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The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm

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Abstract For more than a century, coastal wetlands have been recognized for their ability to stabilize shorelines and protect coastal communities. However, this paradigm has recently been called into question by small-scale experimental evidence. Here, we conduct a literature review and a small meta-analysis of wave attenuation data, and we find overwhelming evidence in support of established theory. Our review suggests that mangrove and salt marsh vegetation afford context-dependent protection from erosion, storm surge, and potentially small tsunami waves. In biophysical models, field tests, and natural experiments, the presence of wetlands reduces wave heights, property damage, and human deaths. Meta-analysis of wave attenuation by vegetated and unvegetated wetland sites highlights the critical role of vegetation in attenuating waves. Although we find coastal wetland vegetation to be an effective shoreline buffer, wetlands cannot protect shorelines in all locations or scenarios; indeed large-scale regional erosion, river meandering, and large

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tsunami waves and storm surges can overwhelm the attenuation effect of vegetation. However, due to a nonlinear relationship between wave attenuation and wetland size, even small wetlands afford substantial protection from waves. Combining man-made structures with wetlands in ways that mimic nature is likely to increase coastal protection. Oyster domes, for example, can be used in combination with natural wetlands to protect shorelines and restore critical fishery habitat. Finally, coastal wetland vegetation modifies shorelines in ways (e.g. peat accretion) that increase shoreline integrity over long timescales and thus provides a lasting coastal adaptation measure that can protect shorelines against accelerated sea level rise and more frequent storm inundation. We conclude that the shoreline protection paradigm still stands, but that gaps remain in our knowledge about the mechanistic and context-dependent aspects of shoreline protection.

1 Introduction

Coastal populations benefit from the marine environment, but their proximity to the sea also carries serious risk to human life and property. More than one third of the world's population lives in coastal areas and small islands (UNEP 2006), and more than 10% of people live within 10 m of sea level (McGranahan et al. 2007). Rising sea level is making populations in low-lying coastal areas increasingly vulnerable to catastrophic floods and coastal erosion (IPCC 2007; McGranahan et al. 2007; FitzGerald et al. 2008).

Decades of research on how wetland plants shape coastal geomorphology and can be used for coastal engineering (Shaler 1886; Redfield 1965; Chapman 1974) suggest that coastal marshes and mangroves have the capacity to protect shoreline communities from storm and erosion damage, a hypothesis formally communicated in 1971 following a catastrophic storm on the coast of Bangladesh (Fosberg 1971). Understanding coastal wetlands' ability to protect shorelines is critical to account for the full cost of wetland degradation and the value of restoration (Barbier 2007). Because wetland protection is a low cost alternative to barrier construction, the protection and restoration of coastal wetlands can be a cost-effective approach for rural communities to reduce storm damage (Walton et al. 2006; Halpern et al. 2007; Costanza et al. 2008) that does not conflict with additional protection methods, such as early warning systems (Das and Vincent 2009). Moreover, coastal wetlands provide multiple benefits for local coastal communities beyond just storm protection, such as support for fisheries, wood and non-wood products, and tourism opportunities (Barbier 2007).

Yet the ability of coastal wetlands to attenuate waves and protect coastal communities from storm damage has been called into question. After the devastating Indian Ocean Tsunami of December 2004, which resulted in the loss of over 250,000 lives and several billion dollars in damage, international attention focused on improving shoreline protection for tsunami-vulnerable areas (GCRMN 2006), and scientists observed that intact mangrove forests shielded villages from the worst destruction and that mangrove deforestation magnified storm damage and loss of life (Danielsen et al. 2005; Braatz et al. 2007; Cochard et al. 2008). With the resulting push for mangrove protection and reforestation (Barbier 2006), concerns were raised about the degree to which mangrove forests, rather than correlated variables such as topography and shoreline geomorphology, were responsible for the reduced damages (Chatenoux and Peduzzi 2007; Kerr and Baird 2007; Kerr et al. 2009). In 2005, Hurricanes Katrina and Rita on the Gulf of Mexico coast of the United States caused over 1,500 deaths and prompted the same concern over reduced shoreline protection due to salt marsh habitat loss and degradation (Stokstad 2005; Day et al. 2007) and the same skepticism about salt marsh's capacity to mitigate storm damage (Feagin 2008).

Recent small-scale flume and field experiments have also challenged the paradigm that wetland vegetation reduces shoreline erosion. Comparing small wave (<10 cm) erosion of salt marsh with and without vegetation in a water flume and from plots at the margin of a rapidly eroding salt marsh, Feagin et al. (2009) concluded, "common salt marsh plants do not significantly mitigate the total amount of erosion along a wetland edge."

In this review, we address these concerns by assessing the evidence for shoreline protection by salt marshes and mangroves. We first describe the risks of the sea to coastal communities, and then the mechanisms by which coastal wetlands reduce these risks. With a literature review and meta-analysis, we challenge recent conclusions about shoreline protection services provided by coastal wetlands, namely: (1) that wetland vegetation does not contribute to shoreline protection and (2) that aboveground wetland plant structure plays a limited role in reducing wave heights and erosion. Finally, we discuss the future of wetland shoreline protection services with respect to predicted changes in climate and sea level rise and the development of new approaches, such as the pairing of wetland restorations with man-made reef mimics for greater shoreline protection services.

2 Risks of the sea to coastal communities

Coastal communities face serious risks from the sea that result in damage to human property and loss of life (Fig. 1). For example, shoreline erosion from small wind waves and larger storm and tsunami waves compromises coastal property (Fig. 1, Friedman et al. 2002), and inundation from storm surge and tsunami cost billions of dollars in damage (Fig. 1) and thousands of lives annually (GCRMN 2006; Costanza et al. 2008).

Erosion, storm, and tsunami shoreline impacts, which can severely damage human communities, are all caused by waves (Fig. 2). Small wind- and boat-generated waves can chronically erode beaches and back-barrier habitats (Zhang et al. 2004), and larger storm surge and tsunami waves can have abrupt and catastrophic effects farther inland.

