



# Estuarine and Coastal Ecosystems as Defense Against Flood Damages: An Economic Perspective

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The rapid loss of estuarine and coastal ecosystems (ECEs) in recent years has raised concerns over their role in protecting coastal communities from storms that damage property, cause deaths, and inflict injuries. This paper reviews valuation studies of the protective service of ECEs in terms of reducing flood damages. Although the number of studies have grown significantly, there is still a need for a greater range of studies in more locations and for a wider variety of ecosystems. This review also examines, from an economic perspective, the issues and challenges surrounding estimating the protective benefits of ECEs, as exemplified by some of the recent valuation studies. Recent developments in valuation methods are summarized and critically reviewed. Important challenges remain in valuing coastal ecosystems as a defense against flood damages. The review discusses two of them, such as how protective benefits are subject to spatial variability and dependent on connectivity across “seascapes.” These challenges, along with analyzing the multiple benefits of estuarine and coastal ecosystems, are important areas of future research priority.

**Keywords:** estuarine and coastal ecosystems, marsh, mangroves, storm protection service, economic valuation, wave attenuation

## INTRODUCTION

The rapid loss of estuarine and coastal ecosystems (ECEs) globally has focused attention on their role in protecting coastal communities from storms that damage property and cause deaths and injury. It is now well-documented that many of these habitats provide such protection (Koch et al., 2009; Loder et al., 2009; Wamsley et al., 2010; Gedan et al., 2011; Paul et al., 2012; Armitage et al., 2019). These include specific studies of marshes (Shepard et al., 2011; Rupprecht et al., 2017), mangroves (Cocharde et al., 2008; Zhang et al., 2012; Dasgupta et al., 2019; Montgomery et al., 2019), near-shore coral reefs (Ferrario et al., 2014; Reguero et al., 2018) and seagrass beds (Paul et al., 2012; Christianen et al., 2013; Ondiviela et al., 2014). This protective value of ECEs is increasingly used to justify coastal conservation and restoration efforts worldwide (Temmerman et al., 2012; Arkema et al., 2013; Duarte et al., 2013; Barbier, 2014; Elliott et al., 2016; Narayan et al., 2016; Ruckelshaus et al., 2016; Hochard et al., 2019; Menéndez et al., 2020; Newton et al., 2020).

Given that ECE conservation and restoration are increasingly advocated for protecting coastal communities from flooding hazards that damage property and cause deaths and injury, there is growing interest in quantifying and valuing such benefits. But despite the importance of this coastal protection service, there are still not many economic studies that have estimated a value for it and geographic coverage is still thin. In addition, questions have been raised about some of the methods used, and whether they are sufficiently robust to serve as a guide for policy

(Barbier, 2007; Kousky, 2010; Arnold, 2013). However, more reliable economic estimates of the protective value of mangrove and marsh systems are emerging. The purpose of this review is to examine, from an economic perspective, the issues and challenges surrounding measuring the protective benefits from ecosystem restoration, as exemplified by some of the recent valuation studies.

The paper begins with an overview of selective economic studies globally that have valued the protection benefit provided by ECEs. The paper then briefly summarizes recent developments in valuation methods that have been employed to estimate coastal protection. It is also important to recognize that ECEs provide other valuable benefits in addition to protection service. Nevertheless, important challenges remain in valuing coastal ecosystems as a defense against flood damages. The paper identifies two of them, such as how protective benefits are subject to spatial variability and dependent on connectivity across “seascapes.” The final section of this paper concludes by discussing how further research can address these challenges in valuing the protective service of estuarine and coastal ecosystems.

## VALUATION STUDIES AND METHODS

### Review of Valuation Studies

**Table 1** lists 41 studies, selected from peer-reviewed academic journals, which value the storm protection service of estuarine and coastal ecosystems (ECEs). This value is estimated for the ability of various ECEs to reduce the flood damage to property and other economic assets, and in some instances the risk of loss of life or injury, from coastal storms. The studies are grouped by type of ECE and geographical location.

The key ecological function that allows ECEs to provide a protection service is their ability to attenuate, or reduce the height, of the storm surges and waves as they approach shorelines, or to buffer winds (see **Table 1**). Both wave attenuation and wind buffering are directly related to the vegetation contained in some ECEs, such as marsh, seagrass beds and mangroves. However, the effects may vary for types and characteristics of hazardous events, the presence of emergent vs. submerged vegetation, and tidal and other seasonal conditions (Koch et al., 2009). For example, studies of wave attenuation by marsh wetlands consistently demonstrate significant wave height reductions per unit distance across marsh vegetation, although most of this wave attenuation effect was measured only for small to moderate waves (Shepard et al., 2011). Montgomery et al. (2019) note that numerous studies have found that mangroves provide effective coastal protection from storm waves, but their research in New Zealand and Florida show that mangroves can also reduce storm surge, which is the temporary increase in water level resulting from the combination of high winds and low atmospheric pressure during a weather event. Their study also shows that the effectiveness of mangroves in reducing surges depends not only on storm characteristics but also the density of the vegetation and the extent and depth of the mangroves along shorelines. Seagrass meadows on their own may provide only limited coastal protection in shallow waters and low wave energy environments, with the most effective protection provided by large, long-living

and slow-growing seagrass species (Ondiviela et al., 2014). In contrast, the coastal protection from near-shore coral reefs can be significant, as it is their reticulated structure that provides a natural barrier to storm waves (Koch et al., 2009; Ferrario et al., 2014; Reguero et al., 2018).

