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Valuation of Marine Ecosystems

Achilleas Vassilopoulos

Phoebe Koundouri

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Summary

Water accounts for more than 70% of Earth's surface, making marine ecosystems the largest and most important ecosystems of the planet. However, the fact that a large part of these ecosystems and their potential contribution to humans remains unexplored has rendered them unattractive for valuation exercises. On the contrary, coastal zones, , being the interface between the land, the sea, and human activities competing for space and resources, have been extensively studied with the objective of marine ecosystem services valuation. Examples of marine and coastal ecosystems are open oceans, coral reefs, deep seas, hydrothermal vents, abyssal plains, wetlands, rocky and sandy shores, mangroves, kelp forests, estuaries, salt marshes, and mudflats. Although there are arguments that no classification can capture the ways in which ecosystems contribute to human well-being and support human life, very often policymakers have to decide upon alternative uses of such natural environments. Should a given wetland be preserved or converted to agricultural land? Should a mangrove be designated within the protected areas system or be used for shrimp farming? To answer these questions, one needs first to establish the philosophical basis of value within the ecosystems framework. To this end, two vastly different approaches have been proposed. On the one hand, the nonutilitarian (biocentric) approach relies on the notion of intrinsic value attached to the mere existence of a natural resource, independent of whether humans derive utility from its use (if any) or preservation. Albeit useful in philosophical terms, this approach is still far from providing unambiguous and generally accepted inputs to the tangible problem of ecosystem valuation. The utilitarian (anthropocentric) perspective, on the other hand, assumes that natural environments have value to the extent that humans derive utility from placing such value. According to the total economic value (TEV) approach, this value can be divided into "use" and "nonuse." Use values involve some interaction with the resource, either directly or indirectly, while nonuse values are derived simply from the knowledge that natural resources and aspects of the natural environment are maintained. Existence and altruistic values fall within this latter category.

Not surprisingly, economists have long revealed a strong preference for the utilitarian approach. As a result, the valuation of marine ecosystems requires that we understand the ecosystem services they deliver and then attach a value to the services. But what tools are available to economists when valuing marine ecosystems? For the most part, ecosystem services are not traded in formal markets and thus actual prices are usually not available. Valuation techniques essentially seek different ways to estimate measures like Willingness To Pay (WTP), Willingness To Accept (WTA), or expenditures and costs. The techniques used for the valuation of ecosystem services can be divided into three main families: market-based, revealed preference, and stated preference. Finally, value-transfer methods are also used when estimates of value are available in similar contexts. All these methods have advantages and disadvantages, with different methods being suitable for different situations. Hence, extra caution is required during the design and implementation of valuation attempts.

Keywords

marine ecosystems, coastal ecosystems, ecosystem services, market valuation, nonmarket valuation

Marine and coastal environments include a number of ecosystems that provide a wide range of ecosystem services to human society. These ecosystem services are the cornerstone of environmental valuation, since they constitute the link between the ecosystem functions and the utility of individuals. According to the Millennium Ecosystem Assessment (MEA; 2005b), ecosystem services can be classified into supporting, regulating, cultural, and provisioning. Supporting services are those that enable the production of other services, such as nutrient cycling by deep-sea organisms. Regulating services are those that ecosystems provide by acting as regulators of ecosystem processes, such as wastewater treatment. Finally, provisioning and cultural services are those related to the material and nonmaterial products obtained from ecosystems, respectively. The former includes fish and other resources, while services like recreation in sea parks or aesthetic experiences derived from a surface or subsurface landscape fall into the latter category.

In environmental valuation, the main concern is changes in the marine and coastal environment that result in changes in the delivery of ecosystem services. Accordingly, alternative policy measures and human actions that involve various environmental variables can be converted into monetary units and compared while opting for optimal resource allocation or accurate estimation of environmental degradation. Unless all the above ecosystem services are taken into account, the value of the ecosystems will be underestimated, leading to policy measures that tilt toward competing uses of resources and eventually promoting their degradation.

The Total Economic Value (TEV) framework is increasingly used to assess the value of ecosystem services because it helps to ensure that components of value are not omitted or double counted when multiple valuation methods are employed. TEV assumes that individuals hold multiple values for ecosystems pertaining to use and nonuse aspects, and it aims to capture the full value of different natural resources. The range of values recognized by TEV can be categorized in three groups. Use values are derived from the contribution of an environmental asset to the current production or consumption, through direct use (e.g., seafood, sailing, diving, or recreational angling) or from the support services of the natural environment that are "indirectly used" (such as air purification or regulation of water flows). Nonuse values, in contrast, include existence values that are derived from knowing that resources will not cease to exist, independently of any actual or prospective use by the individual. At last, option and bequest values reflect the value people place on their own and heirs' future ability to use a resource, respectively. There is no full agreement yet on whether option and bequest values should be classified as use or nonuse values. For example, for three different treatments of option and bequest values, the interested reader may refer to UNEP (2007), Pagiola, Bishop, & Von Ritter (2004), and Ledoux & Turner (2002).

