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Marshes and Mangroves as Nature-Based Coastal Storm Buffers

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Abstract

Tidal marshes and mangroves are increasingly valued for nature-based mitigation of coastal storm impacts, such as flooding and shoreline erosion hazards, which are growing due to global change. As this review highlights, however, hazard mitigation by tidal wetlands is limited to certain conditions, and not all hazards are equally reduced. Tidal wetlands are effective in attenuating short-period storm-induced waves, but long-period storm surges, which elevate sea levels up to several meters for up to more than a day, are attenuated less effectively, or in some cases not at all, depending on storm conditions, wetland properties, and larger-scale coastal landscape geometry. Wetlands often limit erosion, but storm damage to vegetation (especially mangrove trees) can be substantial, and recovery may take several years. Longer-term wetland persistence can be compromised when combined with other stressors, such as climate change and human disturbances. Due to these uncertainties, nature-based coastal defense projects need to adopt adaptive management strategies.

1. INTRODUCTION

Coastal and estuarine shorelines and adjacent lowland areas are particularly vulnerable to impacts from storms, such as tropical cyclones (hurricanes and typhoons) or other storm depressions. A storm propagating from the ocean toward a shoreline can generate a storm surge that can raise the sea level by several meters for durations of a few hours to days (Resio & Westerink 2008). Storm waves generate additional sea level fluctuations of up to several meters with a periodicity of seconds to minutes. Together, storm surge and storm waves imply risks of shoreline erosion and flooding of adjacent lowlands, with potentially dramatic impacts, including human fatalities and major damage (e.g., Day et al. 2007, Zhu et al. 2020). Storm-induced coastal hazards will grow in the coming decades due to global climate warming, which accelerates sea level rise and increases storm intensity in certain regions, and due to human population growth and coastal landscape engineering, which further enhance flood risks (e.g., Lin et al. 2012, Neumann et al. 2015, Tessler et al. 2015). Hence, there is a growing need for science-based sustainable strategies for coastal hazard mitigation.

Nature-based approaches are increasingly proposed as an essential element to mitigate storm-induced coastal hazards (e.g., Borsje et al. 2011, Morris et al. 2018, Narayan et al. 2016, Sutton-Grier et al. 2015, Temmerman et al. 2013). Nature-based approaches consist of the conservation, restoration, or creation of ecosystems that can contribute to the mitigation of storm impacts on human infrastructure located landward behind these ecosystems. Two ecosystems that are widely considered important as storm buffers are tidal marshes (occurring in temperate to tropical climate zones) and mangrove forests [occurring in the (sub)tropics]. Below, we use the term tidal wetlands to refer to both marshes and mangroves. As storm surges and waves propagate from open water through marshes or mangroves, the friction between the water motion and the dense wetland vegetation and sediment surface reduces wave heights (e.g., Garzon et al. 2019a, Möller et al. 2014) and to some extent storm surge levels (e.g., Wamsley et al. 2010). Apart from marshes and mangroves, which are the focus of this article, other ecosystems, such as seagrass beds, kelp forests, and coral and shellfish reefs, can play similar roles (e.g., Ferrario et al. 2014, Paul & Amos 2011) but are generally less effective because they prevail lower in the intertidal or subtidal zone and hence are more deeply submerged during storm surges.

The effectiveness of marshes and mangroves as coastal storm buffers conceptually depends on (*a*) their functionality, that is, their capacity to reduce storm surge levels, storm waves, and associated flood and erosion risks, and (*b*) their persistence, that is, their capacity to resist and recover from storm damage to the ecosystem and from other potential stress and disturbance factors, such as their ability to persist under long-term sea level rise through sediment accretion (e.g., Bouma et al. 2014, Gijsman et al. 2021). While an increasing number of studies have emphasized the value of marshes and mangroves for storm protection, our review demonstrates that there are also limitations to how much attenuation of storm impacts marshes and mangroves can really provide under extreme conditions. Knowledge remains incomplete, partly because direct empirical observations of the functionality and persistence of wetlands during and after extreme events remain relatively scarce and show case-dependent results. Modeling studies, attempting to evaluate storm surge and wave attenuation functionality across different wetland settings for rare extreme conditions, have intrinsic uncertainties and limited validation with observations and show various responses. So far, these limitations have hampered rigorous science-based implementation of nature-based coastal defense programs. In an attempt to advance this rapidly expanding research field, we present a critical review of the scientific evidence so far, summarizing promising advancements and pinpointing key knowledge gaps.