Storm surge waves are a steady build-up of water resulting from a combination of wind and wave setup, low atmospheric pressure, rainfall, and their interaction with tidal conditions (Gönnert et al. 2001). Storm surges of up to 7.5 m can accompany tropical storms (cyclones, typhoons, and hurricanes), and smaller surges of up to 5 m occur in sub-tropical and temperate regions (Gönnert et al. 2001).

In contrast to the steady rise of water during storm surges, tsunami are faster and often larger wave events that result from seismic activity or landslides in the ocean floor. On rare occasions, tsunami run-up can reach 50 m elevation above mean sea level at shore, although the size of tsunami waves varies greatly depending on the event magnitude, proximity to the event, and local bathymetry (National

Fig. 1 The risks of the living along the shore: **a** Coastal erosion in Rhode Island, USA has severely compromised a home (http://cels.uri.edu/news/nBoothroyd.aspx, URI College of the Environmental and Life Sciences). **b** Boats pushed ashore in the storm surge of hurricane Katrina, September 2005 (NOAA). **c** A U.S. Navy hovercraft brings aid to Meulaboh, Sumatra, Indonesia on Jan. 10, 2005, just days after it was decimated by the 2004 Indian Ocean Tsunami (U.S. Navy)

Geophysical Data Center 2010). In deep water, tsunami waves are imperceptible at the surface; they magnify in height as they reach shore and emerge as a fast tidal flow or bore, frequently arriving without warning and with catastrophic effects on human life and property (Fig. 1c, National Geophysical Data Center 2010).

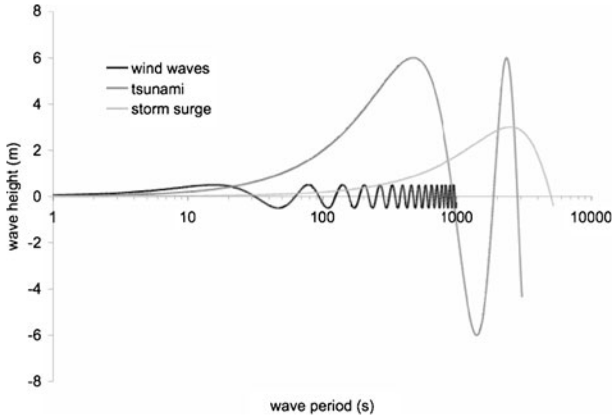

Fig. 2 Generalized hydrograph of wind, storm surge, and tsunami waves. Values are based on period of coastal waves from Gönnert et al. (2001)

Sea level rise, which could exceed 9 mm year^{-1} this century (IPCC 2007), will magnify flooding and erosion risk in low elevation coastal areas (Nicholls et al. 1999; Zhang et al. 2004; McGranahan et al. 2007; Cooper et al. 2008). Nicholls et al. (1999) modeled flooding from future storm surges with moderate sea level rise and coastal population change and predicted that the number of people affected by flooding would increase five-fold by 2080. These predictions assume no change in storm frequency or intensity. Since global climate change may alter tropical storm intensity (IPCC 2007), the increase in affected population size may be underestimated.

3 Mechanisms of shoreline stabilization and wave attenuation by coastal wetlands

In ecology, it has long been recognized that organisms can have both direct and indirect effects on other species and the physical environment, and evidence for either pathway is used to designate biological control over ecosystem processes (e.g. Strauss 1991; Brown et al. 2001; Silliman and Bortolus 2003; Wardle et al. 2004). Correspondingly, coastal wetland plants can affect physical processes on shorelines via both indirect and direct effects. For example, the aboveground portion of plants can directly dampen waves through their structural presence and indirectly dampen wave impacts by stabilizing and building up sediment. In this paper, we carefully consider all mechanisms, both direct and indirect, by which coastal wetland plants protect shorelines, and consider evidence for either as justification for designating wetland plants as significant mitigators of shoreline erosion.

3.1 Direct mechanisms

Aboveground, coastal wetland plants are in direct contact with seawater and waterborne sediment. Plant stems and leaves slow water velocity, reduce turbulence, and increase deposition (Redfield 1972; Christiansen et al. 2000). When water flows through a vegetated canopy, vegetation exerts a drag force counter to the direction of motion. At low stem densities, the drag locally enhances turbulence, causing increased shear stress and potential scour of the bed (Nepf 1999; Bouma et al. 2009). However, under stem densities characteristic of a typical marsh canopy, vegetation reduces turbulence, slows water velocity, and diminishes shear stress near the bed (Leonard and Luther 1995; Nepf 1999). Comparisons between paired vegetated and unvegetated sites indicate that marsh vegetation reduces near-bed water velocity (Neumeier and Ciavola 2004). In fact, basal shear stresses are rarely high enough for sediment entrainment in a vegetated canopy inundated by tidal flow (Christiansen et al. 2000) or wind waves (Carniello et al. 2005). This effect appears to be true whether wetland vegetation is partially submerged or deeply submerged. When deeply submerged, water velocities and shear stress near the bed (those relevant for sediment erosion) remain strongly dampened, and become decoupled from velocities near the water surface (Neumeier and Ciavola 2004). These hydrodynamic impacts tend to reduce sediment erosion (Le Hir et al. 2007) and promote sediment settling (Leonard and Luther 1995; Furukawa et al. 1997; Mudd et al. 2010).

Belowground, plant roots directly slow rates of erosion by stabilizing the soil substrate. The shear strength of wetland soils increases with belowground vegetation biomass since plant roots tend to enhance the cohesion and tensile strength of their substrate (Micheli and Kirchner 2002). Roots also provide a physical barrier between open water and soils, which stabilizes tidal creeks (Mazda et al. 2007; Wolanski 2007). Physical protection against erosion is limited to the depth of the roots, typically 1 m, resulting in greater protection in micro- and meso-tidal estuaries than macro-tidal estuaries where erosion occurs and bank slumping results below the root level (Fig. 3, see also Fig. 13 of Wolanski et al. 2009).