Ecosystem Types

Among the different distinctions between marine and coastal areas (for a range of definitions, see Kay & Alder, 1999), the one followed by the MEA reports (2005a) is probably the fittest, when it comes to environmental valuation. According to the MEA definition, the ocean and coastal realm is divided into two major systems: the coastal systems and the marine fisheries systems.

MEA (2005a) defined coastal ecosystems as areas between 50 meters below mean sea level and 50 meters above the high tide level (or extending landward to a distance of 100 km, whichever is closer to the sea), where land-based influences dominate. They included many different types of ecosystems, such as mangroves, estuaries, coral reef systems, beaches, salt marshes, rocky shores, kelp forests, and seagrass beds, as well as coastal waters of the continental shelves. Coastal ecosystems deliver various types of provisioning (e.g., food, fiber, fuel, medicines), regulating (e.g., nutrient cycling, flood and erosion control), supporting (e.g., nursery and habitat for several species), and cultural (e.g., recreational fishing, diving, beach visiting) services. In particular, estuaries, salt marshes, mangroves, and seagrasses provide many benefits, such as supporting the shellfish and demersal fisheries; maintaining hydrological balance; protection from floods, hurricanes, and excessive erosion; and provision of water purification, habitat for birds, food, fuel wood, building materials, carbon sequestration, and further opportunities for tourism/recreation. Sand beaches and dunes provide recreational space to humans and food to both humans and migratory birds, land-based nutrients to the near-shore coastal system, carbon sequestration, raw materials, and coastal erosion protection. Coral reefs are sources of a considerable part of the protein intake of millions of people who depend on fish catches from these areas (Salvat, 1992). They also support biodiversity, contribute to the diving tourism industry, help in the prevention of beach erosion, and provide pharmaceutical compounds and construction materials. Finally, coastal systems support biodiversity by being nurseries and homes to many marine mammals, turtles, reptiles, and seabirds.

Anthropogenic factors have been intervening in the delivery of ecosystem services. Land use has caused deforestation and has affected agrochemical disposal; communities and industries increasingly exploit fisheries or other extracted materials, while demographic, tourism, and urbanization trends lead to increasing coastal populations and higher demand for coastal areas, which in turn implies overdevelopment and excessive pollution. Also, hazards like oil spills and other related accidents may have a major impact on coastal systems. In terms of climate change, coastal ecosystem services are mostly affected by temperature and sea level alterations, ocean acidification, floods, and droughts.

Marine systems are characterised as waters from the low water mark (50 meters deep) to the high seas that support marine capture fisheries and deepwater habitats (MEA, 2005a). As a result, marine systems entail parts of all marine system biomes (Longhurst, Sathyendranath, Platt, & Caverhill, 1995): the coastal boundary zone (except shelves of 50 meters or shallower), trade winds, westerlies, and polar easterlies. These ecosystems receive much less attention than coastal environments, because they are remote and there are many challenges associated with studying their environment directly. Marine systems support water circulation and CO_2 exchange, nutrient cycling, and highly diverse habitats and species. They also play a significant role in mitigating the climatic changes caused by anthropogenic emissions and in facilitating the freshwater cycle, as well as in providing pharmaceutical compounds, recreation, aesthetic experiences, and inspiration for art. Finally, they can be an important source of economic development, with fisheries, hydrocarbon, gas, and oil extraction, as well as with mining of nonnutritional materials.

Coastal development and land-based pollution unavoidably affect marine ecosystems, especially the coastal biome. For the rest of the biomes, overexploitation of deep-sea fisheries, overhunting of marine mammals, seabird mortalities, ocean dumping of waste, and oil spills seem to be the main problems. Climate change phenomena that are most relevant to marine ecosystems are sea surface temperature, changes in salinity, wave climate, and ocean circulation (McLean et al., 2001).

Valuation Methods

In general, economic valuation seeks to monetize the benefits that ecosystems provide to humans. Valuation techniques result in value estimates—e.g., willingness to pay (WTP) or willingness to accept (WTA)—using market and nonmarket information related to the ecosystem that is subject to valuation. Although market-based methods provide a straightforward link between ecosystem services and money value (directly or through production functions), relevant markets are scarce in most cases. As a result, nonmarket valuation techniques have gained popularity in valuation of marine and coastal ecosystems. The tools available to environmental economists when valuing such ecosystems, in conjunction with the TEV frameworks and MEA classification, constitute a road map that facilitates the selection of valuation methods in empirical analysis.

Market Valuation

Market-based methods rely on data from markets that are related to the use of ecosystem services. The market price method can be used to value changes in either the quantity or quality of a good or service, but since only a few environmental goods/services are bought and sold in the markets, the method has limited coverage. For example, market values may not be found for a big part of the marine raw materials exploited. However, based on well-established economic principles, even if such services are not directly marketed, inferences can be based on prices and purchased quantities of other goods or services that are complements of the resource with regard to consumption.

Prices of fisheries and aquaculture for example can readily capture the direct-use values of provisioning services. Aside from data availability, the limitation of the market-based approach is that market imperfections and other distortions may affect the efficacy of prices (either direct or proxy) in measuring underlying values. The reasons for this may be spatial and seasonal variation of prices, illegal catches, further revenue, and employment in food processing that is not accounted for, etc.