Below, we first discuss studies on the functionality of wetlands as storm buffers, starting with reports claiming reduced damage from storms in settlements sheltered behind marshes or mangroves. Next, we examine biophysical studies of the attenuation of storm waves, storm surge levels, and erosion; summarize knowledge on the persistence of marshes and mangroves during and after storm impacts; and critically reflect on nature-based coastal defense approaches. Finally, we summarize critical questions and directions for future research.

2. INDIRECT INDICATIONS OF STORM PROTECTION

It was after the extremely devastating Indian Ocean tsunami in 2004 that the first reports claimed reduced damage and a lower death toll in coastal settlements located behind mangroves (e.g., Danielsen et al. 2005, Kathiresan & Rajendran 2005). However, these conclusions were challenged because other factors—topographic elevation, distance from the coast, strength of built infrastructure, and so on—were not accounted for (Dahdouh-Guebas & Koedam 2006, Feagin et al. 2010, Kerr et al. 2006).

More analyses were performed for coastal flood events due to storms, especially after the US Gulf Coast was hit by the particularly deadly and costly Hurricanes Katrina and Rita in 2005 (Day et al. 2007). For marshes, for instance, Costanza et al. (2008) analyzed 34 major US hurricanes, indicating that 60% of the variation in property damage was explained by wind speed and marsh area in the storm path and that 1 hectare of marsh loss corresponded to a median US$5,000 increase in storm damage. A similar analysis for 127 storms in China gave qualitatively comparable results (Liu et al. 2019). For mangroves, for example, Das & Vincent (2009) reported that, following a 1999 cyclone in Orissa, India, villages sheltered by wider mangroves experienced significantly fewer deaths than villages with narrower or no mangroves, but Baird et al. (2009) commented that this analysis did not account for other variables that can affect storm surge inundation. Large-scale assessments of cyclone impacts on loss of coastal economic activity (through remote sensing of nighttime luminosity) indicated lower losses with increasing mangrove width (Hochard et al. 2019) and full mitigation behind mangroves wider than 1 km (del Valle et al. 2020). However, further analysis by Hochard et al. (2021) demonstrated that protection from direct cyclone exposure (<100 km from the cyclone eye) occurred only in settlements with high elevation (>50 m above mean sea level) combined with mangroves (>10 m wide).

Other indirect evidence comes from regional-, continental-, and global-scale assessments, based on modeling approaches assessing factors such as surface areas, population numbers, and economic assets sheltered by tidal wetlands (e.g., Arkema et al. 2013, Menendez et al. 2020, Van Coppenolle et al. 2018, van Zelst et al. 2021). Although model assessments on such large scales are inevitably approximate, they indicate that storm protection benefits from wetlands might be widespread. To complement such indirect assessments, we focus below on biophysical studies directly assessing the capacity of wetlands to reduce waves, storm surges, and erosion.

3. BIOPHYSICAL MECHANISMS DETERMINING THE FUNCTIONALITY OF TIDAL WETLANDS AS STORM BUFFERS

First, we highlight that short-period wind waves (**Figure 1**) and long-period storm surges interact with wetlands at fundamentally different scales of length and time (**Figure 2**). Consequently, wave or surge attenuation rates—commonly expressed as the vertical reduction in wave or surge height as a function of distance traveled through the wetland—are generally approximately two orders of magnitude smaller for storm surge attenuation (~1% per 100 m) than for wave attenuation (~1% per 1 m) (e.g., Gedan et al. 2011) (**Figure 2**).

