

# Valuing and Evaluating Marine Ecosystem Services

## Putting the Right Price on Marine Environments?

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■ **ABSTRACT:** The flow of ecosystem services from coral reefs, seagrass meadows and mangrove forests sustains the livelihoods of billions of people worldwide. Faced with the global degradation of marine and coastal ecosystems, policy makers are increasingly focusing on ecosystem service valuation techniques to encourage conservation and sustainable use of marine resources. Here we provide a review and synthesis of the available information on economic valuation techniques as applied to tropical marine habitats. Our study demonstrates the high variability and lack of consistency in outcomes from these studies. We conclude that, if the concept of ecosystem goods and services is to make a positive contribution towards managing the impacts of humans on the environment, then economic valuation approaches must reflect the inherent limitations of economic theory whilst emphasizing the complexity and heterogeneity of the natural environment and human decision making.

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■ **KEYWORDS:** conservation policy, coral reefs, environmental valuation, mangrove forests, seagrass meadows

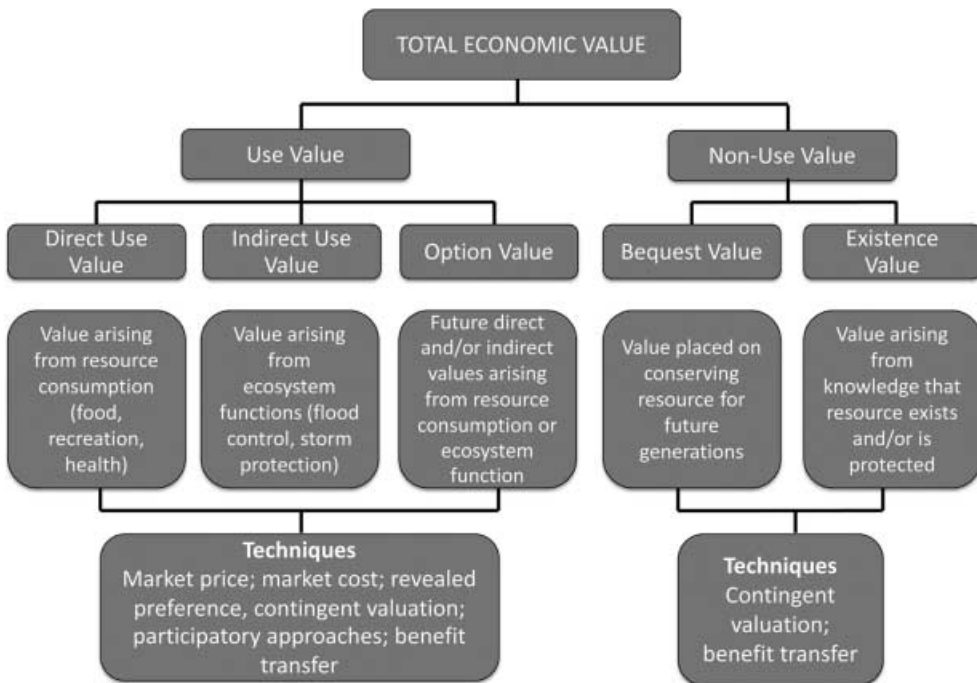
### Introduction

Managing the interaction of people with nature requires a comprehensive framework for understanding and quantifying the impacts of society on the environment. Economists have sought to contribute to this goal through conceptualizing the role of natural resources in models of economic development. The neoclassical school of economic thought that dominated much of the twentieth century is characterized by the belief that natural resources are substitutable by labor or capital. This view implied that technological innovation would allow continued economic growth beyond the limits imposed by natural resource availability. Furthermore, this perspective assumes that the regulatory capacity of the market would cause increasingly scarce resources to be substituted by cheaper alternatives, thus avoiding the complete exhaustion of natural resources (Gómez-Baggethun et al. 2010).

However, as environmental problems multiplied and concern about resource scarcity grew from the 1960s onward, attention increasingly focused on the implications of economic substitutability. The school of environmental economics that developed in the 1970s reflected the view that the neoclassical economic approach systematically undervalued the environment through only recognizing those environmental goods or services that could be readily monetized and substituted. The benefits arising from these goods and services were termed “direct use values”



and included the profits from, for example, fishing, forestry or mining activities. Advances in environmental science and the growing recognition of diverse cultural values associated with the environment clearly demonstrated that human society benefited from ecosystems and their functions in many other ways. Thus, economists' attention was increasingly directed toward monetizing the full range of ecosystem goods and services used both directly and indirectly by human society. This articulation of the concept of monism, that is, the belief that valuation can only be achieved through expression in a single accounting unit (Røpke 2005), enables the capture of non-market values (or non-use values) along with the more readily monetized use values. Aggregating these monetized components results in the "total economic value" of the environment, as depicted in figure 1.



In this article, we explore how marine ecosystems and the services they provide to society are being valued through the application of neoclassical economic approaches. We provide a summary of evaluation techniques and their relationship to categories of ecosystem services, along with an outline of the importance of coral reefs, mangrove forests and seagrass meadows in environmental and societal terms. We then examine the recent peer reviewed academic literature dealing with the valuation of these habitats and offer an explanation as to the continued proliferation of these studies, along with some comments as to how these may be refined to better assist marine conservation practitioners and resource user communities.