Bathymetric effects, such as from sediment trapping and sedimentation that cause shorelines to become higher, are additionally important factors for the wave attenuation function of marshes (Koch et al., 2009; Loder et al., 2009; Wamsley et al., 2010; Rupprecht et al., 2017; Armitage et al., 2019). Sea-to-land shoreline elevation also contributes to the wave attenuation function provided by coastal landscapes populated by mangroves (Alongi, 2008; Cochard, 2011; Laso-Bayas et al., 2011; Armitage et al., 2019). For example, Alongi (2008) suggests that the extent to which mangroves offer protection against catastrophic natural disasters, such as tsunamis, may depend on a range of structural features and conditions within the mangrove ecosystem, such as width of forest, slope of forest floor, forest density, tree diameter and height, proportion of above-ground biomass in the roots, soil texture and forest location (open coast vs. lagoon).

In recent years, there have been a growing number of economic studies estimating the protective value of ECEs, especially for marsh and mangroves (see **Table 1**). Of the 41 studies listed, 31 have been published since 2010. In addition, estimates for coral reefs are starting to emerge. In contrast, few studies have valued the coastal protection benefits of seagrass meadows, which are more effective in shoreline stabilization than attenuation of large waves (Paul et al., 2012; Christianen et al., 2013; Ondiviela et al., 2014). Many additional studies for ECEs exist other than those listed in **Table 1**; however, especially for some of the earlier efforts, there have been problems in the reliability of the valuation methods employed (Barbier, 2007; Kousky, 2010; Arnold, 2013; Chaikumbung et al., 2016; Vedogbeton and Johnston, 2020).

Geographical coverage is also limited, with most valuation studies occurring in the United States and tropical Asia. This is not surprising, given that since Hurricanes Katrina, Rita, Sandy, and Harvey in the United States, the Indian Ocean tsunami in South and Southeast Asia, and Typhoon Haiyan in the Philippines, there has been increasing alarm that the loss ECEs in these regions has made their coastal areas and communities more vulnerable. But as the concern about damaging and life-threatening storms in all low-lying coastal areas grows, there are likely to be more studies in other parts of the world. For example, in Europe, the recognition that tidal marshes and other habitats provide protection against coastal flooding has led to increased studies of the potential wave attenuating function of these ECEs (Liquete et al., 2013; Guisado-Pintado et al., 2016; Schoutens et al., 2019).

Finally, because of the growing interest in the storm protection service provided by estuarine and coastal ecosystems, global analyses of this benefit are beginning to emerge. **Table 1** lists one example for coral reefs (Beck et al., 2018) and two for mangroves (Hochard et al., 2019; Menéndez et al., 2020). For example, Beck et al. (2018) estimate that the absence of the protective benefit of coral reefs would double the annual expected damages from flooding globally and triple the costs from frequent

**TABLE 1** | Examples of studies that estimate the protective value of estuarine and coastal ecosystems.

Ecosystem structure and function	Ecosystem service	Valuation examples (chronologically by location)
Attenuates and/or dissipates waves, buffers wind	Protection of coastal communities against property damage, loss of life and/or injuries, risk.	(Badola and Hussain, 2005; Das and Vincent, 2009; Das and Crépin, 2013, mangroves, India) (Sathirathai and Barbier, 2001; Barbier, 2007, 2012; Barbier et al., 2008; Barbier and Lee, 2014, mangroves, Thailand) (Farber, 1987, 1996; Landry et al., 2011; Petrolia and Kim, 2011; Barbier et al., 2013; Kim and Petrolia, 2013; Barbier and Enchelmeyer, 2014; Petrolia et al., 2014; Boutwell and Westra, 2016, marsh and coastal wetlands, Louisiana, United States) (Rezaie et al., 2020, marsh, New Jersey, United States) (Highfield et al., 2018; Atoba et al., 2020, coastal wetlands, Texas, United States) (Boutwell and Westra, 2015, coastal wetlands, US Gulf Coast) (Narayan et al., 2017, marsh, northeastern United States) (Costanza et al., 2008, marsh, US Atlantic and Gulf Coasts) (Vázquez-González et al., 2019, coastal wetlands, Mexico) (King and Lester, 1995; Mangi et al., 2011, marsh, United Kingdom) (Liu et al., 2019, coastal wetlands, China) (Ouyang et al., 2018, coastal wetlands, China and Australia) (Huxham et al., 2015, mangroves, Kenya) (Menéndez et al., 2018, mangroves, Philippines) (Mahmud and Barbier, 2016, mangroves, Bangladesh) (Laso-Bayas et al., 2011), mangroves, Aceh, Indonesia) (del Valle et al., 2020), mangroves, Central America) (van Zanten et al., 2014, coral reefs, U.S. Virgin Islands) (Wilkinson et al., 1999, coral reefs, Indian Ocean) (Pascal et al., 2016, coral reefs, Caribbean and Pacific) (Beck et al., 2018, coral reefs, global) (Hochard et al., 2019; Menéndez et al., 2020, mangroves, global) (Rao et al., 2015; Narayan et al., 2016, estuarine and coastal ecosystems, global)

The 41 studies selected are from peer-reviewed academic journals only, and are grouped by type of estuarine and coastal ecosystem and its location.