Another technique of market valuation is the productivity or production-function approach (also referred to as the net factor). The approach is used for the estimation of values related to ecosystems whose services are used, along with other inputs, in the production of marketed goods.i A necessary condition for the use of this method is an appropriately specified production function that indicates the contribution of the environmental assets as inputs to the final output, from which information from the benefit attributed to these inputs may be deduced. This method requires the collection of data on how changes in the quantity or quality of the environmental resource affect the costs of production for the final good, the demand for and supply of the final good, and the demand for and supply of other factors of production. Nonetheless, changes in the availability, quantity, and quality of ecosystem services may also affect the market price of the final good, or the prices of any other inputs, creating an overwhelmingly complicated system of nonlinear relationships, since such treatments exhibit very high model dependence, and inaccuracies in modeling will carry over to the values. Also, this method is restricted to ecosystems that are related to the production of marketed goods.

Nonmarket Valuation

Nonmarket valuation techniques can be further divided into revealed and stated preference methods. Revealed preference methods rely on observations of real choices, which are then interpreted within a predefined theoretical framework. Stated preference methods are based on hypothetical answers and choices people make in survey questionnaires.

Revealed Preference

Revealed preference techniques include the travel cost (TC), random utility, and hedonic property pricing methods. The TC method is usually used to estimate values derived from recreation. It relies on data concerning the frequency and cost associated with visits to recreation sites in order to calculate the demand for a site and to infer the welfare loss resulting from closing it, reducing accessibility to it, or raising the entrance fee. Since people face both budget and time constraints, the calculation of consumer surplus requires data for the distance travelled (to estimate the actual cost) and for wage rates (to adjust for opportunity costs). Although single-site valuation can provide reliable estimates, it falls short as the number of substitutes becomes larger. For this reason, multiple-site models have been developed; in these models, the demand for related sites is estimated simultaneously through systems of equations, which allows the presence of cross-price effects. Nevertheless, due to the use of aggregate data, the results of such exercises do not readily provide information on the value of a change in quality of recreation, while important determinants of value might be confounded. A more elegant approach is the estimation of the models using microdata. Microdata are not always easy to collect, while at the same time they require much more complicated analysis (e.g., zero-truncated Poisson or negative binomial models) to account for nonrepresentativeness of the sample (missing data from nonvisitors, overrepresentation of frequent users, etc.). Nevertheless, even when microdata are available, such models cannot readily provide information on the marginal value of changes in the quality of ecosystem services.

For this reason, the use of random utility models has prevailed in valuing changes in site characteristics that affect the delivery of ecosystem services. According to random utility theory, the deterministic part of the utility of visiting a site can be decomposed into a vector of observed characteristics related to the site. Visitors' observed choices among recreation sites can be analyzed under this framework, by making certain assumptions regarding the joint distribution of the vector of random error component, as well as the functional form of the deterministic utility function. The conditional logit (CL), proposed by McFadden (1974), is the most well known and widely applied model, since it yields a closed and readily interpretable formula for choice probabilities. Individual characteristics can also be accommodated in CL as demand shifters. Aside from data on observed choices regarding site visits, the CL requires a listing of all ecosystem services types and levels delivered by the competing sites.

Finally, the hedonic pricing technique relies on transactions occurring in the housing market to infer the value of key underlying ecosystem services, through the part of the price differential that is explained by the proximity to specific environmental amenities. Calculation of welfare measures requires assumptions on the functional form of the hedonic price function, testing for spatial dependence between prices at different locations and collecting data on house prices as well as on the amount and type of ecosystem services within a given radius of each property.

Stated Preference

Although revealed preference methods have the advantage of relying on actual market behavior, it is not always the case that such markets exist, and even if they do, they cannot capture the nonuse values that are crucial under the TEV framework. Stated preference techniques are able to fill this gap. Using hypothetical scenarios, they provide the possibility not only of measuring the nonuse values of environmental amenities but also of isolating the variables of interest and studying their interactions. Stated preference methods include survey-based techniques, such as the contingent valuation (CV), and other attribute-based methods, such as choice experiments (CE).

Ever since the importance of the CV has been recognized by the National Oceanic and Atmospheric Administration (NOAA),[ii] the CV method has been by far the most popular in estimating reliable WTP or WTA values for public goods or services. CV surveys involve a hypothetical valuation scenario in which consumers are asked to respond to questions related to their maximum WTP or WTA for upgrades or downgrades (or both) in the quality of various ecosystem services. Elicitation formats can vary from simple referendums to open answers or choices from multiple price lists.

Attribute-based methods are based on the elicitation of people's preferences for attributes. The attributes refer to characteristics of the environment that affect the delivery of certain ecosystem services or the ecosystem services per se. By far the most widely used attribute-based method for marine ecosystems is choice experiments (CE), which presents respondents with different choices, varying in the attributes of interest, and asks them to choose the one they prefer or the best and worst alternative. The choices are converted to values using the random utility framework analyzed for revealed preferences. In fact, the only difference here is that alternatives are fictitious and choices are stated rather than observed. Of course, the CE is not the only, and certainly not the first, attribute-based method. Conjoint analysis, a concept adapted from psychology, is an alternative method that is based on the theory of conjoint measurement and involves the ranking or rating of alternatives (or of the attributes per se), which is then converted to values. Although conjoint analysis is widely used in economic applications, Louviere, Flynn, and Carson (2010) showed that it is inappropriate for economic evaluation and thus it should be used with caution in economic applications.