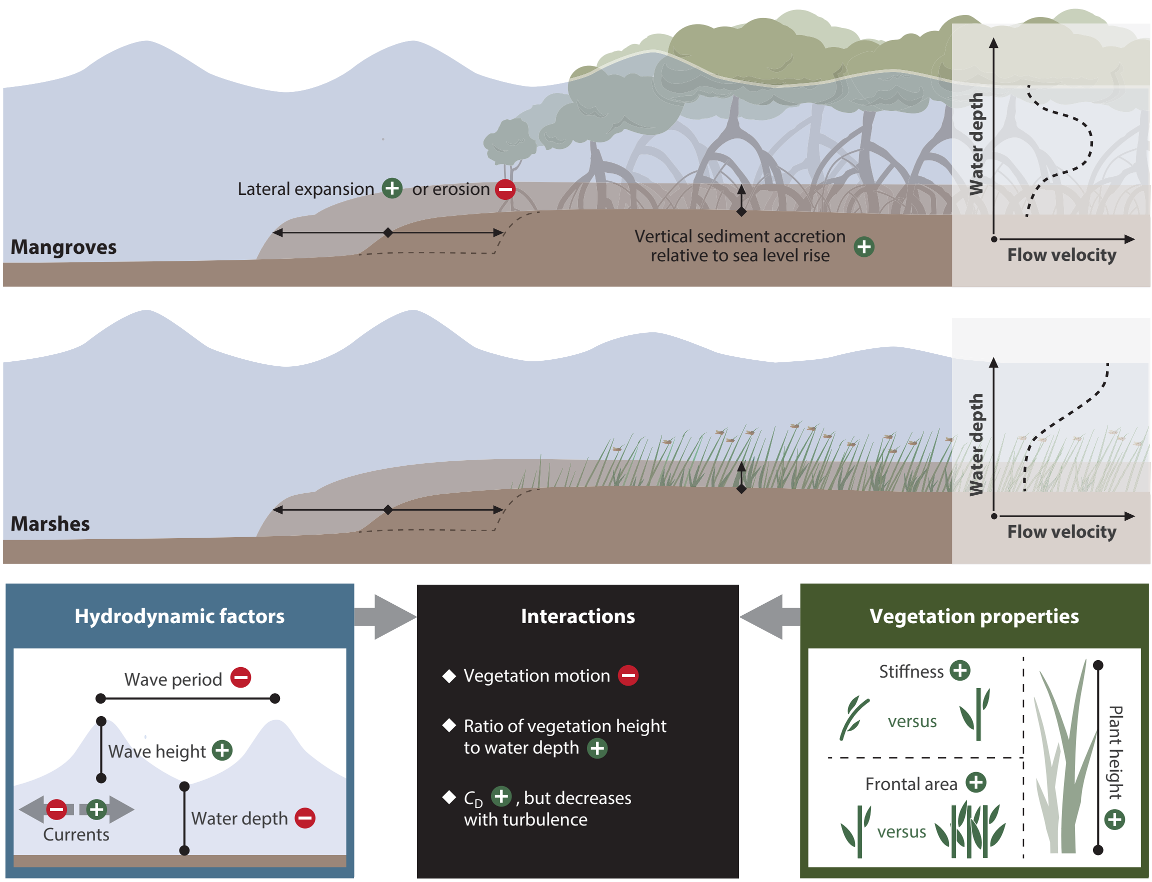

Figure 1

Summary of the main factors affecting rates of wind wave attenuation in mangroves and marshes. Factors that usually have a positive or negative effect on the wave attenuation rate are denoted with a plus or minus sign, respectively, but note that the complexity is usually larger than suggested in this simple scheme. C_D is the vegetation drag coefficient.

3.1. Wave Attenuation

As wind waves propagate through an inundated vegetated wetland, wave height and energy are reduced through drag induced by the vegetation, bottom friction, and potential wave breaking. Below, we focus on wave attenuation specifically by vegetation-induced drag. Pioneering theoretical work by Dalrymple et al. (1984) considered that, for regular linear waves propagating through a uniform canopy of emergent cylindrical stems, the vegetation-induced wave energy dissipation ε can be formulated as

$$\varepsilon = \frac{\rho g^3}{12\pi c^3}\left(\frac{a}{k}\right) C_D \frac{\sinh^3(k\alpha h) + 3\sinh(k\alpha h)}{3\cosh^3(kh)} H^3, \qquad 1.$$

where ρ is the water density, g is the gravitational acceleration, h is the water depth, H is the root-mean-squared wave height, k is the wave number, and c is the wave phase speed. The vegetation-dependent variables are the plant frontal area density, a (i.e., the plant area in the vertical plane perpendicular to the flow, per unit volume); the vegetation submergence ratio, α (i.e., the ratio of vegetation height to water depth); and the vegetation drag coefficient, C_D (representing