3.2 Indirect mechanisms

In addition to the direct effects of plants on water velocity, sediment deposition, and cohesion, plants indirectly affect coastal hydrodynamics. The belowground contribution of decaying roots enriches soil organic matter, and fine, organic-rich soils tend to erode more slowly than mineral soils in wetlands (Feagin et al. 2009). These

Fig. 3 Mangrove roots cover the upper banks of the Daly Estuary, Australia, providing a protective barrier against erosion of the upper banks, although not protecting against undercutting in the lower banks (E. Wolanski)

properties are strong enough that vegetation-bound tidal channels are significantly narrower than mudflat channels for a given channel depth and discharge (D'Alpaos et al. 2006), and migrate laterally at rates much slower than fluvial channels (Redfield 1972). The morphology of eroding marsh edges also demonstrates the ability of vegetation to slow erosion by strengthening soil. Vegetation favors the formation of vertical scarps and overcut banks, common features of a retreating marsh edge (van de Koppel et al. 2005; Mariotti and Fagherazzi 2010).

Importantly, wetland plants indirectly affect the propagation of waves by building peat and altering coastal bathymetry, a primary control of wave energy (Le Hir et al. 2000). In relatively shallow water, bed friction and sediment entrainment attenuate wave heights as waves propagate landward (Le Hir et al. 2000). Therefore, the maximum height of a wave in shallow water is proportional to the depth of water between the bed surface and sea level. Since near-bed velocities are nonlinearly related to wave height (Friedrichs and Aubrey 1996), wave induced shear stress and sediment erosion rates decline with bed elevation in intertidal environments (Fagherazzi et al. 2006). Since vegetation growth influences the elevation of intertidal surfaces and bed elevations largely determine wave heights in shallow water, plants indirectly determine the dissipation of wave energy.

Once the mudflat rises above mean sea level and intertidal vegetation establishes, plants generate stagnation zones behind them in which fine mineral sediment is deposited, and the accumulation of this sediment further raises the substrate. Mangroves fringing muddy open waters can trap up to 1,000 tons km^{-2} year^{-1} of sediment by this mechanism (Wolanski et al. 1998; Ellison 2009; Woodroffe and Davies 2009). Indeed, mangroves that cover 3.8% of the catchment area of a muddy river can trap 40% of the mud inflow (Victor et al. 2006). For salt marshes, plant-induced sedimentation can also be very rapid, for example, averaging 4.3 mm year^{-1} of vertical accretion in United Kingdom salt marshes, with peak values of 82 mm y^{-1} (Boorman 1996; Wolanski 2007).

In the longer term, decaying plants also build up the substrate by contributing organic matter to the soil profile (Redfield 1972; Mudd et al. 2010). The vertical accretion rate of organic peat is a balance between plant productivity (organic inputs) and the decomposition of organic materials. Over long timescales (decades to centuries), organic and inorganic sediment accretion build vegetated surfaces to elevations that are up to meters higher than they would be in the absence of vegetation (Cahoon et al. 2003; Langlois et al. 2003; Kirwan and Murray 2007; Marani et al. 2007).

The cumulative impact of these mechanisms by which wetland plants promote sediment deposition and inhibit erosion is lasting shoreline stability and accretion (Fig. 4). Consequently, disturbance to vegetation triggers an immediate decline in surface elevation relative to sea level, and recovery of vegetation triggers a rapid increase in elevation (DeLaune et al. 1994; Kirwan et al. 2008). In fact, experimental exclusion of geese on tidal flats along the Fraser River Delta promoted vegetation recovery and a switch from rapid erosion (~10 mm year^{-1}) to rapid vertical accretion (~10 mm year^{-1}) (Kirwan et al. 2008). Similarly, mass plant mortality from a storm caused sediment collapse and rapid conversion to mudflat of vegetated wetlands in the Everglades (Wanless et al. 1994). Therefore, the presence or absence of vegetation can lead to large changes in bed elevation and its effect on wave attenuation and erosion.

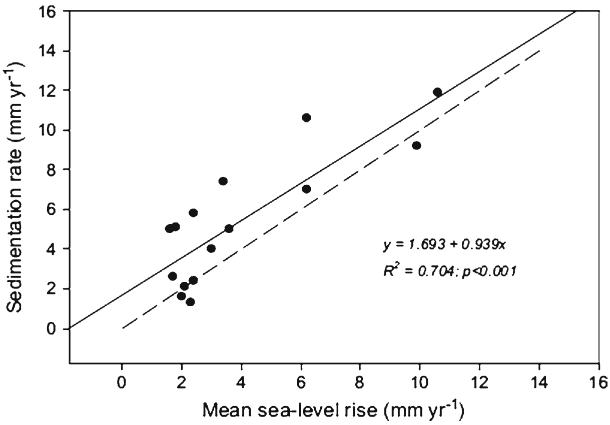

Fig. 4 Sedimentation rate in mangroves is positively correlated with rate of sea level rise, a feedback mechanism that allows for stability and growth of the mangrove during changes in sea level. The *solid line* is a regression line, and the *dashed line* is a 1:1 line. Data were compiled from various studies of sedimentation rate in mangroves. Figure reprinted from Alongi (2008)

4 Evidence of shoreline protection by coastal wetlands

4.1 Protection from coastal erosion

Measurements of erosion in controlled and field environments demonstrate that coastal wetland plants can reduce erosion through direct and indirect mechanisms. In a wave tank, Coops et al. (1996) measured sedimentation and erosion from beds of planted *Scirpus lacustris*, planted *Phragmites australis*, and exposed sediment (i.e. a wetland vegetation addition experiment with controls) and found that *S. lacustris* reduced net erosion rates by 33% and *P. australis* reduced net erosion rates by 82% relative to unvegetated sediments. Contrary to recent findings that substrate material is the key determinant of wetland erosion (Feagin et al. 2009), Coops et al. (1996) found no difference in erosion on sand and silt substrates in the flume tank, suggesting plants reduce erosion through direct (e.g. effects of plant structure on water velocity and sediment) and indirect (e.g. effects of plant decay on soil organic matter) mechanisms in different situations.