### Economic Valuation Techniques and Valuing Ecosystem Services

A range of techniques to measure use and non-use values have been developed and refined over the past four decades. We summarize these briefly here, but the reader is referred to Christie et al. (2012) for a more thorough, recent review. *Market price techniques* rely upon direct obser-

vation of prices paid to receive environmental services to gain an approximation of the value of that service. For example, tourism revenues from a particular site can be used to project the total monetized value of the resources within that attraction. Second, *market cost approaches* involve measuring the value of an environmental good or service that acts as a proxy for the value of the good or service in question. Thus, the *opportunity cost approach* would measure the value of alternative uses (for example, roads or buildings) to identify the value of a specific resource (for example, a lake). Alternatively, the *replacement cost method* uses the cost of replacing an environmental resource or service, such as the cost of reforestation, as a proxy for the value of a forest. *Revealed preference approaches* represent a third technique, using information on people's behavior to determine environmental value. The *travel cost method* is one example whereby the cost of trips to a site is used to measure its value. Alternatively, *hedonic pricing* uses the value of a complementary environmental good as a guide to the value of a non-market good or service. This approach is most common when considering the effects of environmental quality, as reflected by factors such as proximity to green space, on property prices. A fourth category is represented by *contingent valuation*, which involves surveys of individuals' "willingness to pay" to receive an environmental good or service or, alternatively, the sum that people would view as appropriate compensation, or the "willingness to accept", for the loss of a good or service. Contingent valuation is also referred to as a "stated preference" technique as it rests upon hypothetical scenarios to infer value. Choice modelling is a variant of contingent valuation whereby individuals are asked to select from a range of policy scenarios, each of which is presented with a theoretical cost attribute, rather than asking for choices based directly upon financial considerations. The results of individual responses are then analyzed to derive a monetary value for each scenario. The fifth method is *participatory approaches*, which involve a combination of the preceding techniques set in a small group or workshop context, with time given for reflection, discussion and re-evaluation to identify individual or collective values associated with environmental assets. Finally, *benefit transfer* represents a technique whereby use or non-use values measured in one geographic region using one of the preceding methods are transposed to a different location and weighted or otherwise calibrated according to local site specifics.

These refinements in economic approaches to valuing the environment occurred in conjunction with moves towards defining and categorizing the environmental goods and services utilized by human society. Schumacher was among the first to popularize the notion of "irreplaceable capital" (Schumacher 1973: 19) as providing the foundation for economic activity, which came to be referred to as "natural capital" or the "ecosystem services" provided by nature. The existence of these services was used to demonstrate how their loss through progressive environmental degradation, and biodiversity reduction in particular, would impact upon human wellbeing (de Groot 1987; Costanza and Daly 1992). While these relationships were given considerable impetus by the focus on sustainable development in the 1990s, Costanza and colleagues' (1997) mid-range estimate of the annual global economic value of ecosystem services at US\$33 trillion, approximately double that of the annual global gross domestic product (GDP) at that time, represented a landmark in drawing policy makers' attention to this concept and its application. This was reflected in the publication of the United Nations Millennium Ecosystem Assessment (hereafter MA), which was conducted to provide guidance on achieving the Millennium Development Goals and which drew heavily on the concept of ecosystem services. These ecosystem services were defined in very broad terms as "the benefits people obtain from ecosystems" (MA 2003: 27) and categorized into provisioning, regulating, cultural or supporting services. The MA definition of ecosystem services remains the most widely used in the literature and has been applied to measure use and non-use values using a variety of techniques as summarized in table 1.

**Table 1.** Ecosystem services and their valuation (adapted from MA 2003, Farber et al. 2006)

Category	Description	Examples	Valuation techniques
Provisioning Services	Products obtained from ecosystems	Food supply, potable water, building materials	Market pricing; opportunity cost; travel cost
Regulating Services	Benefits obtained from natural regulation of ecosystem processes	Climate regulation, water regulation, soil erosion control	Contingent valuation; hedonic pricing; opportunity cost; travel cost
Cultural Services	Non-material benefits obtained from ecosystems	Recreation, education, spiritual, aesthetic benefits	Contingent valuation; travel cost
Supporting Services	Processes necessary for supply of all other services	Nutrient cycling, soil formation	Contingent valuation; opportunity cost

A common critique of the MA approach to ecosystem service definition relates to how it can facilitate the erroneous double counting of goods and services (Boyd and Banzhaf 2007; Fisher et al. 2009). The overlap between categories, particularly relating to supporting services, is of concern in that, for example, drinking water is a provisioning service while water purification arising from filtration through the soil is a supporting service. Furthermore, recreational activity at a site may be at least partly dependent on the presence of clean water; hence, the valuation of potable water as an ecosystem service will be included under more than one category and its value inflated through double counting. The natural complexity of ecosystems, together with spatial variability and complementarity in ecosystem processes, renders the MA's relatively simplistic categorization particularly susceptible to errors in valuation (Fu et al. 2011).