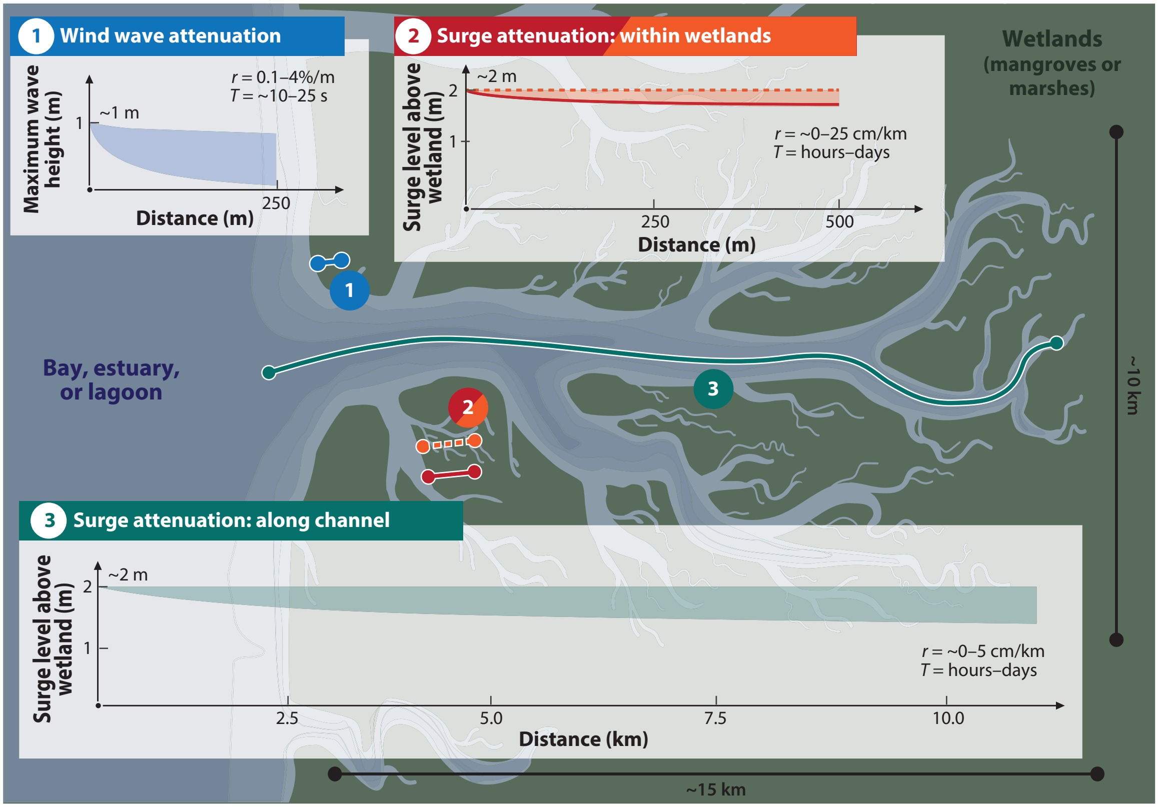

Figure 2

Schematic overview of strong differences in length scales and timescales of wind wave attenuation (transect ①) and storm surge attenuation (transects ② and ③), illustrating for the latter that the presence and dimensions of channels strongly affect the variability in storm surge attenuation rates. In the transect panels, r is the wave attenuation rate, and T is the wave period.

drag from pressure difference and skin friction, but also processes not directly accounted for) (**Figure 1**). Adapted formulations (and agreement with laboratory flume data) were subsequently presented by Mendez & Losada (2004), for cases with irregular waves and plane bathymetry, and by Kobayashi et al. (1993), who used a linear drag approximation, revealing that wave height decreases exponentially with propagation distance.