The main potential drawbacks of stated preference methods are hypothetical and informational bias, as well as scope insensitivity and protest bids. As a result, stated preference methods should be designed and implemented carefully, aiming at mitigating and/or eliminating biases in the best possible way.

Hypothetical bias (also known as "yeah-saying" in referendum-format surveys) refers to the phenomenon of willingness to pay values that are inflated relative to the true ones, due to the fact that valuation questions are hypothetical, so there is no cost for exaggerating one's responses. Scope insensitivity refers to the fact that respondents are not willing to pay more for a greater quantity of a particular good or service, thus violating basic economic theory. The idea originated with Kahneman (1986), who argued that respondents' WTP for cleaning up all of the lakes in the province of Ontario was not much larger than their WTP for cleaning up the lakes of a much smaller part of the province. Information bias may arise when respondents are faced with unfamiliar attributes (such as those of deep-sea ecosystems) and therefore are highly affected by the amount and type of information presented to them. Finally, protest bids are often registered by respondents who, for ethical or other reasons, refuse to pay or to respond. Aside from the decision to exclude or include the protest bids in the analysis, their presence may bias valuation results in a direction that is indeterminate a priori (Halstead, Luloff, & Stevens, 2002).

Other Methods

An alternative to preference-based methods of valuing environmental goods involves quantifying replacement, restoration, or avoided costs. This approach assigns values to ecosystems based on the cost of artificial substitutes or of restoring the degraded ecosystems to ensure delivery of an ecosystem service or to avoid damages that would result from the deterioration of an ecosystem. Although the values assigned do not result in a reliable measure of benefits, but rather in actual or forgone expenditures, they can prove very useful if damage avoidance/replacement expenditures have actually been made or in serving as a benchmark for estimates from other valuation exercises.

Another method of valuation is the value or benefits transfer technique. The estimation of economic values for ecosystem services in this case is based on primary research results that come from previous studies in similar contexts. These results are extrapolated in cases where primary data are absent and infeasible to collect (e.g., due to budget or time constraints). Value-transfer methods can be divided into three categories: unit value transfer (with or without adjustments for differences in income and price levels), value function transfer (using an estimated value function from an individual primary study), and meta-analytic function transfer (using a value function estimated from the results of multiple primary studies). According to Brander (2013), the key challenges of conducting accurate and credible value transfers are to account for important differences, to measure and communicate the level of uncertainty regarding a transferred value, to engage different stakeholders in the value-transfer process, and to communicate the results clearly.

Literature Review

Several results from the literature on valuation of coastal and marine ecosystems are presented here. (All papers included in the review are presented in Table 1.) Marine protected areas and fauna are presented separately, since they usually refer to both coastal and marine ecosystems. The focus is on the variety of methods, ecosystem types, and regions, and not on comprehensiveness; as a result, the literature review cannot by any means be characterized as exhaustive. Also, monetary values are reported as they appear in the original studies, so currencies and real values differ between cases. Thus, readers should refer to the original articles before they attempt any comparison among the values presented in the review. Interested readers are encouraged to consider other reviews on the topic as well, such as those of Barbier et al. (2011) on valuation studies of estuarine and coastal ecosystems, Brander et al. (2006) on papers regarding wetlands valuation, Heal et al. (2005) on research on aquatic and related terrestrial ecosystems, and Remoundou et al. (2009) on valuation tasks in the Mediterranean and Black Sea region regarding various ecosystem services.

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Coastal Systems

The literature reveals that coastal ecosystems are by far the most widely studied system. Of these, coral reefs, mangroves, and beaches seem to have drawn the most attention so far. An example of coral reef valuation is the study by Ahmed, Umali, Chong, Rull, and Garcia (2007), who used the TC method to estimate the value of recreational benefits of coral reefs along the Lingayen Gulf, Bolinao, Philippines. The authors surveyed 92 respondents on site and gathered data on the respondents' demographics, reasons for visiting the site, demand for recreation, purpose of trip, number of persons included in the trip, length of stay, number of visits made to the recreation site, point of origin, and expenditures incurred during the trip. The data allowed the estimation of a demand function for recreation that revealed a consumer surplus of around $223 per person per year. Another example of coral reef valuation is Parsons and Thur's (2008) study that estimated the economic value of coral reef quality changes in the Caribbean, using a CE survey mailed to 211 scuba divers who visited the Bonaire National Marine Park in 2001. Divers were asked how they might have altered their trip choice had the quality of the coral reef system been different from what they experienced. The quality of coral reefs (attributes of the CE) was defined in terms of visibility, species diversity, and percent coral cover. The study's results indicated that for modest changes in quality, the annual losses would be about $45 per person, while for larger changes, they would be an estimated $192 per person. Finally, Bishop et al. (2011) employed a hybrid model to value the Hawaiian coral reef ecosystem protection/restoration (introduction of no-fishing zones) and repair of localized damage (repairs of injuries from ship strikes). Their survey had the form of conjoint ranking, with respondents ranking their preferred choice among one program for MPA expansion, one for ship-strike repairs, one for both, and the status quo (i.e. no program); the cost was presented in the form of annual federal taxes. The model was called hybrid since no experimental design was employed and the alternatives were the same for all respondents, a fact that distinguishes it from other conjoint-ranking surveys and makes it more similar to a CV survey with multiple valuation tasks. The results were based on two Internet samples that were recruited in different ways: through the American National Election Study (*N* = 2,335) and through Stanford University's Faceto-Face Recruited Internet Survey Panel (*N* = 942). The estimated mean WTP for substantial protection and restoration of degraded main Hawaiian Islands ecosystems was $224.81 per household per year; for restoration of reefs after localized injuries, it was $62.82 per household per year; while the WTP for achieving both goals combined was $287.62 per household per year.