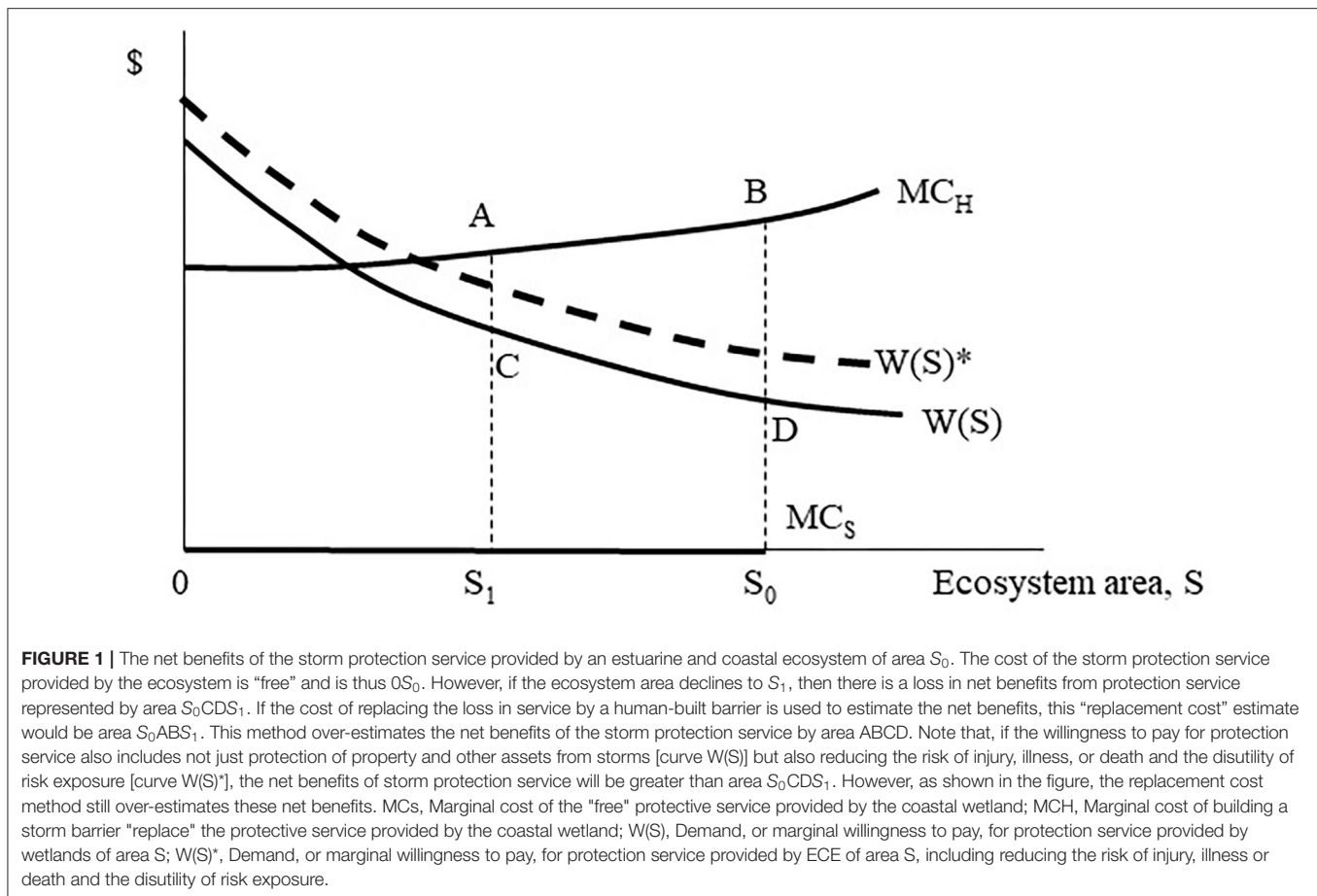
storms. Improved reef management would especially benefit Indonesia, Philippines, Malaysia, Mexico, and Cuba, with each country reducing annual flood damages by at least \$400 million. Hochard et al. (2019) analyze the impact of mangrove extent in protecting economic activity in coastal regions from cyclones over 2000 to 2012 for nearly 2,000 tropical and sub-tropical communities globally. For a community with an average cover of 6.3 m of mangroves extending inland from the seaward edge, direct cyclone exposure can reduce economic activity permanently by 5.4–6.7 months, whereas for a community with 25.6 m of mangroves extending inland from the shoreline, the loss in activity is 2.6–5.5 months. Menéndez et al. (2020) value the global flood protection benefits of mangroves at over \$US 65 billion per year, and estimate that the loss of all mangroves would mean that 15 million more people worldwide would be susceptible to annual flooding. The countries benefiting the most include the United States, China, India, Mexico, Vietnam, and Bangladesh.

## Economic Valuation Methods

As can be seen from the valuation studies in **Table 1**, as the number of studies valuing the protective value of ECEs has increased, important developments have occurred in the methods used to estimate the protective value of estuarine and coastal ecosystems (ECEs). Many of the early studies employed the replacement cost method to value the storm prevention and flood mitigation services, which involves estimating the costs of constructing physical barriers to perform the same services

provided for free by ECEs (King and Lester, 1995; Sathirathai and Barbier, 2001; Mangi et al., 2011; Huxham et al., 2015; Narayan et al., 2016). However, as a valuation method, there are two overall limitations to this replacement cost approach. First, it estimates a benefit (e.g., storm protection) by a cost (e.g., the expenses incurred for constructing seawalls, breakwaters, dykes, groins and other physical structures), and second, human-built structures are not always cost-effective as an alternative to ECEs in providing the same level of coastal protection benefit (Barbier, 2007; Freeman et al., 2014; World Bank, 2016; Kousky and Light, 2019).