Nevertheless, the notion that ecosystems contribute to human wellbeing has found enthusiastic support in related fields of study (Daily and Matson 2008; Carpenter et al. 2009; Braat and de Groot 2012). It has also found expression in a reinvigoration of neoclassical economic approaches to environmental issues, ranging from the economic rationale for action on climate change (Stern 2007) to the value of ecosystem services at the global, national and regional level (Hein et al. 2006; de Groot et al. 2012; Bateman et al. 2014; Kubiszewski et al. 2013) and the value of services provided by specific ecosystems (Costanza et al. 2008; Lal et al. 2009). A recent survey indicated that the publication of academic articles involving ecosystem services has increased exponentially since the late 1990s, these being published in over 50 different journals (Crossman et al. 2013).

The justifications put forward by proponents of economic valuation of environmental resources come from one of several standpoints. One school of thought holds that monetary valuation is necessary if we are to recognize the importance of ecosystems to human society, a view reflected by the MA in its statement that "current decision-making processes often ignore or underestimate the value of ecosystem services" (MA 2003: 33). Ecologists may contribute to this perspective through supporting valuation processes in the belief that they elevate conservation priorities (Balmford et al. 2002; Emerton and Bos 2004; Conservation International 2008; Naidoo et al. 2008). Secondly, valuation is favored by those who ascribe primacy to the market in issues of resource allocation, as this ensures that all users pay for the benefits they receive from the environment. Thus, for example, negative externalities associated with climate change may be costed and included in policy design (Altemeyer-Bartscher et al. 2010). This is further justified through reference to increasing resource scarcity and the need for trade-offs between competing resource users and uses, which in turn necessitate an objective economic basis for

decision-making (MA 2003). Finally, orthodox economists argue that, as Western human society expresses its preferences through the use of money, this utilitarian lens is the logical framework through which ecological processes should also be valued (Liu et al. 2010).

However, counter-arguments question this reductionist approach to resource management, reflecting both the complexities of the natural environment and perceived limitations of the valuation techniques. From a biological standpoint, the simplification inherent in valuing individual marine habitats is inappropriate when one considers the inter-connected nature of these ecosystems. While estimates of economic value largely consider marine habitats in isolation, the vast majority of species move between multiple habitats on a daily and tidal basis (Unsworth et al. 2008). As such, direct use values assigned to one habitat type based on where fish may be caught are not reflective of the range of habitats that support that species. For example, Islam and Ikejima (2010) assigned values to a mangrove fishery in Thailand, although many of the species of fish caught also depend upon seagrass meadows and coral reefs for feeding or at certain points in their life cycle (Watson et al. 1993). Second, there is extensive ecological connectivity between individual marine habitats. Tropical seagrass meadows are commonly net exporters of organic matter, and as such provide trophic subsidy to adjacent or nearby habitats (Heck et al. 2008). Wolff (1976) demonstrated how seagrass meadows of the Great Bahama Bank supported whole deep-sea ecosystems through supply of organic carbon. Most valuations of tropical fisheries fail to consider this connectivity between multiple habitat types, again leading to a potential underestimation of the value and range of marine ecosystem goods and services.

Conservation in maritime environments is rendered difficult by the frequent absence of, or confusion over, property rights (Kidd and Ellis 2012). Thus, whereas in a terrestrial setting private ownership, or communal practices, may dictate how resources are exploited or conserved, it may be more logical and practical to enable market forces to determine resource usage in a marine context, because these seek to change the behavior of users and thus the demands made on particular resources. It should be no surprise, therefore, to see that economic valuation and commodification of marine resources is being promoted as part of the current conservation agenda. However, there are serious questions relating to the tacit assumptions of economic valuation processes that cast doubt on the relevance of these exercises to the real world.

First, all valuation techniques rest upon the assumption of economic rationality and self-regarding behavior, that is, actions are determined through prioritizing those that maximize the economic benefit accruing to the individual. While this assists economic modelling, there remains scant evidence for such behavior in reality, with a majority of individuals displaying “irrational” behavior reflecting the existence of other preferences (Camerer and Fehr 2006). Non-economic reasons driving conservation-oriented behavior may include religious beliefs, spiritual practices, environmental awareness, social mores, peer group pressure or the desire to exclude other resource users (van Helden 2001; Kosoy et al. 2007). Equally, one might assume that irrational motives resulting in resource exploitation may exist, which could include immediate livelihood priorities such as food, fuel or the need for shelter, uncertainty over resource tenure and ownership or religious values that encourage resource degradation through a belief in human supremacy or inviolability. Collective drivers of decision-making may also be expected to be particularly strong in maritime contexts where collective choice rules often determine access rights to common pool resources (Schlager and Ostrom 1992).

A myriad of reasons for irrational behavior that contradict the assumptions of valuation techniques may therefore exist, which can only be understood through reference to the local political, economic, cultural or social context (Selin 2003). This problem may be compounded through economic valuation surveys attempting to include local resource users and foreign visitors in order to provide a “representative” sample of resource users in developing countries (see

table 2). Clearly, marine resource dependency within a local community would be manifest in elevated direct use values expressed through surveys of that community. However, these will inevitably be eclipsed by the magnitude of non-use values derived from surveys of relatively affluent foreign visitors, leading to situations where the total economic value may be unrepresentative of local respondents' opinions.