In the valuation of mangroves, the production function approach is most prominent. Sathirathai (1998) used this approach to value changes in Surat Thani (zone three in the Gulf of Thailand) mangrove area under alternatively open access and managed fishery conditions. In particular, Sathirathai assumed that supporting services of mangrove areas in providing offshore fisheries through serving as breeding grounds and nurseries were inputs to the production of demersal fish, crab, and shrimp. Using data across all five zones of the Gulf of Thailand over the 1983 to 1993 time period and assuming that harvesting of these types of fisheries is a Cobb–Douglas function of the level of fishing effort and mangrove area, he ended up with value estimates in the range of $33 to $110 per hectare change in mangrove area, depending on the magnitude of demand elasticity and whether the fisheries are open access or managed. Barbier and Strand (1998) used the same method to value the effects of mangrove deforestation in the Laguna de Terminos, through the mangroves' role in supporting the shrimp fishery of Campeche, Mexico. Using actual price data on shrimp catches over 1980 to 1990 for the Campeche fishery, the researchers found that the value of the mangrove habitat was affected by the level of exploitation, while on average over this period, a marginal (in km^2) decline in mangrove area produced a loss of nearly $140,000 per year in revenues from the Campeche fishery.

Estuarine-marsh areas were the subject of research by Lynne, Conroy, and Prochaska (1981), who developed a bioeconomic model for the areas' valuation through the production function of a blue crab fishery on the Gulf Coast of Florida. To model fish population dynamic relationships, the authors chose a stock adjustment model and included a one-year lag for marsh availability. Using published catch-effort data over the period of 1952 to 1974 and marsh area measurements from air photos on intermittent dates during the period, they concluded that at the mean level of effort (33,000 traps), the total present value of a marsh acre in blue crab production is $3.00 per acre (with a 10% capitalization rate). Sale, Hosking, and Du Preez (2009) conducted a CV survey in two South African estuaries, Kowie and Kromme, using a sample of 150 recreational users in order to estimate WTP for securing an increase in the freshwater inflow into the estuaries so as to maintain or improve the environmental service flows provided. The survey followed the payment card format and was conducted on each estuary site with personal interviews; the cost was presented as an additional annual levy to those who would benefit from the improvement. The values of freshwater inflows into the Kowie and the Kromme Estuaries were calculated at R0.072/m^3 and R0.013/m^3, respectively.

Regarding the recreational values of coastal waters and beaches, both revealed and stated preferences can be found in the literature. Nunes and Van (2004) monetized the loss of cultural ecosystem services (beach visits, swimming, sailing) in North European waters brought about by the harmful algal-bloom invasive exotic species that were primarily introduced through ballast water of ships. They used the TC method to measure recreation benefits derived from the prevention of such species, based on microdata collected from visitors at Zandvoort, a famous Dutch beach resort. Besides monthly income and other demographics, all of the 242 respondents were asked about their means of travelling to the beach. For those travelling by car, additional information was obtained with regard to the postal code of their address, the brand and model of their car, the size of the engine, the type of fuel used, and the parking fees at the beach. Based on this information, calculation of individual travel costs was possible, and this figure was added to the time cost (estimated by multiplying the amount of time that a respondent spent on the two-way trip by the value of time, which varied according to respondent's monthly income). Several instance data were also recorded (e.g., weather, day of the week, etc.). The results indicated a recreational value of €55 per individual per year. Eggert and Olsson (2009) conducted a CE concerning the recreational values and the value of various biodiversity levels of coastal waters on the west coast of Sweden, Skagerrak and Kattegat. The sample consisted of 324 participants from Västra Götalands and Hallands län, in the southwest part of Sweden. The questionnaire was mailed to participants and the attributes of the CE design were cod stock level, bathing water quality, and biodiversity level, while the payment vehicle was a common fee to be collected for one year. The investigators found a mean WTP for improved bathing water and cod stock, high biodiversity, and avoiding lower biodiversity of SEK 639, 1330, 667, and 1348 per household, respectively. The CE approach was also used by Hynes, Tinch, and Hanley (2013) to estimate the economic benefits attached to a number of parameters of coastal water-quality improvements, based on face-to-face interviews conducted with a sample of 365 recreationalists on beaches on the west coast of Ireland. The attributes employed in the experimental design were benthic health, human health risks, and beach debris. Based on the results, the mean WTP for a program that largely improves benthic health, reduces health risks from 5% to virtually 0%, and changes beach debris management from prevention only to prevention and collection was estimated at €6.78 per recreationalist per beach visit. In one of the very few studies employing the hedonic pricing method, Gopalakrishnan, Smith, Slott, and Murray (2011) modeled the price of residential property as a function of property characteristics, beach quality characteristics, distance from oceanfront, and width of the beach at the property location. Using data on residential property in ten coastal towns of North Carolina and combining geomorphologic models of coastal erosion with hedonic coastal property valuations, they found that the value of an average oceanfront property would increase by $8,800 with an additional foot of beach width; the magnitude of this effect was nearly five times that estimated from a model that failed to account for dynamic ecological feedbacks ($1,440). Castaño-Isaza, Newball, Roach, and Lau (2015), using the CV method, elicited the WTP of 1,793 tourists to San Andres Island to support the protection of the island's beaches. The annual consumer surplus from the implementation of the protection program was found to be around $997,468, while the potential annual revenue reduction from beach erosion was estimated at $73 million. Finally, in the limited literature studying the effects of climate change, Remoundou, Diaz-Simal, Koundouri, and Rulleau (2015) elicited respondents' preferences for different climate change mitigation approaches in the Santander Bay area. The study used the CE methodology and addressed biodiversity of shells, birds and invertebrates, number of days of beach closure due to jellyfish blooms, beach size (recreation), and annual municipal tax on drinking and waste water. A sample of 80 Santander Bay residents responded to the survey and the results showed a WTP value of €162.95 (€147.72), €99.04 (€119.06), and €73.68 for moving away from the status quo, toward medium (high) biodiversity, reduction of closure days by 5 (10), and high recreation experience, respectively.