Using field surveys and satellite images, Thampanya et al. (2006) examined coastal erosion in southern Thailand, where 50% of mangrove forests have been lost since 1961, and found that coastlines eroded by 0.01 to 0.32 km^2/year from 1967 to 1998. Only coastlines where mangrove forests were intact showed net accretion, and mangrove presence was the most important predictor in a multivariate model of erosion rates (Fig. 5). Notably, deforested coastlines eroded as quickly as coastlines where mangroves were sparse or absent, suggesting that mangrove plant structure, rather than physical variables correlated with mangrove forest development, prevented erosion (Fig. 5). These findings have validated the long tradition of introducing coastal wetland plant species, such as *Spartina anglica* and *Spartina alterniflora*, to stabilize shorelines (Ranwell 1967; Daehler and Strong 1996) and restoring wetlands for shoreline stabilization, despite the often detrimental effects on native biodiversity of introducing non-native species, as in the case of *Casuarina* plantations in India (Feagin et al. 2010) and *S. anglica* in Tasmania (Kriwoken and Hedge 2000).

The stabilizing influence of mangroves against erosion from tidal currents is particularly well illustrated in the Daly Estuary, Northern Australia, where channels migrate through the estuarine flood plains at a rate of up to 25 m year^{-1} (Chappell

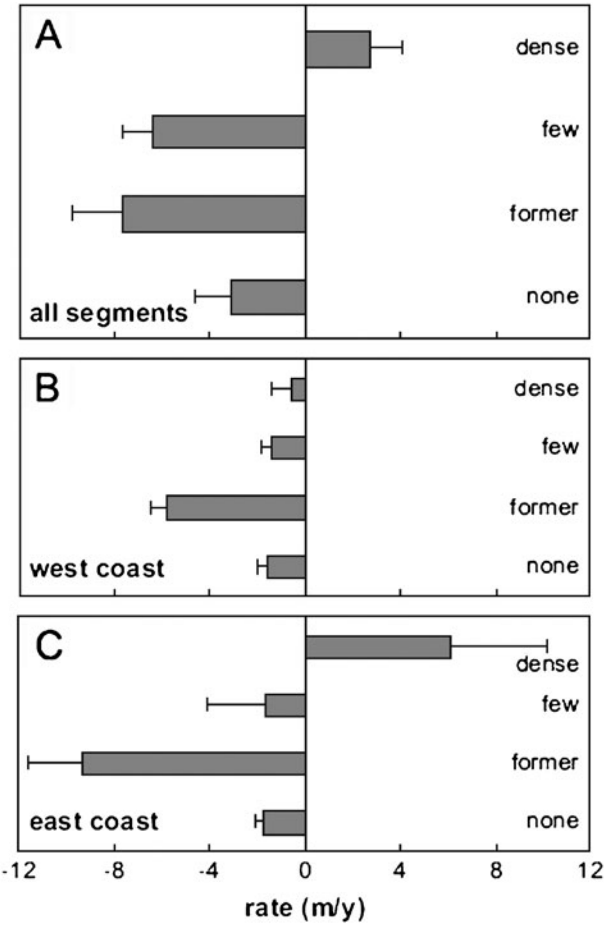

Fig. 5 Trends in shoreline accretion or erosion (meters per year) in coastal southern Thailand in areas of dense, sparse, deforested, and non-mangrove areas, measured in satellite images from 1967–1998. Rates are shown for **a** all coastal areas, **b** the west coast, where mangroves are rare, and **c** the east coast, where mangroves are abundant. Reprinted from Thampanya et al. (2006)

1993), but have been stabilized by mangroves for at least 120 years, which is the extent of historical bathymetric data. Along tidal channels, mangroves form a biological fortress against meandering.

The ability of coastal wetlands to limit erosion, however, is not without bounds. Coastal wetlands are effective at reducing erosion in low wave energy environments, but less so in areas of high wave energy. Large waves can tear up plant rhizomes, expose deeper sediments, and increase erosion (Coops et al. 1996; Cahoon 2006). Although large waves, such as those from storms, can erode wetlands, they can also be a major source of inorganic sediment for wetlands (Cahoon et al. 1996; Turner et al. 2006). In a study of the ability of salt marshes to stabilize the shoreline in different wave environments in the Chesapeake Bay, fringing salt marshes planted in areas with fetch greater than 10 km were quickly eroded (Hardaway et al. 1984). Similarly, in a meta-analysis, Knutson et al. (1981) found that wetland restoration projects in areas of high fetch (>9 km) failed. By modeling marsh presence as a function of wind and wave stress, Roland and Douglass (2005) determined that areas with wave heights predominantly under 0.2 m can support marshes. Successful restoration of coastal wetlands for shoreline protection services will initially depend on an appropriate environmental setting. However, if moderately inappropriate conditions, such as high wave energy or mobile mineral sediments, can be temporarily minimized, for example, by a temporary seaward structure (e.g. reef balls or oyster domes, discussed below) or substrate mat, wetland restorations can often persist after the establishment of plant-sediment feedbacks (Wolanski 2007).

4.2 Protection from storm surge

The storm surge attenuation capacity of coastal wetlands is sometimes given in absolute terms (e.g. "An oft quoted figure is that every kilometer of wetland reduces the storm surge by 7 centimeters..." p. 1266, Stokstad 2005). However, attenuation and shoreline protection, like other ecosystem services, are likely to exhibit notable nonlinearities across time and space (Barbier et al. 2008; Koch et al. 2009).

Important variation and nonlinearities in wave attenuation may result from differences in the identity, phenology, and morphology of wetland species (Koch et al. 2009). For example, dense mangrove forest more effectively attenuates waves than low density forest (Mazda et al. 1997; Massel et al. 1999). Stem stiffness (Bouma et al. 2005; Peralta et al. 2008) and the presence of pneumatophores (Mazda et al. 2006) can also affect attenuation rates. Moreover, wave attenuation may vary even across an entirely homogeneous wetland landscape. For example, greater attenuation often occurs at the seaward wetland margin where there is an abrupt shift in bottom elevation (Koch et al. 2009).