The limitations of employing the replacement cost method to value the protective benefit of an ECE are illustrated in **Figure 1**. Assume that the initial landscape area of a marsh or mangrove is  $S_0$ . Because the ecosystem provides this service for “free,” there is no cost, and thus it corresponds to the horizontal axis  $0S_0$ . However, suppose conversion causes some of the ECE area to decrease to  $S_1$ . The replacement cost method would value any subsequent loss in protection benefit by the additional cost of “replacing” it with seawalls, breakwaters, levies and other human-built structures to reduce storm surge and waves. However, the additional—or marginal cost—of building more and more structures to provide coastal protection is likely to rise as the level of protection increases. In **Figure 1**, the marginal cost of building such a physical storm barrier is  $MC_H$ . Thus, the “replacement cost” of using the human built barrier to provide the same storm protection service as the  $S_0S_1$  amount of wetlands lost is measured by the difference between the two cost curves, or



area  $S_0ABS_1$ . However, this cost difference is not measuring the benefit of having the wetlands provide the storm protection service. Instead, this benefit is represented by the demand curve, which indicates how much extra individuals are willing to pay for the additional protection provided by having more wetlands. This demand curve is represented by  $W(S)$  in **Figure 1**. Consequently, if  $S_0S_1$  amount of wetlands is converted, the loss in net benefit is the difference in the demand for protection that would have been met by that amount of wetlands, less the costs of the wetlands providing this service (which as noted previously is “free”). In **Figure 1**, this net benefit corresponds to area  $S_0CDS_1$ . Thus, the replacement cost method overestimates the net benefits of the storm protection service by area ABCD.

More recently, some valuation studies of the protective value of estuarine and coastal ecosystems (ECEs) have developed the expected damage function approach as an alternative to the replacement cost method (Barbier, 2007; Barbier and Enchelmeier, 2014; World Bank, 2016; Beck et al., 2018; Highfield et al., 2018; Menéndez et al., 2020; Rezaie et al., 2020). This method assumes that an ECE provides a non-marketed service, such as “protection” of economic activity, property and even human lives, which benefits individuals through limiting damages. Consequently, the expected damage function adopts the production function methodology of valuing

the environment as an input into the production of a final benefit, which is the protection of human lives, property or economic activity (Barbier, 2007). Utilizing this approach requires modeling how an ECE provides the “production” of this protection service, and then estimating its value of this environmental input or service in terms of the expected damages avoided to property, lives or activity. For example, suppose in **Figure 1**, the benefits of marsh or mangrove are from reducing flood damage to coastal property, and the loss of the wetland will increase the willingness to pay to avoid these damages as represented by the demand curve in the figure. When applied correctly, the expected damage method will yield the true net benefit of this service in terms of protecting coastal property and other assets from flood damages, which in **Figure 1** is denoted by area  $S_0CDS_1$ .

Barbier (2007) estimates the welfare impacts of a loss of the storm protection service due to mangrove deforestation in Thailand by both the expected damage function approach and the replacement cost method, and finds that the benefits estimated by the latter method are eight times more than those of the expected damage function approach. Similarly, Narayan et al. (2016) compare the cost of building submerged breakwater compared to natural-based defense provided by mangrove restoration projects. They estimate that the costs of building

artificial breakwaters is on average five times more expensive (ranging from 3.1 to 6.9 times expensive across the sample) in providing the same level of storm protection as restored mangroves. Increasingly, it is recognized that in remote and inaccessible sheltered bays where mangroves are normally found, artificial barriers, breakwaters, and seawalls are not the least-cost options for providing storm protection benefits, especially when compared to conserving existing mangrove forests or restoring them.

However, with respect to mangrove restoration for coastal protection, other considerations are also important. For example, there have been problems with the restoration success of mangrove replanting schemes, especially for the large-scale programs in the Philippines, Thailand, and elsewhere throughout South East Asia that have been instigated in response to major storm events in the regions (Primavera et al., 2016; Thompson, 2018; Lee et al., 2019). These issues include poor long-term survival rates of afforested or reforested mangroves, the over-reliance on area-based planting targets over long-term ecosystem restoration, and planting at sites that are unsuitable for mangroves. As a study in Central Philippines reveals, the result is that mangrove plantations and reforested areas are significantly less reliable in providing coastal protection compared to natural forests (Primavera et al., 2016).

As **Table 1** indicates, the expected damage function method is increasingly used in many studies that value coastal protection provided by ECEs. However, under certain conditions, this approach may under-estimate this benefit. When households living in coastal areas are risk averse, the expected damage function may not necessarily capture the entire *ex ante* willingness to pay to reduce or avoid the risk from storm damages from ECE protection (Barbier, 2016). Instead, the reduction in expected storm damages to, say, coastal property may be only one component of the marginal willingness-to-pay (WTP) associated with greater protection against storms. This *ex ante* WTP will also depend on avoiding or lowering the risks associated with the storm, such as the threat of death, illness or injury or the general dislike of violent storms, which may be substantial for risk-averse households. Nevertheless, despite its limitations, the expected damage function is a direct compensation surplus measure for estimating an important component of the protective value of ECEs, and thus can be considered a lower-bound estimate of this benefit. This is illustrated in **Figure 1**, where the demand, or marginal willingness to pay, for protection service provided by ECE of area  $S$ , is now the dashed  $W(S)^*$  curve, as it includes reducing the risk of injury, illness or death, and lowering the disutility of risk exposure.