Aside from recreation, citizen's preferences on a number of other values related to the eutrophication of the Baltic Sea was the subject of a study by Markowska and Żylicz (1999), who attempted to provide an efficient model of Baltic Sea ecosystem services delivery. To do so, they derived value estimates related to eutrophication reduction using CV surveys in three countries that were considered representative for three subregions: Sweden for Western Europe market economies, Lithuania for the former Soviet Union republics, and Poland. In Poland the valuation experiment included an open-ended (WTP = $14/capita, *N* = 820) and two discrete choice surveys administered face to face (WTP = $56/capita, *N* = 1,162) and via mail (WTP = $102/capita, *N* = 304).[iii] In Lithuania (Sweden), based on a face-to-face open-ended (mail discrete choice) survey to 697 (700) respondents, the values indicate a WTP of $7 ($458) per capita.[iv] These values were then transferred to the rest of the countries in the same subregion by calibration based on the GDP per capita at purchasing power parity.

Marine Systems

The valuation of marine systems is clearly less represented in the relevant literature. Particularly studies of the deep seas (below 200 m) that include several ecosystems (such as hydrothermal vents, cold-water corals, deep-sea sponge fields, and abyssal plains) are very scarce. Lately, however, deep-sea mining has drawn the attention of environmental economists, mainly due to the interest of private companies investing in such activities. In a report prepared for such a project, Earth Economics (2015) used the value transfer method to monetize the ecosystem services related to the Solwara 1 seabed mining project in Papua New Guinea (PNG). They relied on WTP estimates for provisioning, regulating, cultural, and supporting services provided by a number of other studies. Due to the lack of studies conducted on the deep-sea bed, their estimates were based on values related to the impact of terrestrial copper mines on cloud forests. The resulting value of $1,766 per hectare per year, as the authors stated, is more likely to be an overestimate of impacts, given that cloud forests are some of the most productive and biodiverse ecosystems on the planet. On the same note, Wakefield and Myers (2016), as part of their cost-benefit analysis of mining deep-sea minerals in the Pacific Island region, provided value estimates for a number of ecosystem services that are to be affected by seafloor massive sulfide mining in PNG and by the extraction of metal values from manganese nodules in the Cook Islands (CI). Once again, values were based on benefits transfer methods with benthic damages, risk of environmental changes associated with spills, and project-related changes for PNG (CI) being estimated at $0.5 million ($18.3 million), $40,000 ($260,000), and $135,000 ($1.35 million), respectively.

Cold-water coral reefs are also much less studied than their continental-shelf counterparts. Glenn et al. (2010) illustrated the application of a CE to the quantification of aspects of socioeconomic value of cold-water deep coral reefs in the Republic of Ireland. The CE survey attributes were the level of fishing activity allowed in the MPA, the spatial extent of the MPA in terms of the area of coral protected, and the cost in terms of a personal additional yearly tax to be paid. With a sample size of just over 500, the cost parameter estimate was not found to be statistically significant and as a result the relevant WTP values could not be calculated. Yet, the comparisons of choice probabilities ranking of the attributes and levels suggested banning trawling and protecting all coral areas as the most preferred management option. Jobstvogt et al. (2014) took into account both nonuse values (number of protected species as a measure of existence value) and use values (potential for new medicines from deep-sea organisms as a measure of option value) associated with deep-sea environments around the coast of Scotland. Their choice experiment survey asked Scottish households to state their preferences regarding additional marine protected areas plans in the Scottish deep sea; attributes were the potential for organisms to contribute to the development of new medicines and biodiversity, expressed as number of marine species that are protected. CE questionnaires were mailed out to respondents and resulted in 397 usable answers. The results showed that there was a positive WTP on the part of Scottish residents for both attributes, and that WTP for the "best" protection option is in the range of £70–77 per household per year. Finally, Aanesen et al. (2015) elicited participants' WTP for increasing the protection of cold-water coral using a CE with a sample of 397 respondents across Norway. Aside from cost, their design included attributes like "size of protected area," "habitat for fish," and "raw material in medicinal products." Their work's novelty derived from the organization of valuation workshops, where respondents had the opportunity to learn more about cold-water coral before they revealed their WTP for coral protection. The results indicated a high degree of preference heterogeneity, and an average WTP in the range of €274–287 per household per year.