The preceding line of argument implies that the assumption of economic rationality that underpins all valuation methods illustrated in figure 1 is open to question, rendering the process of estimating use values and non-use values susceptible to unknown degrees of error. This is compounded in the case of contingent valuation, which is often the most commonly used technique on account of its capacity to be able to measure all components of total economic value. Many detailed critiques of the contingent valuation process have been published (Carson et al. 2001; Vatn 2004; Spash 2008a). These include reference to various aspects of inappropriate survey design and manipulation of responses, particularly with regard to the treatment of so-called *zero bids*. These involve researchers' decisions as to whether respondents indicating a zero willingness to pay for goods or services are registering a protest against the assumption of pricing such goods or are indicating their genuine valuation. If the former is assumed, these "protest bids" are often removed from the results, while their inclusion inevitably lowers the calculated resource value. Furthermore, the tendency of survey respondents to associate themselves with environmental protection regardless of the actual mechanism or policy focus, termed the "warm glow", may be responsible for some inconsistencies in measures based on contingent valuation (Diamond and Hausman 1994). While some researchers maintain that such technical considerations do not negate the validity of the contingent valuation method (Ressurreição et al. 2012), these issues clearly raise concerns as to the choice of phrasing and survey design as well as the relevance and comparability of data generated through this technique (Spash 2008a).

## Environmental and Societal Functions of Marine Ecosystems

In this study, we concentrate on valuation in relation to key elements of the tropical seascape. Coral reefs, seagrass meadows and mangrove forests are subject to significant degradation and loss (Alongi 2002 Pandolfi et al. 2003; Waycott et al. 2009), but are renowned for their biological diversity, high productivity and natural beauty, resulting in significant contributions to local and national economies as well as human well-being. For example, coral reefs support 32 of the 34 known animal phyla (Wilkinson 2002). For comparison, just nine of these phyla are found in tropical rainforests. However, it is the high productivity of these ecosystems within otherwise unproductive waters that makes them critical to maintaining marine biodiversity, meeting the needs of local resource users, and contributing to national economies and export markets (Wilkinson 1996; Berg et al. 1998, Hoegh-Guldberg 1999).

Calculations from Fiji found that one square kilometer of actively growing reef could support over 300 people if no other protein sources were available (Jennings and Polunin 1996). Reef-related tourism is also a major foreign currency earner for many countries, while reefs also provide natural protection from wave action and potential storm damage (Cesar 2000; Burke et al. 2011). On the global scale, reefs are valued for their role in the carbon and calcium cycles and for bioprospecting, which can benefit developments in agriculture and the food, pharmaceutical and chemical industries (Spurgeon 1992; Pendleton 1995).

Mangrove forests are one of the most productive ecosystems on earth. These spatially extensive habitats, covering over six million hectares in Southeast Asia alone, provide a range of valuable ecosystem services that contribute to human wellbeing and have a potentially large

economic value. Mangroves provide key goods to subsistence economies such as timber, fuel wood, and charcoal, and are important in regulating the coastal environment through flood, storm and erosion protection, and the prevention of salt water intrusion (de Groot et al. 2002; Brander et al. 2012; Vo et al. 2012). A significant role provided by mangrove forests is that of habitat for biodiverse fauna (birds, fish, reptiles, invertebrates), including a key role as nurseries and habitat for commercially important fish species. This role was clearly highlighted by a Caribbean-wide study that found coral reefs on islands lacking mangroves had 50 percent less fish biomass (Mumby et al. 2003). Mangrove forests are additionally valued for the cultural services they provide to people around the world (e.g., recreation, aesthetic, non-use) (Brander et al. 2012).

Seagrass meadows are equally important to the coastal seascape, with evidence now available of their global value to human wellbeing (Cullen-Unsworth et al. 2014). These extensive intertidal, subtidal and deepwater habitats in the tropics are important for their ecological functions and ecosystem services, such as their role in food web dynamics, seascape interactions and biogeochemical processes (Duarte 2002; Moberg and Ronnback 2003). While the economic value of seagrasses for cycling nutrients has been long established (Costanza et al. 1997), there is now increasing recognition of how seagrass meadows store and sequester globally significant amounts of carbon (Fourqurean et al. 2012; Lavery et al. 2013) and buffer seawater acidity (Unsworth et al. 2012; Hendriks et al. 2013). Seagrass meadows and their fauna also directly and indirectly provide food security and livelihoods globally (Jackson et al. 2001; Unsworth and Cullen 2010). In addition, seagrass meadows have long been known to provide cultural benefits to coastal peoples (Wyllie-Echeverria and Cox 1999).

## Methodology

We conducted a review of the literature focusing on the economic value ascribed to specific coral reefs, mangroves and seagrass meadows through an audit of academic journal articles published between 2000 and 2013. Research published by non-academic institutions and non-governmental organizations where peer review is absent were not included, as there is no similar guarantee of academic rigor. We carried out this survey through combining the keywords “coral reef”, “seagrass”, “mangrove”, “ecosystem services” and “economic valuation” in the title search engine function of various databases. These included the academic journal databases operated by Science Direct and Web of Science together with web-based databases relating to environmental economics (Environmental Valuation Reference Inventory n.d.; Marine Ecosystem Services Partnership n.d.). We recorded the date of fieldwork, research location, element(s) of economic value and ecosystem services measured, economic valuation techniques, survey methods used and resource values obtained. All studies expressed resource values in US dollars on an annual basis, obviating the need to convert between currencies. The effect of inflation was corrected through adjusting all figures to 2005 values through applying the Consumer Price Index. Where possible, resource values are expressed in spatial terms with the hectare (ha) used as the standard unit of measurement.

