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TITLE: The value of estuarine and coastal ecosystem services

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ABSTRACT:

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Silliman Department of Biology, University of Florida, Gainesville, Florida 32611 USASearch for more papers by this author First published: 01 May 2011 https://doi.org/10.1890/10-1510.1 Corresponding Editor: A. M. Ellison. 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. 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). 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.? Figure 1Open in figure viewerPowerPoint Key interrelated steps in the valuation of ecosystem goods and services. This figure is adapted from NRC (2005: Fig. 1-3). 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$), we do not think such a transformation is warranted for the purposes of this study. Second, we do not alter the original valuation estimates into a common unit of measure (such as US$·ha?1·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 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. Figure 2Open in figure viewerPowerPoint 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). 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. Table 1. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for nearshore coral reefs. Table 5. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for sand beaches and dunes. 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 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 mutualisti

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