Erwin, López-Legentil, and Schuhmann (2010) provided an estimate of the pharmaceutical value of marine biodiversity though the provision of marine natural products (MNPs) that can lead to the development of new drugs for the treatment of human diseases, mostly as chemotherapeutic agents. The value is approximated using data on the phylum-specific potential to produce MNPs, the market hit rates for MNPs, and the lifetime net present value of new anticancer drugs. Their value estimates ranged from $563 billion to $5.69 trillion, varying with total biodiversity estimates and discount rates applied (see original study for details).

Marine Protected Areas and Fauna

Togridou, Hovardas, and Pantis (2006) estimated the recreational value attached to the National Marine Park of Zakynthos, Greece. Using the introduction of entrance fees as a payment vehicle, they collected open-ended contingent valuation responses from 495 visitors accessing three main beaches of the park (Laganas, Kalamaki, and Gerakas). After the valuation task, respondents were asked to elaborate on their responses; in this way, existence and bequest values emerged as the most frequent reasons for WTP. Based on the results, visitors were willing to pay on average a €6.15 entrance fee, with 81% of them placing a value of at least €1. Another example is the work of Hussain et al. (2010), who valued changes associated with the designation of a network of Marine Conservation Zones (MCZs) in English territorial and U.K. offshore waters. The researchers considered 11 categories of ecosystem services[v] that could be affected under two alternative management regimes (“Less Restricted” and “More Restricted”) and three different network configurations. Providing predictions about the possible trajectories of the delivery of the ecosystem services with and without MCZs and adjusting and aggregating the values from previous studies, they monetized the difference between the status quo and the protection policies. Their present value estimates reached £10.2 billion for the least extensive proposed network with the “Less Restrictive” management regime and £23.5 billion for the most extensive/restrictive scenario, while the undiscounted mean annual benefits across the study period (years 0–20) ranged from around £0.92 billion to £1.95 billion, depending on network/restrictions.

Regarding fauna, the vast majority of studies concern marine species' use values for recreational fishing and tourism. This is quite contradictory, considering that these activities are some of the main reasons for biodiversity loss. Only a few studies present nonuse values for the conservation of fish and mammals. Giraud, Turcin, Loomis, and Cooper (2002) elicited WTP values for a protection program for the Steller sea lion, using the CV method. The survey was administered by mail to a random sample of 3,000 households, equally split among the Alaskan Boroughs (68.93% response rate), Alaska statewide (70.22% response rate), and across the entire United States (51.16% response rate). Using the referendum format, the annual value for the expansion of the current Steller sea lion recovery program was estimated at about $100 per household (adjusted to $61 per household if nonresponses are assigned a zero WTP). Another CV study was undertaken in China, where Jin et al. (2010) elicited the WTP for marine turtle conservation from a sample of 3,680 respondents from Beijing (China, *N* = 600), Davao City (Philippines, *N* = 847), Bangkok (Thailand, *N* = 789), and Ho Chi Minh/Hanoi (Vietnam, *N* = 1,444). The survey was set up in a referendum format and involved three treatments (regional mandatory program, regional voluntary program, and national mandatory program); questionnaires were delivered and collected in person. WTP estimates ranged from $0.32 (Davao) to $1.28 (Beijing) per household per month. Other than the Vietnamese sample, no scope (regional vs. national) or payment vehicle (mandatory vs. voluntary) differences were detected among respondents. Finally, Boxall, Adamowicz, Olar, West, and Cantin (2012) examined Canadians' WTP to recover the populations of three marine mammal species found in the St. Lawrence Estuary (belugas, seals, and whales), using a series of six discrete choices between recovery programs (use of a marine protected area or restrictions on whale watching and shipping industries) with additional annual cost to the household (through increased federal income taxes and increased prices for goods due to new restrictions on shipping) and the status quo at no extra cost. The impacts of the programs on the species ranged from none (i.e., remaining "threatened" for the belugas/seals and "endangered" for whales) to plausible levels of improvements. The estimated values for different levels of marine mammal recovery ranged from $77 to $229 per year per household. Results also indicate that the additional value of programs that improve a species' status beyond the "at risk" threshold is relatively small.

Discussion

There have been increasing calls around the world for urgent action to tackle environmental issues related to the oceans, seas, and marine resources. In addition, the 2030 Agenda for Sustainable Development clearly supports this direction in one of its 17 main goals (Goal 14, Life Below Water). The prevention and mitigation of marine and coastal pollution are key factors in achieving sustainable management of these ecosystems and avoiding the significant adverse impacts caused by human activities.