These theoretical expressions are supported by a plethora of laboratory flume experiments showing wave attenuation for both rigid (e.g., Augustin et al. 2009, Hu et al. 2014, Phan et al. 2019) and flexible (e.g., Anderson & Smith 2014, Luhar & Nepf 2016) vegetation or vegetation mimics. Additionally, many field studies have shown wave attenuation along transects through salt marshes (e.g., Foster-Martinez et al. 2018, Garzon et al. 2019a, Schoutens et al. 2019, Vuik et al. 2016) and mangroves (e.g., Horstman et al. 2014, Mazda et al. 2006, Quartel et al. 2007). Reported wave attenuation rates vary from approximately 0.1% to 4% per meter (Gedan et al. 2011, Narayan et al. 2016). As summarized in **Figure 1**, variations in wave attenuation rates depend on vegetation properties (with rates generally increasing with vegetation frontal area density and stiffness), hydrodynamic conditions (with rates often increasing with wave height and decreasing with wave period, water depth, and opposing currents), and interactions between vegetation and hydrodynamics (e.g., rates can decrease by vegetation motion, by reduced ratio of vegetation height to water depth, and by reduced vegetation drag coefficient through increased turbulence). These dependencies are, however, more complex than suggested in the simple scheme in **Figure 1**, and apparently contradictory trends are sometimes reported. For instance, in mangroves, Mazda et al. (2006) and Quartel et al. (2007) found wave attenuation rates increasing with water depth as denser sections of the canopy became submerged, while for fully submerged marsh vegetation, lower attenuation rates are found with increasing water depth due to decreasing drag from the submerged vegetation and bottom (e.g., Garzon et al. 2019a, Schoutens et al. 2019). Anderson & Smith (2014) found that higher wind wave frequencies (above the spectral wave peak) were preferentially dissipated, in contrast to Jadhav et al. (2013), who recorded maximum dissipation at frequencies close to the spectral peak, while observations from Riffe et al. (2011) demonstrated that both high and low frequencies were more strongly dissipated by vegetation. Much less is known about the effect of vegetation on infragravity waves (with typical periods between 25 and 250 s) (van Rooijen et al. 2016), although recent observations demonstrated that infragravity waves in mangroves are substantially less attenuated than shorter-period waves (Norris et al. 2021), consistent with the theory of Henderson et al. (2017). Ultimately, temporal variations in hydrodynamic forcing and in vegetation properties (e.g., due to seasonal cycles) can result in substantial temporal changes in wave attenuation rates (Foster-Martinez et al. 2018, Schoutens et al. 2019).

Numerical models, which are used to assess wave attenuation in marshes and mangroves (e.g., Maza et al. 2021, Smith et al. 2016, van Rooijen et al. 2016, Yin et al. 2021), usually compute wave propagation based on a wave energy balance approach, as noted above (e.g., Equation 1) (Dalrymple et al. 1984, Mendez & Losada 2004). However, predicting wave attenuation in vegetated regions involves several challenges. Direct measurement of frontal area density (a in Equation 1) involves substantial logistical difficulties, especially over large areas (Tinoco et al. 2020), although some recent progress has been made by using forest age as a proxy (Maza et al. 2021) and remote sensing approaches (Oteman et al. 2019). Another major challenge is the determination of the vegetation drag coefficient (C_D in Equation 1). This parameter is typically found through calibration to field or laboratory measurements, whereby a bulk C_D is specified by the fit that best matches the observed reduction in wave height. For a given canopy, C_D is typically found to decrease with increasing turbulence of the flow, which has led to parameterizations of C_D values as a function of either the Reynolds number, Re (Maza et al. 2019), or the Keulegan–Carpenter number, KC (Mendez & Losada 2004), with additional modifications for parameters such as wave steepness (Sanchez-Gonzalez et al. 2011) and vegetation submergence ratio (Garzon et al. 2019a). However, neither of these approaches provide a satisfactory universal solution as relationships vary across different environmental conditions and vegetation species (e.g., Mullarney & Henderson 2018). Moreover, Augustin et al. (2009) have suggested that the vegetation submergence ratio controls whether a Re or KC formulation for C_D should be used, while Hu et al. (2014) argued that a direct measurement approach with force sensors is preferred. Similarly, a direct approach was used by Chen et al. (2018) to obtain a relationship between Re and KC.

Further uncertainties in the prediction of C_D and wave attenuation capacity occur for flexible vegetation, when wave motion induces swaying, which is particularly relevant for marsh species. The empirical calibration approach yields C_D values that encompass vegetation motion, but these estimates can diverge substantially from values expected when stem movement is explicitly accounted for (Riffe et al. 2011). Some authors have suggested that predictions can be improved by using the relative velocity between vegetation and water movement in the calculation of Re (van Veelen et al. 2021). A direct approach solving for the vegetation motion found that dissipation depended on a dimensionless stiffness that combined wave and vegetation parameters (Mullarney & Henderson 2010). The theory yielded dissipation estimates that were approximately 30% less than those for equivalent rigid stems and were supported by field measurements (Riffe et al. 2011).