In addition to nonlinear variation in attenuation due to biological characteristics of wetlands, considerable variation in storm characteristics and coastal geology make wave and storm surge attenuation too complex to be described with a simple rule of thumb (see review by Resio and Westerink 2008). Storm duration and track, nearshore bathymetry, and the presence of barrier islands and shallow water marine habitats affect surge dynamics (Suhayda 1997; Gönnert et al. 2001; Resio and Westerink 2008). Steady winds can build storm surge as it progresses inland, regardless of the terrain it is crossing. During Hurricane Rita, surge heights increased across nearly 40 km of salt marsh in Louisiana, as steady winds overwhelmed drag from wetland vegetation (Resio and Westerink 2008). Shoaling can also amplify waves as they enter shallow areas, magnifying wave heights at wetland margins.

Despite these complexities, there are several lines of evidence that suggest that coastal wetlands do reduce inundation from storms in many instances. Hydrodynamic models of storm surges traversing landscapes suggest that vegetation roughness slows and reduces surge. Wamsley et al. (2010) modeled several storms approaching the Louisiana coast across present wetland cover and a predicted future coast with reduced wetland cover and found that wetlands can play a large role in attenuating storm surge (up to 16.6 cm attenuation per kilometer of wetland), but that this effect is dependent upon the characteristics of the wetland and the storm. Field based observations of storm surges traversing wetlands also indicate a dampening effect (Lovelace 1994; Day et al. 2007; Krauss et al. 2009; Wamsley et al. 2010). These effects range from a dampening of 4.4 cm (Hurricane Andrew, Lovelace 1994) to 15.8 cm/km of coastal wetland traversed (Hurricane Charley, Krauss et al. 2009).

Since waves are often a large component of storm surge (Dean and Bender 2006; Resio and Westerink 2008), we conducted a small meta-analysis of wave and storm surge dampening by wetlands. We calculated wave attenuation in five mangrove and ten salt marsh sites from published studies that investigated the propagation of small wind or larger storm surge waves across coastal wetlands (Appendix). These studies were compiled during the extensive (>150 studies) literature review for this paper. Following Mazda et al. (2006), wave attenuation (r) is equal to the proportion of wave height reduction per meter of land traversed. For example, $r = 0.01$ indicates a 1% reduction in wave height per m, suggesting complete attenuation across a 100 m stretch. This approach standardizes measurements of wave attenuation with different initial wave heights and varying traverse distances.

This analysis found similar values of r in marshes and mangroves (one-way ANOVA, $F_{1,13} = 2.67$, $p > 0.1$, Fig. 6a). Wave attenuation was smaller in all coastal wetlands when measured during actual storm surges (mean $r_{\text{storm}} = 0.0001$ vs. mean $r_{\text{wind}} = 0.018$, Fig. 6b), when wave heights and sampled traverse distances were much larger (mean wave height$_{\text{storm}}$ = 164 cm [range 44 to 353 cm] vs. mean wave height$_{\text{wind}}$ = 22 cm [range 4 to 60 cm]; mean traverse distance$_{\text{storm}}$ = 5,863 m [range 900 to 14,100 m] vs. mean traverse distance$_{\text{wind}}$ = 68 m [range 10 to 200 m]). Finally, attenuation was greater across vegetated wetlands than unvegetated mudflats indicating that the vegetation is a critical component for the wave attenuation capacity of coastal wetlands (Fig. 6c).

Among these 15 studies, wave attenuation was nonlinearly and negatively correlated with traverse distance (Fig. 7a). Indeed, attenuation of wind waves and storm surge shows a remarkably consistent pattern across traverse distance (Fig. 7a). Short distances at the seaward margin of wetlands exhibited greater wave attenuation than equivalent landward distances. Across traverse distances of less than 50 m, values of r were greater than 0.01 (1%/m); Knutson et al. (1982), Wayne (1976), and Morgan et al. (2009) all found >60% attenuation of small waves (<0.3 m) in under 20 m.

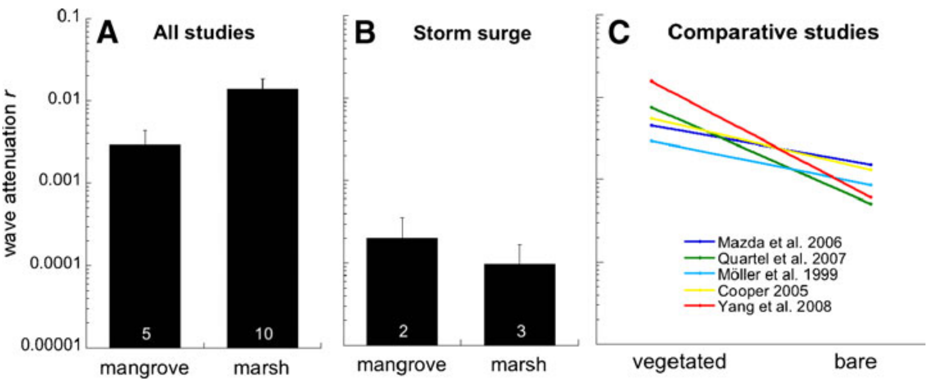

Fig. 6 Attenuation of wind waves and storm surges by marshes and mangroves measured in **a** all fifteen studies (Appendix), **b** observations of storm surge, **c** comparative studies that looked at attenuation of wind waves across vegetated wetland and unvegetated mudflat surfaces. For each study, we calculated the rate of wave attenuation r, described by Mazda et al. (2006). $r = (h_2 - h_1/h_1)/x$, where h_1 is the initial height of the wave and h_2 is the height of the wave after crossing distance x. Note the logarithmic scale of the y-axis