Market and nonmarket valuation methods are valuable approaches to this target, since failing to account for the value of marine and coastal natural capital can lead to decision making that is not fully informed and undermines the importance of such ecosystems. Nonetheless, not all valuation exercises come without caveats; before any estimate is taken into consideration for policy and/or decision-making purposes, it should be checked against valuation standards. Apart from the pitfalls related to each valuation technique, some of the common problems that need to be taken into account are those of the omission of important ecosystem services when valuing an environmental asset and the sensitivity of cost–benefit analyses to the selection of a discount rate, a parameter that cannot be objectively measured but on which the evaluation and implementation of investments heavily relies.

In conclusion, a very sound understanding of coastal systems seems to have been developed, while marine ecosystems, being much less tangible, remain challenging, at least for the very near future. Since there is no first-hand interaction of deep-sea ecosystems with humans and the potential and complexity of these systems are not fully explored, some important ecological services of the marine environment have yet to be valued reliably.

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Table 1. Studies Included in the Review

Study	Region	Ecosystem	Method	Value Estimate
Conroy & Prochaska (1981)	Florida	Marsh	Production function	\$3.00/acre
Sathirathai (1998)	Thailand	Mangrove	Production function	\$33–\$110/hectare
Barbier & Strand (1998)	Mexico	Mangrove	Production function	\$140,000/year/km^2
Markowska & Żylicz (1999)	Poland, Sweden, Lithuania	Coastal waters	CV (multiple)	Multiple
Giraud, Turcin, Loomis, & Cooper (2002)	Alaska	Steller sea lion	CV (N = 1,900)	\$100/household/year
Nunes & Van (2004)	Netherlands	Coastal water	TC (N = 242)	€55/person/year

Study	Region	Ecosystem	Method	Value Estimate
Togridou, Hovardas, & Pantis (2006)	Greece	Marine park	CE (N = 495)	€6.15/visit
Ahmed, Umali, Chong, Rull, & Garcia (2007)	Philippines	Coral reefs	TC (N = 92)	$223/person/year
Parsons & Thur (2008)	Caribbean	Coral reefs	CE (N = 211)	$45–$192/person/year
Sale, Hosking, & Du Preez (2009)	South Africa	Estuaries	CV (N = 150)	R0.072/m^3 of freshwater inflow and R0.013/m^3 of freshwater inflow
Eggert & Olsson (2009)	Sweden	Coastal water	CE (N = 324)	Multiple
Glenn et al. (2010)	Ireland	Cold-water corals	CE (N = ~500)	NA
Hussain et al. (2010)	United Kingdom	Marine conservation zones	Value transfer	£0.92–£1.95 billion/year
Jin et al. (2010)	China, Philippines,Thailand, Vietnam	Marine turtle	CV (N = 600, 847, 789, 1,444)	$0.32–$1.28/household/month
Gopalakrishnan, Smith, Slott, & Murray (2011)	North Carolina.	Beach	Hedonic pricing	$8,800 additional foot of beach width

Study	Region	Ecosystem	Method	Value Estimate
Bishop et al. (2011)	Hawaii	Coral reefs	Hybrid (N = 3,277)	$288/household/year
Boxall, Adamowicz, Olar, West, & Cantin (2012)	Canada	Belugas, seals, and whales	Hybrid (N = 1,606)	$77–$229/household/year
Hynes, Tinch, & Hanley (2013)	Ireland	Coastal water	CE (N = 365)	€6.78/beach visit
Jobstvogt et al. (2014)	Scotland	Deep sea	CE (N = 397)	£70–£77/household/year
Aanesen et al. (2015)	Norway	Cold-water coral	CE (N = 397)	€274–€287/household/year
Castaño-Isaza, Newball, Roach, & Lau (2015)	Colombia	Beaches	CV (N = 1,793)	$997,468/year
Remoundou, Diaz-Simal, Koundouri, & Rulleau (2015)	Spain	Beach	CE (N = 80)	Multiple
Earth Economics (2015)	Papua New Guinea	Deep sea	Value transfer	$1,766/hectare/year
Wakefield & Myers (2016)	Papua New Guinea	Deep sea	Value transfer	~$675,000
Wakefield & Myers (2016)	Cook Islands	Deep sea	Value transfer	~$20 million

Note. Estimates are reported as they appear in the original studies in terms of currencies and years.

Abbreviations: CE = choice experiments; CV = contingent valuation; TC = travel cost.

Notes

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[i] Bioeconomic models can also be considered as forms of production function modeling.

[ii] NOAA panel recommendations bring a set of specific instructions on use for nonmarket valuation (Arrow et al., 1993).

[iii] All values are converted to 1995 USD from the national currencies, using the existing rate at that time.

[iv] These values were converted to discrete choice equivalents, using the results of the Polish sample. The converted values were $28 and $252 per capita for Lithuania and Sweden, respectively.

[v] Eventually, they were able to provide value estimates for seven of them. Also, even for the existing ones, the authors suggested that the estimates used were likely to be underestimates for these ecosystem services categories.