Luhar & Nepf (2016) extended the theoretical approach to simulate wave-induced vegetation motion over a wider range of plant characteristics and larger movement. Recent numerical modeling approaches in which a model for vegetation movement is coupled to a wave model demonstrated reasonable agreement with laboratory results and may offer a promising way forward (Yin et al. 2021).

A major challenge to assess the wave attenuation capacity under extreme storm conditions is that the majority of C_D relationships discussed above have been established based on laboratory experiments and observations under mild wave conditions, with wave heights of a few tens of centimeters and shallow water depths (<1 m). These results are not directly transferable to storm conditions, when wave heights, water depths, and turbulence levels can be substantially larger. Indeed, Pinsky et al. (2013) highlighted that extending *Re*-based parameterizations for C_D to storm conditions could lead to significant overestimation of the wave attenuation (up to 1,600%). Similarly, Henderson et al. (2017) noted that using frictionless linear theory under storm conditions for high vegetation densities (e.g., Dalrymple et al. 1984) results in a threefold overestimation of dissipation. Garzon et al. (2019b) evaluated five drag coefficient formulations under storm conditions and found differences of up to 40% in wave height reduction at 100 m inside a marsh. Numerical modeling of waves in salt marshes under storm conditions by Marsooli et al. (2017) revealed an order-of-magnitude variability in drag coefficient owing to spatial changes in *Re*, and therefore they cautioned against use of a constant coefficient. An additional key source of uncertainty is the extent to which vegetation stems break under storm forcing, as relevant data exist for very few species (Vuik et al. 2018).

A few studies nonetheless exist, showing that vegetation retains the ability to attenuate waves under storm conditions. Flume experiments by Möller et al. (2014) found that submerged salt marsh vegetation was effective at reducing the height of waves under storm surge conditions ($H =$ 0.9 m in above-marsh water depths of 2 m), with 60% of the attenuation attributed to the presence of vegetation. Additionally, for emergent trees, van Wesenbeeck et al. (2022) found a reduction of wave heights (5–25% over 40 m) for large waves ($H_{sig} = 1.5$ m, $H_{max} = 2.5$ m). Field observations during storm conditions [$H = 0.4$ m in 0.8-m water depth (Jadhav et al. 2013), and $H = 0.7$ m in 2.5-m water depths (Vuik et al. 2016)] showed that wave energy is quickly attenuated at the leading edge of a salt marsh (∼50% over widths of 30–50 m). Garzon et al. (2019a) used an empirical modification of *Re* and *KC* formulations for drag coefficients to account for the submergence parameter validated under storm conditions and predicted that the salt marsh would retain the ability to reduce wave heights (by ∼50%) even for a once-in-10,000-years event.

3.2. Storm Surge Attenuation

Storm surges are generated primarily through transfer of momentum from wind and waves to the water, low atmospheric pressure, and setup of water levels through interaction with the coastal bathymetry and shoreline geometry (Resio & Westerink 2008). The resulting increased sea level, which can be several meters higher for a few hours to more than a day, is superimposed on the astronomical tides, together producing a fluctuation in sea level that propagates as a long-period wave through estuaries, bays, lagoons, and their associated wetlands (**Figure 2**). As tides and storm surges propagate through wetlands, the water motion experiences drag from the vegetation and sediment bed, which limits the water exchange and thereby can reduce the peak water level reached within and behind the wetland.

Early field observations of reduced storm surge levels were reported from marshes many kilometers wide along the US Gulf Coast, with attenuation rates ranging from 4 to 25 cm/km for individual hurricanes and different marsh transects (Lovelace 1994, McGee et al. 2006, USACE

1963). More recent observations, including tidal inundation events and storm surges, confirmed large variations in attenuation rates and in some cases amplification (i.e., increase of peak water level) of tides and storm surges over wetlands (for a review, see, e.g., Glass et al. 2018). Although observations remain relatively scarce and site specific, we identify some general mechanisms for the variability in attenuation rates.

First, temporal variability between flood events seems to depend on wetland inundation depth, with a tendency toward lower attenuation rates with increasing inundation depth of marshes (Glass et al. 2018, Stark et al. 2015) and mangroves (Horstman et al. 2021), owing to lower drag from the bottom and vegetation. Limited attenuation or even amplification of deep marsh inundation events has also been attributed to water setup against landward barriers, such as levees or topographic barriers that limit further inland flood propagation (Glass et al. 2018, Kiesel et al. 2019, Stark et al. 2016).