## Results

A number of reviews and meta-analyses have been conducted in relation to valuing marine ecosystems in recent years. Liqueste et al. (2013) identified 145 publications that included marine

Table 2. Results of literature survey: coral reef valuation

Research location and date	Economic values measured	Ecosystem services	Valuation techniques	Data collection methods	Resource value (US\$ 2005)	Reference
Philippines 2000	Direct use; indirect use; option; bequest	Cultural: recreational	Contingent valuation; travel cost	Questionnaire of local public and foreign visitors	\$5.3M yr <sup>-1</sup>	Ahmed (2007)
Philippines 1997	Direct use; indirect use; option; bequest	Cultural: recreational	Contingent valuation; travel cost	Questionnaire of local public and foreign visitors	\$4,000-\$1.2M yr <sup>-1</sup>	Arin and Kramer (2002)
SE USA 1996	Direct use; indirect use; option; bequest	Cultural: recreational	Contingent valuation; travel cost	Questionnaire of local public and foreign visitors	\$13,680 ha <sup>-1</sup> yr <sup>-1</sup>	Bhat (2003)
Great Barrier Reef Australia 2000	Direct use; indirect use	Cultural: recreational	Travel cost	Questionnaire of local public and foreign visitors	\$795M-\$1.8 billion yr <sup>-1</sup>	Carr and Mendelsohn (2003)
Mesoamerican Barrier Reef/Mexico 2005	Direct use; indirect use; option; bequest	Cultural: recreational	Contingent valuation	Questionnaire of local public and foreign visitors	\$100M-\$400M yr <sup>-1</sup>	Casey et al. (2010)
Hawaii 2001	Total economic value	Provisioning; regulating; cultural	Market price; contingent valuation; travel cost; hedonic pricing	Questionnaire of local public, resource users and foreign visitors	\$375 ha <sup>-1</sup> yr <sup>-1</sup>	Cesar and van Beukering (2004)
Philippines 2007	Total economic value	Provisioning; regulating; cultural; supporting	Market price; travel cost; contingent valuation	Questionnaire of local public, resource users and foreign visitors	\$180 ha <sup>-1</sup> yr <sup>-1</sup>	Cruz-Trinidad et al. (2011)
Netherlands Antilles 1993	Direct use; indirect use; option; bequest	Cultural: recreational	Contingent valuation; travel cost	Questionnaire of local public and foreign visitors	\$640,000 yr <sup>-1</sup>	Dixon et al. (2000)
Indonesia 2003	Total economic value	Provisioning, regulating, cultural	Market price; market cost; contingent value	Questionnaire of local resource users	\$139 ha <sup>-1</sup> yr <sup>-1</sup>	Hargreaves-Allen (2004)
SE USA 2001	Direct use	Cultural: recreational	Market price; contingent valuation	Postal survey of local public	\$250M yr <sup>-1</sup>	Johns et al. (2001)
Fiji 2006	Direct use; indirect use; bequest	Provisioning; regulating; cultural	Market price; market cost; contingent valuation; benefit transfer	Questionnaires and focus groups of local resource users	\$942 ha <sup>-1</sup> yr <sup>-1</sup>	O'Garra (2012)
Iran 2009	Total economic value	Provisioning; regulating; cultural; supporting	Market price; contingent valuation	Questionnaire of local public and foreign visitors	\$215,000 ha <sup>-1</sup> yr <sup>-1</sup>	Madani et al. (2012)
Philippines 2004	Total economic value	Provisioning; regulating; cultural; supporting	Market price; benefit transfer	Questionnaire of local resource users and foreign visitors	\$2,350 ha <sup>-1</sup> yr <sup>-1</sup>	Samonte-Tan et al. (2007)
Thailand 2000	Direct use; indirect use; optional; bequest	Cultural: recreational	Travel cost; contingent valuation	Questionnaire of local resource users and foreign visitors	\$17,180 ha <sup>-1</sup> yr <sup>-1</sup>	Seenprachawong (2004)



ecosystem services, while de Groot et al. (2012) cited 94 estimates of coral reef monetary value. However, Londoño and Johnston (2012) listed just 27 studies in academic journals published between 1986 and 2007 that included estimates of recreational value associated with coral reefs. Our search reflected this relative paucity of monetary valuations of marine ecosystems in peer-reviewed academic journals, finding a total of 14 papers on coral reefs, nine dealing with mangroves and seven focusing on seagrass meadows published between 2000 and 2013. The majority of studies were located in the developing world, with developed countries represented mainly by the United States and Australia. While this is not particularly surprising, given our focus on coral reef, mangrove and seagrass ecosystems, there is a distinct emphasis of research in Southeast Asia, and the Philippines in particular, amongst the developing world case studies. In terms of economic values, most coral reef studies measured total economic value or a broad spectrum of use values, while almost all mangrove and seagrass studies focused on direct and indirect use value. Cultural, and specifically recreational, ecosystem services were the sole focus of almost half of the coral reef case studies, while provisioning and regulating services were most common in the mangrove and seagrass articles. Reflecting this, market price and market cost techniques were most frequently used in both mangrove and seagrass valuation exercises, while contingent valuation was used in all but one of the coral reef studies. Most coral reef researchers used a combination of local resource users and foreign visitors in their work, in contrast to mangrove and seagrass studies, which focused on local resource users.