Very few studies are able to estimate this entire marginal willingness to pay for the protective benefit of ECEs. The studies that do estimate *ex ante* willingness to pay often employ survey-based methods, and have difficulty distinguishing the various components that comprise this storm protection value (Barbier, 2016). Some willingness to pay estimates for an ECE intervention that might reduce future storm event risks may include other values as well. For example, Landry et al. (2011) estimate that the average U.S. household is willing to pay \$103 to reduce future flood risk in New Orleans through coastal restoration,

but this value may also reflect concern by these households over the past devastation caused by the 2005 Hurricane Katrina to New Orleans.

Some studies have documented the role of ECEs, notably mangroves, in reducing storm-related deaths after major events. For example, one estimate indicates that, during the 1999 cyclone in Orissa, India, there would have been 1.72 additional deaths per village within 10 km of the coast if mangroves had not been present (Das and Vincent, 2009). Similarly, during the 2004 Indian Ocean tsunami, mangroves, forests and plantations may have decreased loss of life by 3 to 8% in Aceh, Indonesia (Laso-Bayas et al., 2011). In the Philippines, an analysis of 384 coastal villages impacted by flooding from the 2013 Typhoon Haiyan found that the presence of mangroves was significantly correlated with both lower deaths and less structural damage (Seriño et al., 2017).

Other studies have employed survey methods to estimate the entire marginal willingness to pay for storm protection benefits, as represented by the dashed  $W(S)^*$  curve in **Figure 1**. For example, by employing a choice experiment survey for different coastal wetland restoration programs in southeast Louisiana, Petrolia et al. (2014) are able to determine how much a typical U.S. household is willing to pay for different levels of protection as the amount of restored wetland area increases. The average U.S. household is willing to pay \$149 for an intermediate increase in storm surge protection through coastal wetland restoration, but will pay only \$2 more for a further increase to high levels of protection. In a follow-up study of Louisiana households, Petrolia and Kim (2011) find that each household is willing to pay \$111 to prevent future coastal wetland losses. However, households citing storm protection benefits as a top priority were 48% more likely to pay for preventing coastal wetland loss, which allowed the authors to estimate the overall storm protection benefits as \$53 per household.

Finally, it should be pointed out that ECEs and artificial protection may also be complementary at the early stages of restoration efforts, and fully restored ECEs may also reinforce the effectiveness of artificial storm barriers, such as dykes and seawalls. For example, when mangrove tree seeds or seedlings are artificially reintroduced or naturally propagated, both frequent storms and the high energy of tides in coastal zones can prevent the establishment of young mangrove trees in bare sediments (Bosire et al., 2003; Moreno-Mateos et al., 2015). In Vietnam, this problem was solved by establishing bamboo T-fences to reduce coastal erosion and protect the sediment balance necessary for natural regeneration of mangroves (Albers and Schmitt, 2015). At US\$50–60 per meter (m), such low-cost and temporary fencing (they last on average 5–7 years) is a relatively inexpensive way to improve the success of mangrove restoration at its crucial early stages of tree establishment. After successful restoration of sites suitable for mangrove growth, natural regeneration of mangroves will occur and the forest area expand. If artificial dykes are constructed inshore from the restored mangroves, then protection of coastal populations and property from sea level rise and the increasing frequency and intensity of storms is further enhanced.

This is especially important in developing countries such as Vietnam, as the construction of dykes is expensive (US\$ 2,270 per m for a 3.5 m high concrete dyke), and the possibility of increasing dyke height is limited due to the load-bearing capacity of the soil (Albers and Schmitt, 2015). Similarly, in China, a comparison of constructing marsh and other coastal wetlands as an alternative of seawalls for storm protection, led the authors to conclude that “wetlands are a less costly alternative for storm protection” and should be incorporated with seawalls in national coastal defense strategies (Liu et al., 2019). A study for the United Kingdom showed that fronting protective structures with coastal wetlands significantly lowered seawall requirements and resulted in subsequent savings in construction costs (Mangi et al., 2011). Such a combination of “green” and “gray” infrastructure may be the most effective way of protecting vulnerable coasts from the variability of sea level rise, increased frequency and intensity of storms, and the risks of climate change (Mangi et al., 2011; Arkema et al., 2013; Barbier, 2014; Sandilyan and Kathiresan, 2015; World Bank, 2016; Dasgupta et al., 2019; Liu et al., 2019).

## OTHER BENEFITS

Storm protection is only one of the many benefits of conserved or restored ECEs. For example, as noted previously, Petrolia et al. (2014) estimate that the average U.S. household is willing to pay \$149 for increased storm surge protection through coastal wetland restoration in southeast Louisiana, but is willing to pay \$973 per household for restoration when the additional ecosystem benefits of supporting wildlife habitat and commercial fisheries are also included.

The additional ecosystem services of mangroves, which include income and subsistence benefits from collecting products from the mangroves, nursery, and breeding habitats for off-shore fisheries, and carbon sequestration, might be smaller compared to storm protection benefits but important to the overall decision as to whether or not to conserve mangroves or invest in their restoration (Barbier, 2007). In addition, products collected directly from the mangroves and also the artisanal fisheries supported by them may also be important in terms of food security and subsistence needs of local coastal communities (Sarntisart and Sathirathai, 2004; Andrew et al., 2007; Walters et al., 2008; Nfotabong et al., 2009; Béné et al., 2010).