Second, spatial variations in attenuation rates may be partly related to the presence and dimensions of tidal channels that typically dissect marshes and mangroves (**Figure 2**). Field studies of tidal flow velocity patterns through marshes (e.g., Vandenbruwaene et al. 2015) and mangroves (e.g., Horstman et al. 2021) show more rapid flood propagation through unvegetated channels relative to the adjacent vegetated wetland platforms. Accordingly, Stark et al. (2015) measured maximum tidal attenuation rates of only 5 cm/km along a wider marsh channel (170–30 m wide over a 4-km length) but larger attenuation rates over marsh transects with narrower channels, reaching up to 70 cm/km over short transects (∼50 m) of nonchannelized continuous marsh vegetation. For mangroves in New Zealand, Montgomery et al. (2018) measured an attenuation rate of 24 cm/km along an approximately 1-km transect of continuous nonchannelized mangroves, while negligible attenuation was observed along a channelized mangrove transect approximately 1 km long. Accordingly, attenuation rate as recorded by a network of water level gauges deployed during two hurricane surge events (Hurricanes Charley and Wilma) in southwest Florida, USA, was 4.2 cm/km along a river channel but 9.5 cm/km through an intact mangrove–marsh complex (Krauss et al. 2009). In another New Zealand mangrove study, Horstman et al. (2021) measured tidal attenuation rates of up to 12 cm/km along an approximately 500-m-long mangrove creek transect and up to 36 cm/km within the adjacent mangrove forest. Hence, river and tidal channels, which are intrinsic landscape features of marshes and mangroves, strongly diminish the surge attenuation capacity of tidal wetlands (**Figure 2**).

As field observations of extreme storm surges in wetlands remain scarce, hydrodynamic models are used as additional research tools to gain an understanding of storm surge attenuation by wetlands (e.g., Chen et al. 2021, Marsooli et al. 2016, Stark et al. 2016). Models of tidal and surge propagation are typically based on the shallow-water equations, where the additional drag from wetland vegetation is implemented in the momentum equations by either increasing the bed roughness or including an additional momentum loss term. The latter can be assessed as the drag force F_d on rigid cylindrical plant structures:

$$F_\mathrm{d} = \frac{1}{2}\,\rho\,a\,C_\mathrm{D}\,|\mathbf{u}|^2, \qquad 2.$$

where $\mathbf{u}$ is flow velocity (for other variables, see Equation 1). The major challenges in parameterizing the frontal plant area density a and drag coefficient C_D, noted in Section 3.1 for wave modeling, also apply here. For instance, Horstman et al. (2021) demonstrated spatial and temporal variations in the bulk drag coefficient during tidal propagation in a mangrove–creek system. Hence, model results, which commonly assume a constant drag coefficient, need to be interpreted with caution. For instance, a comparison of storm surge simulations and data for mangroves in Florida showed considerable model sensitivity to the implemented drag force formulation (Chen et al. 2021).

Comparing modeling studies, we can generally distinguish two main mechanisms of storm surge attenuation by wetlands: (*a*) Surge attenuation within nonchannelized vegetated wetlands is basically driven by a balance between pressure differences and friction from the wetland vegetation and sediment surface (e.g., Montgomery et al. 2019), and (*b*) storm surges propagating through a channel (inside wetlands or larger estuarine or deltaic channels) can be additionally attenuated due to lateral flooding and water storage on wetlands fringing along that channel (e.g., Smolders et al. 2015). The friction effect (the first mechanism) is called here the within-wetland attenuation, and the water storage effect (the second mechanism) is called the along-channel attenuation (**Figure 2**). As channels are common features of marshes and mangroves, surge propagation through tidal wetlands is usually affected by both frictional and lateral water storage effects (Stark et al. 2016).

