981. Economic value of marsh areas for marine production processes. *Journal of Environmental Economics and Management* 8 (2): 175–86.

Mathieu, Lawrence F., Ian H. Langford, and Wendy Kenyon. 2003. Valuing marine parks in a developing country: A case study of the Seychelles. *Environment and Development Economics* 8: 373–90.

Mazda, Y., M. Magi, M. Kogo, and P. N. Hong. 1997. Mangroves as a coastal protection from waves in the Tong King Delta, Vietnam. *Mangroves and Salt Marshes* 1 (2): 127–35.

Millennium Ecosystem Assessment (MA). 2003. *Ecosystems and human well-being: A framework for assessment.* Washington, DC: Island Press.

———. 2005. *Ecosystems and human well-being: Current state and trends.* Washington, DC: Island Press.

Moberg, Frederick, and Patrick Rönnbäck. 2003. Ecosystem services of the tropical seascape: interactions, substitutions and restoration. *Ocean and Coastal Management* 46 (1): 27–46.

Mumby, Peter J. 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. *Biological Conservation* 128 (2): 215–22.

Mumby, Peter J., A. J. Edwards, J. E. Arias-Gonzalez, K. C. Lindeman, P. G. Blackwell, A. Gall, M. I. Gorczynska, A. R. Harborne, C. L. Pescod, H. Renken, C. C. C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of reef fisheries in the Caribbean. *Nature* 427 (6974): 533–36.

Naylor, Rosalind, and M. Drew. 1998. Valuing mangrove resources in Kosrae, Micronesia. *Environment and Development Economics* 3 (4): 471–90.

Othman, Jamal, Jeff Bennett, and Russell Blamey. 2004. Environmental management and resource management options: A choice modelling experience in Malaysia. *Environment and Development Economics* 9 (6): 803–24.

Orth, Robert J., Tim J. B. Carruthers, William C. Dennison, Carlos M. Duarte, James W. Fourqurean, Kenneth L. Heck, Jr., A. Randall Hughes, Gary A. Kendrick, W. Judson Kenworthy, Suzanne Olyarnik, Frederick T. Short, Michelle Waycott, and Susan L. Williams. 2006. A global crisis for seagrass ecosystems. *BioScience* 56 (12): 987–96.

Pagiola, Stefano, Konrad von Ritter, and Joshua T. Bishop. 2004. *How much is an ecosystem worth? Assessing the economic value of conservation.* Washington, DC: The World Bank.

Peterson, G. W., and R. E. Turner. 1994. The value of salt marsh edge versus interior as habitat for fish and decapods crustaceans in a Louisiana tidal marsh. *Estuaries* 17 (1): 235–62.

Rodwell, Lynda D., Edward B. Barbier, Callum M. Roberts, and Timothy R. McClanahan. 2002. A model of tropical marine reserve-fishery linkages. *Natural Resource Modeling* 15 (4): 453–486.

———. 2003. The importance of habitat quality for marine reserve-fishery linkages. *Canadian Journal of Fisheries and Aquatic Sciences* 60 (2): 171–81.

Rönnbäck, Patrick, Beatrice Crona, and Lisa Ingwall. 2007. The return of ecosystem goods and services in replanted mangrove forests: Perspectives from local communities in Kenya. *Environmental Conservation* 34 (4): 313–24.

Rountree, R. A., and K. W. Able. 2007. Spatial and temporal habitat use patterns for salt marsh nekton: Implications for ecological functions. *Aquatic Ecology* 41 (1): 25–45.

Ruitenbeek, H. J. 1994. Modeling economy-ecology linkages in mangroves: Economic evidence for promoting conservation in Bintuni Bay, Indonesia. *Ecological Economics* 10 (3): 233–47.

Sadovy, Y. J., A. C. J. Vincent, and P. F. Sale. 2002. Ecological issues and the trades in live reef fishes. In *Coral reef fishes: Dynamics and diversity in a complex ecosystem*, ed. P. F. Sale, 391–420. San Diego: Academic Press.

Sanchirico, James N. 2005. Additivity properties in metapopulation models: Implications for the assessment of marine reserves. *Journal of Environmental Economics and Management* 49 (1): 1–25.

Sanchirico, James N., and Peter J. Mumby. 2009. Mapping ecosystem functions to the valuation of ecosystem services: implications of species-habitat associations for coastal land-use decisions. *Theoretical Ecology* 2 (2): 67–77.

Sanchirico, James N., Martin D. Smith, and Douglas W. Lipton. 2008. An empirical approach to ecosystem-based fishery management. *Ecological Economics* 64 (3): 586–96.

Sanchirico, James N., and James E. Wilen. 1999. Bioeconomics of spatial exploitation in a patchy environment. *Journal of Environmental Economics and Management* 37: 129–50.

———. 2002. The impacts of marine reserves on limited-entry fisheries. *Natural Resource Modeling* 15 (August): 291–310.

———. 2005. Optimal spatial management of renewable resources: matching policy scope to ecosystem scale. *Journal of Environmental Economics and Management* 50 (1): 23–46.

Sathirathai, Suthawan, and Edward B. Barbier. 2001. Valuing mangrove conservation, Southern Thailand. *Contemporary Economic Policy* 19 (2): 109–22.