For example, Barbier (2007) estimates that local coastal communities in Thailand gain net present value in income from collecting mangrove products worth \$484 to \$584 per hectare (ha), and an additional \$708 to \$987 per ha in net present value from support provided to coastal fisheries provided by mangroves as breeding and nursery habitat. Such benefits are considerable when compared to the average incomes of coastal households in Thailand. For example, surveys of mangrove-dependent communities reveal that the average household annual income ranges from \$2,606 to \$6,623, and the overall incidence of poverty (corresponding to an annual income of US\$180 or lower) in three out of four villages surveyed exceeded

the average incidence rate of 8% in rural Thailand (Sarntisart and Sathirathai, 2004). If the income to households from collecting mangrove forest products is excluded, then the incidence of poverty would rise to 55.3 and 48.1% in two of the villages, and to 20.7 and 13.64% in the other two communities.

Similar evidence exists of the importance of income from mangroves and other ECEs to support the livelihoods and subsistence of poor households across many low and middle-income countries (Bandaranayake, 1998; Naylor and Drew, 1998; Badola and Hussain, 2005; Walton et al., 2006; Rönnbäck et al., 2007; Walters et al., 2008; Nfotabong et al., 2009; Mukherjee et al., 2014; Hassan and Crafford, 2015; Huxham et al., 2015). In addition, coastal people often associate important cultural values with local ECE habitats that goes beyond their support for economic livelihoods. For example, a study of mangrove-dependent coastal communities in Micronesia has shown that the communities “place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves’ marketable products” (Naylor and Drew, 1998, p. 488). An extensive survey of coastal communities in Papua New Guinea found that people ascribed most importance to ECE services that directly contributed to their livelihoods, especially through food, income and shelter, such as fishing, collecting forest and reef materials, and habitats that support these services (Lau et al., 2019). But the survey also found that communities often placed great importance on local traditions, environmental knowledge and importance for future generations of their stewardship of the environment and ECE services.

## DISCUSSION

Despite the considerable progress in valuing the protective service of ECEs and the growing number of empirical studies, important challenges remain. Here, we discuss two of them: how protective benefits are subject to spatial variability and are dependent on connectivity across “seascapes.”

### Spatial Variability

Increasingly, field studies and experiments indicate that the wave attenuation function of marsh, mangroves and other ECEs, which is critical to their protective value, varies spatially across the extent of these habitats (Madin and Connolly, 2006; Koch et al., 2009; Loder et al., 2009; Wamsley et al., 2010; Gedan et al., 2011; Shepard et al., 2011; Zhang et al., 2012; Rupprecht et al., 2017; Schoutens et al., 2019). This implies that, as storm waves travel across the extent of ECE landscape, the force and magnitude of the waves are increasingly dissipated. Equally, the strength and duration of the storm, and the presence of high or low tides, can impact wave attenuation by ECEs significantly (Koch et al., 2009; Loder et al., 2009; Wamsley et al., 2010; Barbier et al., 2011). Only recently are valuation studies taking into account spatial and temporal variability of wave attenuation by ECEs in estimating their potential protective value (Barbier et al., 2008; Barbier, 2012; Dasgupta et al., 2019; Hochard et al., 2019).

For example, storm surge modeling for the US Gulf Coast of southeastern Louisiana indicates how the attenuation of surge by wetlands is affected by the bottom friction caused by

vegetation, the surrounding coastal landscape, and the strength and duration of the storm forcing (Wamsley et al., 2010). Early studies of the protective value of Gulf Coast wetlands in reducing flood damages do not incorporate such factors (Farber, 1996; Costanza et al., 2008). However, more recent studies of this storm protection benefit do incorporate simulations from storm surge modeling that account for the hydrodynamic properties of surges and the influences of varying wetland landscape and vegetation conditions (Barbier et al., 2013; Barbier and Enchelmeyer, 2014).

Similarly, assessments of how well-mangroves and other coastal forests offered protection against the damages and casualties caused by the 2004 Indian Ocean tsunami found that important landscape and spatial characteristics, such as the variations in coastal topography, shoreline slope, distance of villages to shore and other coastal features, were important factors influencing on protection (Cochard, 2011). For example, Laso-Bayas et al. (2011) found that the presence of coastal vegetation significantly reduced the casualties caused by the tsunami in Aceh, Indonesia, and the most important factor in determining casualties and infrastructure damage was the distance of villages from the coast.

## Connectivity

Because estuarine and coastal ecosystems occur at the interface between the coast, land, and watersheds, there is a high degree of “interconnectedness” or “connectivity” in the land-sea interface across these systems. The term *seascape* is now widely used to refer to spatial mosaics of interconnected coastal and near-shore marine habitat types, such as mangroves, saltmarsh, seagrasses and coral reefs, as the *connectivity* between and among these coastal and near-shore marine habitats is the most pronounced (Moberg and Rönnbäck, 2003; Harborne et al., 2006; Boström et al., 2011; Pittman et al., 2011; Olds et al., 2016; Pittman, 2017). This connectivity, in turn, reinforces and augments the storm protection service provided by each of these ECEs individually.