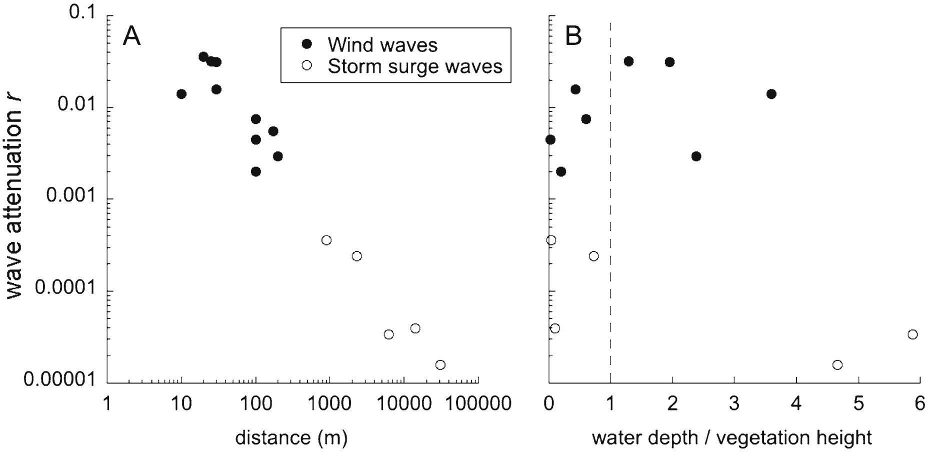

Fig. 7 The relationship of measured wave attenuation *r* with **a** distance across which it is measured and **b** the ratio of water depth to vegetation height. Ratios greater than 1, indicated by a *dashed line*, indicate the wetland vegetation was submerged during measurement. Water depth measurements do not include wave height and can vary with tidal stage; water depth values used are the average water depth for the period of wave attenuation measurement. When published studies did not include vegetation height values, average values for the dominant plant species were obtained from field guides. Studies included ($n = 15$) are in Appendix. Cooper et al. (2008) and Wayne (1976) did not include information on water depth and/or vegetation height and were excluded from panel **b** ($n = 13$). Note the logarithmic scale of axes

This result is consistent with earlier observations of a spatial nonlinearity in wave attenuation by coastal wetlands (Barbier et al. 2008; Koch et al. 2009). Importantly, high attenuation across relatively small traverse distances suggests that even narrow wetlands offer relatively high shoreline protection value (Barbier et al. 2008; Morgan et al. 2009).

In a second exercise, we examined how wave attenuation *r* varied across the ratio of water depth to vegetation height, which indicates how submerged wetland plants are during wave attenuation measurements, and found a second nonlinear relationship (Fig. 7b). Danard and Murty (1994) modeled wave attenuation by partially submerged and totally submerged vegetation, and found theoretical evidence that vegetation would be more effective at dampening waves when only partially submerged. We find preliminary support for this hypothesis: Wave attenuation decreased at high ratios of water depth:vegetation height. Interestingly, wave attenuation values were high even when plants were well-submerged (water depth/vegetation height ratio >1), which agrees with the observation by Neumeier and Ciavola (2004) that water flow was reduced even within a submerged *S. anglica* canopy.

The unimodal relationship that we have found resolves potentially conflicting observations by Danard and Murty (1994) of reduced attenuation rates in submerged vegetation with those of others (Möller et al. 1999; Mazda et al. 2006; Quartel et al. 2007) who found that wave attenuation in wetlands increases with water depth. In our dataset, wave attenuation is minimal when the ratio between water depth and vegetation height is either large or small and greatest at intermediate water depth/vegetation height ratios.

Additional evidence that coastal wetlands protect shorelines comes from a number of studies that indicate that coastal wetlands have had a statistically significant impact in reducing economic damages and deaths associated with major storm events (Badola and Hussain 2005; Barbier 2007; Costanza et al. 2008; Das and Vincent 2009). Mangroves significantly reduced the number of deaths and damages to property, livestock, agriculture, fisheries and other assets during the 1999 cyclone that struck Orissa, India (Badola and Hussain 2005; Das and Vincent 2009). Statistical analysis indicates that there would have been 1.72 additional deaths per village within 10 km of the coast if mangroves had been absent (Das and Vincent 2009). Economic losses incurred per household were greater (US$154) in a village that was protected by a constructed embankment compared to those (US$33) in a village protected by mangrove forests (Badola and Hussain 2005).

An analysis of the economic damages associated with 34 major hurricanes striking the United States coast since 1980 found that coastal wetlands explained 60% of the variation in relative damages inflicted on coastal communities (Costanza et al. 2008). The additional storm protection value per unit area of coastal wetlands from a specific hurricane ranged from a minimum of US$23 per hectare for Hurricane Bill to a maximum of US$463,730 per hectare for Hurricane Opal, with a median value of just under US$5,000 per hectare. Similarly, the contribution of mangrove deforestation to economic damages of storms was estimated for 39 coastal storm events affecting Southern Thailand from 1975 to 2004 (Barbier 2007). Over 1979 to 1996, the marginal effect of a one square kilometer loss of mangrove area was an increase in expected storm damages of about US$585,000 per square kilometer, and from 1996 to 2004, the expected increase in damages from a 1 km^2 loss in mangroves was around US$187,898 per square kilometer.

Although each of these types of data has drawbacks, such as limited representation of the range of storm, site, and wetland characteristics that are important in storm surge attenuation, cumulatively they offer strong support that coastal wetlands provide context-dependent protection of shorelines from storm surge flooding.

4.3 Protection against tsunami

The evidence that coastal wetlands can protect against tsunami impacts is more tenuous because, due to the sudden and catastrophic nature of tsunami events, evidence tends to be anecdotal. However, models and limited observational data do suggest mechanisms by which mangroves may be effective barriers, at least for smaller tsunami waves.

Firstly, models of tsunami hydrodynamics predict that current velocities and wave heights are reduced when tsunami waves traverse a mangrove forest compared to bare land (Hiraishi and Harada 2003; Teo et al. 2009). Additionally, using surveyed morpohological data and damage following the 2004 Indian Ocean Tsunami, Tanaka et al. (2007) modeled vegetation drag forces and found that, of mangroves (*Rhizophora* and *Avicennia* spp.) and other coastal trees (*Pandanus odoratissimus*, *Casuarina equisetifolia*, *Cocos nucifera*, and *Anacardium occidentale*), *Rhizophora* mangroves and *P. odoratissimus* were most effective in slowing water flow and reducing wave heights.