Shackeroff, Janna M., Elliott L. Hazen, and Larry B. Crowder. 2009. The oceans as peopled seascapes. In *Ecosystem-based management for the oceans*, ed. Karen McLeod and Leslie, 33–54. Washington, DC: Island Press.

Smith, Martin D. 2007. Generating value in habitat-dependent fisheries: The importance of fishery management institutions. *Land Economics* 83 (1): 59–73.

Smith, Martin D., and Larry B. Crowder. 2005. Valuing ecosystem services with fishery rents: A lumped-parameter approach to hypoxia in the Neuse River estuary. Working paper, Fondazione Eni Enrico Mattei, Nota di Lavora 115.2005.

Smith, Martin D., N. James Sanchirico, and James E. Wilen. 2009. The economics of spatial-dynamic processes: Applications to renewable resources. *Journal of Environmental Economics and Management* 57 (1): 104–21.

Smith, Martin D., Junie Zhang, and Felicia C. Coleman. 2008. Econometric modeling of fisheries with complex life histories: Avoiding biological management failures. *Journal of Environmental Economics and Management* 55 (3): 265–80.

Smith, Martin D., and James E. Wilen. 2003. Economic impacts of marine reserves: The importance of spatial behavior. *Journal of Environmental Economics and Management* 46 (2): 183–206.

Stephenson, K., and L. Shabman. 1998. Watershed-based effluent trading: The non-point source challenge. *Contemporary Economic Policy* 26 (4): 412–21.

Sumaila, U. R. 2002. Marine protected area performance in a model of a fishery. *Natural Resource Modeling* 15 (4): 439–51.

Swallow, Stephen K. 1994. Renewable and nonrenewable resource theory applied to coastal agriculture, forest, wetland and fishery linkages. *Marine Resource Economics* 9 (December): 291–310.

Tapsuwan, Sorada, and John Asafu-Adjaye. 2008. Estimating the economic benefit of scuba diving in the Similan Islands, Thailand. *Coastal Management* 36 (5): 431–42.

United Nations Environment Programme (UNEP). 2006. Marine and coastal ecosystems and human well-being: A synthesis report based on the findings of the Millennium Ecosystem Assessment. Nairobi: UNEP.

Valiela, Ivan, Jennifer L. Bowen, and Joanna K. York. 2001. Mangrove forests: one of the world's threatened major tropical environments. *BioScience* 51 (10): 807–15.

Wacker, H. 1999. Optimal harvesting of mutualistic ecological systems. *Resource and Energy Economics* 21 (1): 89–102.

Walters, Bradley B., R.önnbäck Patrick, John M. Kovacs, Beatrice Crona, Syed Ainul Hussain, Ruchi Badola, Jurgenne H. Primavera, Edward B. Barbier, and Farid Dahdouh-Guebas. 2008. Ethnobiology, socio-economics and management of mangrove forests: a review. *Aquatic Botany* 89 (2): 220–36.

Walton, Mark E., M. Giselle, P. B. Samonte-Tan, Jurgenne H. Primavera, G. Gareth Edwards-Jones, and Lewis Le Vay. 2006. Are mangroves worth replanting? The direct economic benefits of a community-based reforestation project. *Environmental Conservation* 33 (4): 335–43.

Waycott, Michelle, Carlos M. Duarte, Tim J. B. Carruthers, Robert J. Orth, William C. Dennison, Suzanne Olyarnik, Ainsley Calladine, James W. Fourqurean, Kenneth L. Heck, Jr., A. Randall Hughes, Gary A. Kendrick, W. Judson Kenworthy, Frederick T. Short, and Susan L. Williams. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* 106: 12377–81.

White, A. T., H. P. Vogt, and T. Arin. 2000. Philippine coral reefs under threat: The economic losses caused by reef destruction. *Marine Pollution Bulletin* 40 (7): 598–605.

Wilkinson, C., O. Linden, H. Cesar, G. Hodgson, J. Rubens, and A. E. Strong. 1999. Ecological and socioeconomic impacts of 1998 coral mortality in the Indian Ocean: An ENSO impact and a warning of future change? *Ambio* 28 (2): 188–96.

Wood, E. M. 1985. Exploitation of coral reef fishes for the aquarium fish trade. Marine Conservation Society. Rosson-Wye, UK.

Worm, Boris, Ray Hilborn, Julia K. Baum, Trevor A. Branch, Jeremy S. Collie, Christopher Costello, Michael J. Fogarty, Elizabeth A. Fulton, Jeffrey A. Hutchings, Simon Jennings, Olaf P. Jensen, Heike K. Lotze, Pamela M. Mace, Tim R. McClanahan, C.óilín Minto, Stephen R. Palumbi, Ana M. Parma, Daniel Ricard, Andrew A. Rosenberg, Reg Watson, and Dirk Zeller. 2009. Rebuilding global fisheries. *Science* 325 (5940): 578–85.

Zeller, D., S. Booth, and D. Pauly. 2007. Fisheries contributions to the gross domestic product: Underestimating small-scale fisheries in the Pacific. *Marine Resource Economics* 21 (4): 355–74.

Zhang, J., and M. D. Smith. 2011. Estimation of a generalized fishery model: A two-stage approach. *Review of Economics and Statistics* 93 (2): 690–99.