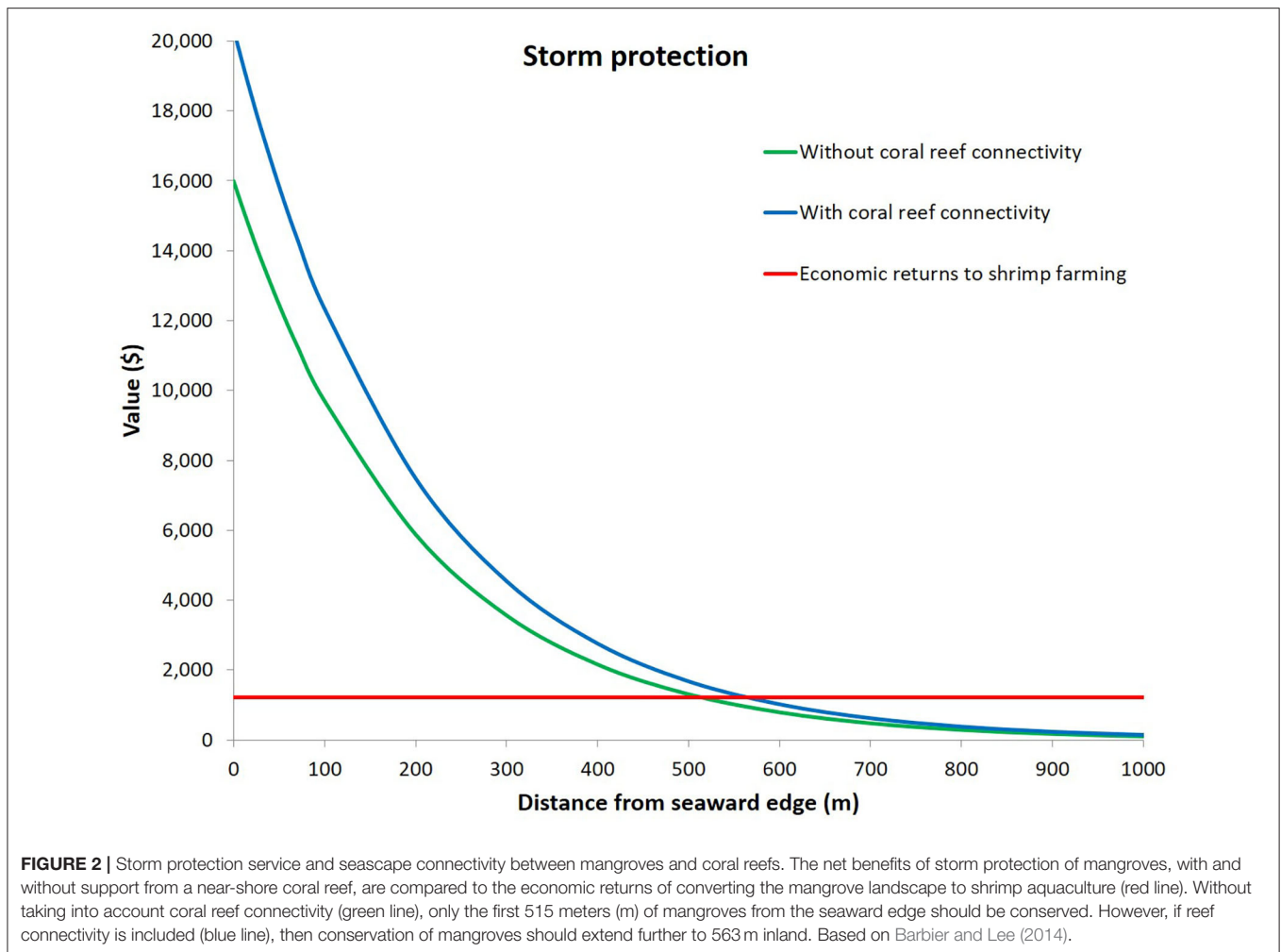
For example, Alongi (2008) has pointed out that the storm protection provided by mangroves will be enhanced further by the presence of foreshore habitats, such as coral reefs, seagrass beds, and dunes. Koch et al. (2009) also note similar cumulative effects occur for attenuating waves that cross seascapes containing coral reefs, seagrasses, and marshes. Modeling simulations based on a Caribbean reef-seagrass-mangrove seascape illustrate that the storm protection service provided by the entire seascape is greater than for either of the three ECE habitats on their own (Sanchirico and Springborn, 2012). Mumby and Hastings (2008) also find that mangroves and coral in the Caribbean not only combine to protect coastlines from storms, but in addition, the mangroves help coral reef fish populations recover from the severe disturbances caused by hurricanes and other extreme events. The cumulative effect of storm protection can sometimes be revealed if an important ECE is absent from the seascape. For instance, Sheppard et al. (2005) document how rising coral reef mortality and deterioration in the Seychelles have increased significantly the wave energy reaching shores, whereas health reefs would normally protect coastlines from storm surges.

To provide further insight into the management implications of valuing the storm protection service across a seascape, Barbier and Lee (2014) develop a model of a two-habitat marine system. The model illustrates how the connectivity of two habitats (a near-shore coral reef and a mangrove habitat) comprising the seascape influences protection against coastal flood damages. That is, the presence of coral reefs in the near-shore marine environment attenuates waves thus enhancing the storm protection service of the coastal mangrove habitat. The model also accounts for spatial variation in wave attenuation across the seascape by allow for the storm protection service provided by mangroves to be greater for their seaward as supposed to the inland boundaries. The model was applied to a representative mangrove-coral reef system, in which the mangroves faced irreversible conversion to commercial shrimp farms. The outcome for this development decision when seascape connectivity was taken into account was compared to the outcome when the storm protection service of the mangroves was considered in isolation from the rest of the seascape (i.e., the coral reef).