Although observational evidence must be considered cautiously due to the possibility of spurious correlations, several observational studies of the spatial distribution of damage from the 2004 Indian Ocean Tsunami found evidence that mangroves reduced tsunami impacts. In coastal southeastern India, villages located behind mangrove buffers were spared tsunami damage experienced by nearby exposed villages (Danielsen et al. 2005; Kathiresan and Rajendran 2005; Vermaat and Thampanya 2006; Olwig et al. 2007). However, failure of these observational studies to adequately investigate the effects of variation in nearshore bathymetry and elevation has discounted the strength of this evidence (Kerr and Baird 2007; Kerr et al. 2009). Furthermore, small-scale heterogeneity in mangrove forest condition also introduced heterogeneity in the protection mangroves afforded from a small tsunami (<4 m), complicating measurement of the effect of mangroves on protection. In field surveys, Dahdouh-Guebas et al. (2005) found that the condition and species composition of the mangrove forest was correlated with the protection service provided during the Indian Ocean Tsunami. Other types of coastal vegetation, such as clumps of coconut trees and *C. equisetifolia* that survived the tsunami, may have provided additional protection, creating additional small-scale heterogeneity in tsunami protection, a phenomenon commonly neglected by biophysicists sampling the impact of the tsunami (Braatz et al. 2007).

Two anecdotal factors also suggest that mangroves may reduce tsunami damage. For one, mangrove trees are surprisingly resistant to the forceful impacts of tsunami. Although some areas of mangroves were heavily damaged during the Indian Ocean Tsunami (Alongi 2008; Cochard et al. 2008), many mangrove forests experienced minimal damage (Alongi 2008) limited to the seaward forest fringe (Dahdouh-Guebas et al. 2005). That mangroves remained standing during the Indian Ocean Tsunami's onslaught indicates that they absorbed wave energy and, at minimum, acted as physical barriers to debris (Tanaka et al. 2007; Cochard et al. 2008). A second piece of anecdotal evidence is a cultural tradition of planting and maintaining mangrove shelterbelts in regions with tsunami and intense tropical storms. For example, after damaging storms in the Philippines in 1969 and 1971, coastal landowners began planting mangroves to protect their properties, as well as for other benefits (Walters 2003). The perceived benefit of mangroves for coastal protection from tsunami, also observed in Sri Lanka and India after the 2004 Indian Ocean Tsunami (Dahdouh-Guebas et al. 2005, 2006; Braatz et al. 2007), adds support to the hypothesis that mangroves can reduce tsunami damage, at least for smaller events.

In extreme tsunami waves, mangroves cannot survive, and thus, cannot protect communities in their lee. Moreover, uprooted or snapped mangrove debris can be carried by the wave and become itself a hazard (Cochard et al. 2008). Forest density and tree size also appear to play a role in the capacity of mangroves to persist during tsunami impacts. Braatz et al. (2007) describe how a 3 m tsunami wave was able to uproot an isolated 3 m mangrove tree, but as a result of interlocking roots of adjoining trees, a densely vegetated mangrove forest with 6 m tall trees was able to survive a 6 m tsunami wave.

5 The future of wetland shoreline protection services

In the coming decades, accelerated sea level rise will simultaneously increase the need for shoreline protection and change the way that wetlands provide it. By affecting water depth, sea level rise will increase inundation risk to low-lying coastal communities and alter wetland accretion and wetland wave attenuation (e.g. Figs. 4 and 7b). Moreover, due to rising water temperatures, storm intensity is expected to increase concurrent to rapid sea level rise, exacerbating coastal risk (IPCC 2007; McGranahan et al. 2007; FitzGerald et al. 2008).

That coastal wetlands will persist and continue providing shoreline protection services during this period of accelerated sea level rise is uncertain. Forecasts for coastal wetland loss due to sea level rise are grim (e.g. Nicholls et al. 1999; Craft et al. 2009).

However, many models do not incorporate two critical feedback mechanisms, (1) submergence-accretion feedbacks (French 1993; Temmerman et al. 2004) and (2) plant productivity-submergence feedbacks (Morris et al. 2002), which couple accretion rates to rates of sea level rise, such that actual wetland loss may be less severe than predicted (Kirwan and Guntenspergen 2009). Firstly, rising sea level inundates wetlands with a greater volume of sediment-laden water, which facilitates faster accretion when suspended sediment concentrations are adequate (French 1993; Temmerman et al. 2004). Secondly, increased productivity of plants when submerged (Morris et al. 2002) helps maintain the marsh platform (Kirwan and Murray 2007). As a result of these feedbacks, moderately submerged wetlands do not degrade, but recover and persist, resulting in long-term stability and growth of the wetland barrier in times of moderate sea level rise (Fig. 4).

However, coastal wetland accretion could fail to keep pace with accelerated sea level rise if a critical threshold is crossed and vegetation is drowned (Morris et al. 2002; Kirwan and Temmerman 2009; Kirwan and Guntenspergen 2009). Where sea level rise, storm impacts (Michener et al. 1997), or other human impacts, such as habitat conversion (Valiela et al. 2001), degrade wetlands, critically required shoreline protection services will be compromised.

To fully utilize coastal wetlands and their natural positive feedbacks for shoreline protection, conservation and restoration of coastal wetlands are being incorporated into coastal adaptation and risk management plans. In this issue, Francis et al. (2011) use life cycle analysis to explore the shoreline protection benefits of wetland restoration to power-supply infrastructure in an urban area. They find that wetland restoration is not the most cost effective solution for power infrastructure protection. However, the authors use a high wetland restoration cost and note that they do not account for all ecosystem service benefits (e.g. carbon storage and protection of other coastal economic activity and property). Also, the model treats the shoreline protection effect of wetlands as a flat rate of storm surge reduction (m) per km traversed, which does not account for the spatial non-linearity of attenuation. The authors note that wetlands may prove beneficial to infrastructure protection in other scenarios, and we agree with Francis et al. (2011) that future economic models such as theirs can inform decisions about using wetland restoration for shoreline protection of power infrastructure. However, it is also imperative that decisions concerning wetland restoration are not based solely on shoreline protection for power generation facilities and other large-scale coastal infrastructure. As we note previously, various studies indicate that wetlands protect against a wide range of damages and risks from coastal storm events, including mitigating damages to coastal property, agriculture, tourism, and other economic activities as well as reducing the

Fig. 8 New restorations are pairing salt marshes and oyster domes to increase the effectiveness of shoreline protection services (B. Silliman)

risk to human lives (Badola and Hussain 2005; Barbier 2007; Costanza et al. 2008; Das and Vincent 2009).