Eight of the coral reef studies included a spatial component, allowing values to be expressed in comparable per hectare units (table 2). Even if the extreme reef value outlier of \$215,000 per hectare per year ( $\text{ha}^{-1} \text{yr}^{-1}$ ) (Madani et al. 2012) is removed, there remains a variation of two orders of magnitude between the lowest annual reef value of \$139  $\text{ha}^{-1} \text{yr}^{-1}$  (Hargreaves-Allen 2004), representing all four reef ecosystem services, and the highest value of \$17,180  $\text{ha}^{-1} \text{yr}^{-1}$  (Seenprachawong 2003), which reflects only one component, cultural services. Two studies on coral reefs in the Philippines, which both focused on the annual value of all four ecosystem services, generated results ranging from \$180  $\text{ha}^{-1} \text{yr}^{-1}$  (Cruz-Trinidad et al. 2011) to \$2,350  $\text{ha}^{-1} \text{yr}^{-1}$  (Samonte-Tan et al. 2007). Comparison of data is clearly limited where no spatial unit was included in coral reef valuations identified through the present study. However, it is notable that the highest reef values of \$250 million  $\text{yr}^{-1}$  (Johns et al. 2001), \$400 million  $\text{yr}^{-1}$  (Casey et al. 2010), and \$1.8 billion  $\text{yr}^{-1}$  (Carr and Mendelsohn 2003) are associated with larger reef systems and developed countries. The two lowest non-spatial total reef values of \$4,000  $\text{yr}^{-1}$  (Arin and Kramer 2002) and \$640,000  $\text{yr}^{-1}$  (Dixon et al. 2000) are associated with smaller reefs in the Philippines and the Netherlands Antilles respectively.

These values can be contextualized with reference to de Groot et al. (2012), wherein a wider sample of coral reef valuation exercises were analyzed. These are presented in table 3. This demonstrates that, despite the differing sample size, the considerable variation in coral reef values identified through the current survey is reflected in the broader sampling conducted by de Groot et al. (2012).

**Table 3.** Comparison of monetary value of coral reefs. All values expressed in US\$  $\text{ha}^{-1} \text{yr}^{-1}$  (2005 equivalent)

Source	Number of values	Mean	Median	Standard deviation	Minimum value	Maximum value
This survey	8	31,230	1,646	74,556	139	215,000
De Groot et al. (2012)	94	333	187	631	35	2,008,606

While the mangrove case studies display a relatively smaller magnitude of variation (table 4), there is a wide discrepancy between the lowest value of \$105 ha<sup>-1</sup> yr<sup>-1</sup> (Islam and Ikejima 2010) and the highest value of \$23,724 ha<sup>-1</sup> yr<sup>-1</sup> (Donato et al. 2011). Furthermore, the magnitude of the calculated economic value does not appear to correspond to the number of ecosystem services measured. By way of illustration, the total economic value for mangrove forest of \$983 ha<sup>-1</sup> yr<sup>-1</sup> (Samonte-Tan et al. 2007) is an order of magnitude less than other studies identified in table 4 focusing solely on direct use services in mangroves (Barbier 2007; Gilman and Ellison 2007).

**Table 4.** Results of literature survey: mangrove forest valuation

Research location and date	Economic values measured	Ecosystem services	Valuation techniques	Data collection methods	Resource value (US\$ 2005)	Reference
Thailand 1996	Direct use; indirect use	Provisioning; regulating	Market price; market cost	Questionnaire of local resource users	\$4,500 ha <sup>-1</sup> yr <sup>-1</sup>	Sathirathai and Barbier (2001)
Philippines 2005	Direct use; indirect use	Provisioning	Market price; contingent valuation	Questionnaire of local resource users	\$315 ha <sup>-1</sup> yr <sup>-1</sup>	Walton et al. (2006)
Thailand 2004	Direct use	Provisioning; regulating; supporting	Market cost	Questionnaire of local resource users	\$12,392 ha <sup>-1</sup> yr <sup>-1</sup>	Barbier (2007)
American Samoa 2006	Direct use	Provisioning	Replacement cost	Questionnaire of local public	\$13,030 ha <sup>-1</sup> yr <sup>-1</sup>	Gilman and Ellison (2007)
Philippines 2004	Total economic value	Provisioning; regulating; cultural; supporting	Market price; benefit transfer	Questionnaire of local resource users	\$983 ha <sup>-1</sup> yr <sup>-1</sup>	Samonte-Tan et al. (2007)
India 2007	Direct use	Supporting	Replacement cost	Survey of nutrient content	\$232 ha <sup>-1</sup> yr <sup>-1</sup>	Hussain and Badola (2008)
Sri Lanka 1998	Direct use; indirect use	Provisioning	Market price; replacement cost; contingent valuation	Questionnaire of local public	\$1088 ha <sup>-1</sup> yr <sup>-1</sup>	Gunawardena and Rowan (2010)
Thailand 2005	Direct use	Provisioning	Market price	Questionnaire of local resource users	\$105 ha <sup>-1</sup> yr <sup>-1</sup>	Islam and Ikejima (2010)
Indo-Pacific 2011	Indirect use	Regulating	Market price	Carbon storage calculation	\$23,724 ha <sup>-1</sup> yr <sup>-1</sup>	Donato et al. (2011)