**Figure 2** illustrates how mangrove-coral reef connectivity across the seascape affects the development decision. As shrimp ponds can be located in any part of the mangroves with little loss of productivity, it is assumed that the returns to shrimp farming is constant across the landscape at a net present value (NPV) of \$1,220 per ha (red line in **Figure 2**). Without considering any connectivity between coral reef and mangrove storm protection, the NPV per ha of this service provided by the mangroves begins at nearly \$16,000 per ha at the seaward edge and declines to \$108 per ha 1 km inland (green line in **Figure 2**). However, taking into account seascape connectivity, the storm protection value is over \$20,000 per ha at the seaward edge and declines to almost \$140 per ha at the inland boundary (blue line in **Figure 2**). Thus, without taking into account coral reef connectivity, it is optimal to conserve the first 515 meters (m) from the seaward edge, and convert the rest to shrimp farms. However, if the enhancement of mangrove storm protection by coral reefs is taken into account, then conservation of mangroves should extend further to 563 m inland.

## CONCLUSION

Due to increasing concerns about sea level rise, climate change and the frequency of coastal storms, there is more interest than ever in the protective value of estuarine and coastal ecosystems (ECEs). As a result, there are a growing number of studies that attempt to estimate this value, for more ECEs around the world. However, as this review has shown, the geographical coverage of these studies is still limited. In addition, valuation has focused mainly on marsh and mangroves. Coral reefs have received more attention in recent years, but there is still a lack of valuation studies of the protective role of sea grass meadows in reducing coastal flood damages, which appear to be more effective in shoreline stabilization than attenuation of large waves (Paul et al., 2012; Christianen et al., 2013; Ondiviela et al., 2014). There is clearly a need for a greater range of studies for different



locations and a wider variety of ecosystems. As Newton et al. (2020) point out, the continuing loss and degradation of coastal wetlands globally are causing ongoing declines in a wide range of ecosystem services, of which the protection service of ECEs is most prominent.

There have been considerable improvements in the valuation methods used to estimate the benefits of ECEs in reducing coastal flood damages. However, there is still too much reliance on the use of less reliable approaches, such as the replacement cost method, which is likely to lead to inflated estimates. Increasingly, studies are valuing the protective service of ECEs more directly, in terms of reducing the expected damages to property and other assets. This value may be an under-estimate of the full benefit of this service, which should also include reducing the risk of injury, illness or death and lowering the disutility of risk exposure.

Improving the reliability and overall methods of valuing the protective role of ECEs is important, given concerns over estimates of such benefits are sufficiently robust to serve as a guide for policy (Barbier, 2007; Kousky, 2010; Arnold, 2013). Overcoming such concerns through better valuation of the protective benefits of ECEs is especially

important for meeting the management challenge of convincing policy makers and other local stakeholders that such “natural defenses” have a role in coastal zones (Kousky, 2010). In addition, valuing the benefits of ECEs in reducing coastal flood damages can aid in the development of more innovative policies to promote the conservation and restoration of these coastal habitats, such as using insurance to protect ECEs and including their protective value to guide buyouts of flood-damaged property (Kousky and Light, 2019; Atoba et al., 2020).

In addition, the storm protection benefit of ECEs may be just one of many important benefits provided by these systems. Nevertheless, many studies confirm that the protective value of ECEs are one of the more significant benefits sacrificed when these habitats are lost or degraded. Global assessments for both coral reefs and mangroves are also illustrating the economic significance of this protective benefit (Beck et al., 2018; Hochard et al., 2019; Menéndez et al., 2020).

Better understanding of how various ECEs attenuate waves and buffer winds has helped in the development of methods for assessing the protective benefits of these



ecosystems. For example, for marsh and mangroves, an important contribution has been to distinguish between the role of vegetation and other ECE attributes in storm protection compared to coastal topography and near-shore bathymetry. Improved hydrodynamic modeling of the storm surge and wind characteristics of various storm events and their interaction with ECE landscape characteristics has also been insightful. An interesting challenge for future research is to account for the connectivity across ECE habitats, such as mangroves, saltmarsh, seagrasses and coral reefs, to assess their cumulative influence on the wave attenuation and wind buffering functions underlying the protection of coastlines against storms and floods. Only recently have valuation studies begun to model this connectivity and assess how it impacts the protective service provided by an entire seascape of ECEs.

Finally, although this paper has focused mainly on the storm and flood protection benefit of ECEs, one should not forget the multiple benefits provided by these natural systems. This array of benefits are what give ECEs an important advantage compared to human-made structures that are built solely to protect coastlines. Consequently, decisions as to whether or not to invest in ECE restoration either in combination with or as an alternative to human-made structures should not be based solely on their storm protection service alone but should take into account all the economic benefits provided by ECEs as well. Such considerations are important to long-term coastal restoration and protection and restorations. A good example is the Master Plan for the Louisiana Coast, which combines human-built coastal defenses and creating or maintaining over 2,000

km<sup>2</sup> of marsh and other coastal land over the next 50 years to provide storm protection and other ecosystem benefits [(Coastal Protection and Restoration Authority of Louisiana (LCPRA), 2012, 2017; Barbier, 2014)]. Even when the focus is exclusively on storm protection benefits, it is clear that ECE protection and restoration have an important role. For example, Arkema et al. (2013) have shown that substantial ECE restoration along the U.S. coast could reduce significantly the vulnerability of populations and property to future natural disasters as well as to sea-level rise. As the studies reviewed here suggest, many important coastal management decisions over the coming years will depend on improving the valuation of the protective service of ECEs, as well as assessing other significant benefits provided by these critical habitats of the land-sea interface.

## AUTHOR CONTRIBUTIONS

EB designed the review, wrote the manuscript, and approved the submitted version.

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**Conflict of Interest:** The author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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