Beyond traditional wetland restorations, novel approaches, such as the pairing of artificial structures that stimulate colonization by native ecosystem engineers with restored coastal wetlands, are being implemented to enhance shoreline protection. Oyster domes and reef balls (concrete structures that baffle waves and provide habitat structure for benthic ecosystem engineers) are paired with coastal wetlands to more effectively attenuate wind waves and prevent erosion (Fig. 8). The Nature Conservancy is employing reef-wetland pairings for erosion control and ecosystem restoration efforts in seven states in the southeastern USA (R. Brumbaugh, pers. comm.). “Living shoreline” restorations of this type are appealing because they provide the service of hard coastal defense structures (e.g. breakwaters, seawalls) with the ancillary benefits of ecological restoration (Swann 2008) and, in addition, are self-maintaining. Perforated hard structures such as reef balls or Coastal Havens (Swann 2008) promote sedimentation at the wetland seaward margin (Meyer et al. 1997; Piazza et al. 2005), allowing restoration and expansion of coastal wetlands in wave climates that might not otherwise permit traditional wetland restoration. These man-made and nature-made combination approaches will likely generate synergistic ecological benefits (Swann 2008), but to date no experiments have specifically tested for the singular and interactive effect of this combined approach. Evaluating this promising approach should be the focus of much future research.

6 Conclusion

Conservation scientists and managers must constantly evaluate whether current practices and research address current conservation objectives and are guided by up-to-date theory (Lawler et al. 2006). Recently, the paradigm that wetlands provide shorelines protection from wave stress was called into question (Feagin et al. 2009). Contrary to conclusions from that paper, however, we find widespread support for the paradigm that coastal wetlands protect shorelines from erosion and wave damage. Coastal wetland plants interact with water and sediment in a variety of direct and indirect ways that slow water flow, facilitate sediment deposition, increase shoreline cohesion, and build peat. We find ample evidence from modeling, observational, and field studies that, in many instances, wetland plants reduce erosion, storm surge, and even small tsunami wave impacts.

Recent articles concluding that wetland plants do not protect shorelines based on small-scale experiments and single storm events misrepresent the nature of shoreline protection by coastal wetlands. Firstly, both the direct and indirect effects of plants must be considered as biological contributions to shoreline protection. In dually considering direct and indirect effects, we see broad agreement between large and small-scale studies that wetland plants provide shoreline protection through a variety of means. We hope that future research and debates can center on the relative contributions of and variation in direct and indirect mechanisms by which plants protect shorelines.

Secondly, although much attention has focused on the mixed evidence for the role of mangroves in mitigating the 2004 Indian Ocean Tsunami, we advise caution in drawing conclusions from a single event. Across a variety of storms and locations, we find evidence that coastal wetlands generally do provide shoreline protection. However, we find that this ecosystem service is context-dependent and exhibits nonlinear characteristics across space and time. Characterizing the contexts in which wetlands will protect shorelines is fundamental for coastal planning and conservation management. We found that even narrow wetlands can dampen waves, a result that can be used to justify smaller-scale restoration projects. Additionally, we found that wetlands are not likely to be effective wave buffers when water depths are much greater than or a small fraction of vegetation height. Controlled experiments and experiments across environmental gradients can test these and other functional relationships to further our mechanistic understanding of shoreline protection services.

An increasing number of studies indicate that coastal wetlands have had a statistically significant impact in reducing the economic damages or deaths associated with major storm events. Not all these events are large-scale tsunamis or major hurricanes. Smaller storms occur more often and have devastating impacts on coastal areas and populations. It is against these frequent smaller, yet damaging, events to which coastal wetlands likely offer the greatest storm protection. As climate change raises the risk and incidence of economically damaging storms, it is important that we continue to examine the role of coastal vegetation in preventing damages and deaths.

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Appendix

Table 1 Studies included in the meta-analysis

Reference	Wetland type	Dominant species	Wave type	Comparison with unvegetated?	Location
Krauss et al. (2009)[a]	Mangrove	*Rhizophora mangle*	Storm surge		Ten Thousand Rivers, Florida, USA
Krauss et al. (2009)[a]	Mangrove	*Rhizophora mangle*	Storm surge		Everglades National Park, Florida, USA
Mazda et al. (1997)	Mangrove	*Kandelia candel*	Wind waves		Tong King Delta Thailand
Mazda et al. (2006)	Mangrove	*Sonneratia* sp.	Wind waves	Y	Ving Quang, Thailand
Quartel et al. (2007)	Mangrove	*Kandelia candel*	Wind waves	Y	Red River Delta, Vietnam
Knutson et al. (1982)	Marsh	*Spartina alterniflora*	Boat-generated		Chesapeake Bay, Virginia, USA
Lovelace (1994)	Marsh	*Spartina alterniflora*	Storm surge		Louisiana, USA
Wamsley et al. (2010)	Marsh	*Spartina alterniflora*	Storm surge		Louisiana, USA
Krauss et al. (2009)[a]	Marsh	Mixed	Storm surge		Ten Thousand Rivers, Florida, USA
Möller et al. (1999)	Marsh	Mixed	Wind waves	Y	Norfolk, England
Wayne (1976)	Marsh	*Spartina alterniflora*	Wind waves		Adams Beach, Florida, USA
Cooper et al. (2008)	Marsh	Mixed	Wind waves	Y	The Wash, Norfolk, England
Bouma et al. (2005)	Marsh	*Spartina anglica*	Wind waves		Paulina salt marsh, Netherlands
Möller (2006)	Marsh	Mixed	Wind waves		Dengie Peninsula, Essex, England
Yang et al. (2008)	Marsh	*Scirpus mariqueter, Spartina alterniflora*	Wind waves	Y	Yangtze Delta, China

[a]Three records from Krauss et al. (2009) were included: storm surge attenuation in marsh and in mangrove during Hurricane Charley and storm surge attenuation in mangrove during Hurricane Wilma

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