Excluding the extreme seagrass meadow outlier of \$2.3 million ha<sup>-1</sup> yr<sup>-1</sup> (Vassallo et al. 2013) determined using the abstract concept of emergy (Brown and Herendeen 1996), there is again considerable variation, from \$78 ha<sup>-1</sup> yr<sup>-1</sup> (Unsworth et al. 2010) to \$141,094 ha<sup>-1</sup> yr<sup>-1</sup> (Engeman et al. 2008), both of which measure provisioning ecosystem services only (table 5). A remarkably similar seagrass value to that of Unsworth and colleagues (2010) is cited by Samonte-Tan et al. (2007), but the latter study purports to measure a much wider range of economic values and ecosystem services than those evaluated by the former. Reflecting the increasing interest in carbon sequestration and storage by marine habitats, the seagrass literature now includes estimates of their regulating services based on carbon valuation in the order of A\$3.9–5.4 billion (approximately US\$3.6–5 billion) (Lavery et al. 2013).

**Table 5.** Results of literature survey: seagrass meadow valuation

Research location and date	Economic values measured	Ecosystem services	Valuation techniques	Data collection methods	Resource value (US\$ 2005)	Reference
Indonesia 2006	Direct use; indirect use	Provisioning; cultural	Market price; travel cost	Questionnaire of local resource users	\$2,287 ha <sup>-1</sup> yr <sup>-1</sup>	Dirhamsyah (2007)
SE USA 2008	Direct use	Provisioning	Contingent valuation	Questionnaire of local resource users	\$141,094 ha <sup>-1</sup> yr <sup>-1</sup>	Engeman et al. (2008)
Australia 2013	Indirect use	Regulating	Market price	Carbon storage calculation	\$394 ha <sup>-1</sup> yr <sup>-1</sup>	Lavery et al. (2013)
Australia 2006	Direct use	Provisioning	Market price	Primary productivity calculation	\$120 ha <sup>-1</sup> yr <sup>-1</sup>	McArthur and Boland (2006)
Philippines 2004	Total economic value	Provisioning; regulating; cultural; supporting	Market price; benefit transfer	Questionnaire of local resource users and foreign visitors	\$76 ha <sup>-1</sup> yr <sup>-1</sup>	Samonte-Tan et al. (2007)
Indonesia 2005	Direct use	Provisioning	Market price	Questionnaire of local resource users	\$78 ha <sup>-1</sup> yr <sup>-1</sup>	Unsworth et al. (2010)
Italy 2006	Indirect use	Regulating	Market cost	Calculation of emergy values	\$2.3M ha <sup>-1</sup> yr <sup>-1</sup>	Vassallo et al. (2013)

## Discussion

Our analysis finds that peer reviewed valuation studies focusing on coral reefs, mangroves and seagrass meadows published over the last decade provide little consistency in data, methods or results. Different combinations of valuation techniques are used to measure a variety of ecosystem services. There appears to be no discernible relationship between resource value on the one hand and the ecosystem services measured or valuation technique used on the other. The variation in resource values across several orders of magnitude noted in this study is also reflected in earlier meta-analyses (Barbier et al. 2011; De Groot et al. 2012; Salem and Mercer 2012).

In addition to the differences we have recorded above, attempts to derive a single value for marine ecosystems display a similar degree of variability over time. In an important recent review Costanza and colleagues (2014) recorded a 40-fold increase in the total economic value of coral reef ecosystems, from \$8,400 ha<sup>-1</sup> yr<sup>-1</sup> in 1997 to \$352,250 ha<sup>-1</sup> yr<sup>-1</sup> in 2011. The authors attributed this difference to resource degradation and improved data availability rather than being used to explore any inherent limitations of the methodology.

Taken together, these points question the principle of commensurability that underpins all ecosystem service valuation exercises (MA 2003). We feel that this diversity could be more prominent in debates about valuing ecosystem services. For example, in Costanza and colleagues' research the range of values (specifically for coral reefs) is acknowledged by the authors (2014: 155), but they still take the mean of this diversity to produce a "global average". Indeed, they do so precisely and explicitly because there are insufficient studies available to understand what drives these differences. This means, however, that a single figure of the value of ecosystem services is communicated in order to raise awareness of the "magnitude of these services" (2014: 157) when it is precisely that magnitude that current incommensurabilities of methods and findings make uncertain.

We suggest that, while the criticisms relating to the complexity of the marine environment and shortcomings in valuation techniques discussed earlier are relevant, the continued influence of economic valuation in terms of policy and resource management owes more to its alignment with the dominant neoliberal conservation paradigm. Economic valuation exercises cannot be considered as ideologically neutral, as they serve to articulate a particular perspective on property rights and environmental resources (Gómez-Baggethun et al. 2010). Despite some assertions to the contrary (Costanza et al. 1997), the monetization of ecosystem goods and services clearly implies that they can be traded for another sum of money through a process of commodification. This is part of a wider capitalist dynamic whereby state-led regulation of the environment is replaced by voluntary schemes, fee-paying mechanisms operated by the private sector and policy decisions based upon economic valuation of resources (Büscher et al. 2012; Dempsey and Robertson 2012). Payments for ecosystem services (PES) mechanisms can then be implemented that create a market for these newly quantifiable ecosystem goods and services, thereby encouraging pro-environmental behavior by linking local resource users and the broader global community through the "beneficiary pays principle" (Pagiola et al. 2002).

If viewed as an exercise in applying an economic ideology, the inconsistencies in method and data that we have documented above may actually be irrelevant. They merely add to the uncertain context within which many PES schemes operate, reflecting the complexities of ecosystems and property rights along with other factors such as limited information and institutional weaknesses (Muradian et al. 2010). As such they underline the need for PES proponents to create a discourse that enables both PES and their constituent economic valuation techniques to be presented as success stories (Büscher 2014). This may be achieved through building and maintaining "interpretive communities" (Mosse 2004, 646) whose authoritative accounts of reality serve to sustain their own members' status and interests.

Interpretive communities in this context are represented by international nongovernmental organizations (NGOs) that have long championed environmental economics as a means to further international conservation priorities (Daily 1997; Kareiva et al. 2011; Tercek and Adams 2013; cf. Benabou, this volume). These organizations continue to actively influence this agenda through supporting the implementation of market-based mechanisms (Wendland et al. 2010; The Nature Conservancy 2011). In Southeast Asia, these environmental NGOs have cemented a decade-long alliance with national governments through the creation of the Coral Triangle Initiative, which focuses upon achieving shared objectives relating to food security, fisheries

management, marine biodiversity conservation and promoting resilience to climate change in coastal communities (Clifton 2009; Foale et al. 2013). Rosen and Olsson (2013) have demonstrated the mechanisms through which these NGOs have crafted new relationships and institutions in pursuit of their common regional agenda for Southeast Asia. While similar research is clearly needed in other situations before more generalized conclusions can be drawn, this example does demonstrate how “interpretive communities” composed of state and private sector alliances with a common interest in supporting economic valuation may direct and dominate the marine conservation agenda.

## Conclusion

Economic valuation is described as an integral aspect of resource management and an essential component of balanced and well-informed policy making (Costanza et al. 1997; Costanza 2000). This view is explored through analysis of case studies of coral reef, mangrove and seagrass valuation published in peer-reviewed academic journals since 2000. These demonstrate that a variety of valuation techniques is used and the calculated ecosystem services values range across several orders of magnitude.

This lack of commensurability should lead to serious questions being raised as to the relevance of economic valuation of ecosystem goods and services to environmental policy-making. This is reflected to some extent in the number of valuation exercises published in gray literature (as cited by de Groot et al. 2012) in comparison to the current study and previous research focusing exclusively on peer-reviewed academic journals (Londoño and Johnston 2012). We consider that the popularity of economic valuation reflects its alignment with the neoliberal orthodoxy that promotes market-based solutions to environmental issues. Furthermore, we suggest that the alliances developed between international conservation NGOs, major financial institutions and state governments are sufficiently influential to overcome numerous detailed and incisive academic critiques relating to the methodology and assumptions inherent in economic valuation procedures (Spash 2008a, 2008c).

The imposition of a theory of economic valuation that assumes homogeneity and predictability in a context characterized by spatially and temporally variable resources and unpredictable stakeholders will inevitably lead to conflicting interpretations over the relevance of data generated and the nature of the “problem” itself (Thompson and Warburton 1985). There are, however, alternatives that could enhance the utility of economic valuation procedures through obviating some of the more fundamental issues, particularly those associated with contingent valuation. Choice modelling is a process whereby respondents rank alternative scenarios that may be differentiated by more than one attribute. Through including cost as one of these attributes, willingness to pay can be derived indirectly rather than explicitly, as in the case of contingent valuation, thus avoiding several difficulties of contingent valuation (Hanley et al. 2001). However, the complexity of ranking multiple scenarios can limit the potential for choice modelling to measure a sufficiently wide range of options. As an alternative, participatory approaches such as deliberative monetary valuation involve practitioners guiding focus group discussions on shared environmental concerns, the pros and cons of valuation and reflections on willingness to pay (Spash 2008b). These mixed-method techniques offer a means to conduct valuation exercises that reflect the innate pluralism of individual perceptions of the natural environment (Oles 2007), while also helping to identify underlying beliefs regarding the values attributed by people to ecosystem services (Kenter et al. 2011; Lo and Spash 2013). Although their usage has been limited thus far, there is clear scope to address some of the critiques of contingent valua-

tion while fulfilling broader policy objectives regarding the promotion of public participation in decision-making processes.

It is therefore apparent that a renewed effort amongst scholars is needed that recognizes the limitations of economic theory whilst emphasizing the complexity of both the natural environment and the nature of human decision-making. This appears essential if the concept of ecosystem goods and services is to make a positive contribution towards managing the impacts of humans on the environment.

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