Models have demonstrated that storm surge attenuation rates depend on specific properties of the storm forcing, the wetland ecosystem, and the larger-scale coastal landscape. In terms of storm properties, several model studies have indicated that attenuation rates generally decrease under extreme storm surge levels, as contributions by within-wetland frictional effects decrease with wetland inundation depth (Deb & Ferreira 2017, Lawler et al. 2016, Liu et al. 2013, Sheng et al. 2021, Wamsley et al. 2010). Similarly, for wetlands located higher in the intertidal zone, simulated storm surge attenuation rates are larger (Loder et al. 2009, Smolders et al. 2015, Stark et al. 2016), implying that wetlands with a sediment accretion deficit relative to sea level rise also lose their effectiveness for storm surge attenuation (e.g., Temmerman et al. 2012). Furthermore, attenuation rates diminish for simulated storms with a longer duration (Resio & Westerink 2008, Wamsley et al. 2010) and slower forward motion (Deb & Ferreira 2017, Hu et al. 2015, Liu et al. 2013, Sheng et al. 2012, Zhang et al. 2012), as both imply more time for landward surge propagation. In terms of wetland ecosystem properties, more effective surge attenuation is simulated for wider wetlands, with a higher sediment surface elevation and larger vegetation-induced friction (Hu et al. 2015, Loder et al. 2009, Sheng et al. 2012). In line with observations, the dimensions of channels within wetlands play a major role: Simulations with deeper or wider channels show lower storm surge attenuation rates (Stark et al. 2016, Temmerman et al. 2012).

Model studies further indicate that rates of storm surge attenuation by wetlands depend predominantly on larger-scale landscape settings (**Figure 3**). For example, storm surge attenuation rates of 2–16 cm/km were simulated for the marshes in the Mississippi deltaic area, which are several tens of kilometers wide (Resio & Westerink 2008, Wamsley et al. 2010), while

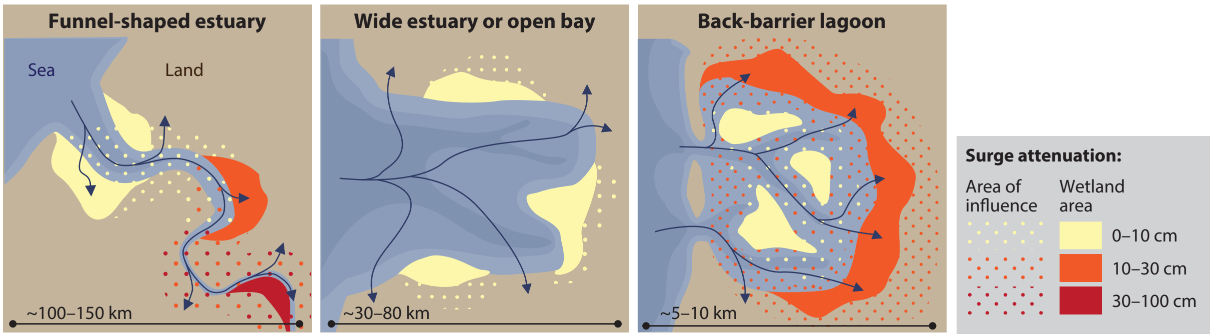

Figure 3

Summary of how the magnitude of storm surge attenuation (*color scale*) depends on coastal and estuarine landscape setting. Arrows indicate storm surge propagation trajectories, solid colored areas indicate wetland areas, and dotted colored areas indicate zones of influence (i.e., areas with reduced surge height due to the presence of wetlands denoted with the same color).

attenuation rates of 6–30 cm/km were simulated for the approximately 20-km-wide mangrove systems in South Florida (Chen et al. 2021, Liu et al. 2013, Zhang et al. 2012). Both cases are essentially based on within-wetland attenuation, where surges are forced to propagate through wetlands that are tens of kilometers wide. For the case of along-channel attenuation by fringing marshes along funnel-shaped estuaries, simulations show that marshes that are the same size but located farther upstream along narrower reaches of the estuary are more effective in attenuating storm surges (Fairchild et al. 2021, Smolders et al. 2015) (**Figure 3**). By contrast, simulations showed more moderate to limited contributions of marshes to storm surge attenuation for wide bays that are several times wider than the width of fringing marshes, such as the Chesapeake Bay (Haddad et al. 2016) and the Albemarle–Pamlico estuarine system (Cassalho et al. 2021) (**Figure 3**). Also, relatively limited effects of the presence of wetlands within back-barrier lagoon systems were simulated for Jamaica Bay, New York (Marsooli et al. 2016), and the Delmarva coast (Lawler et al. 2016). For Jamaica Bay, Orton et al. (2020) simulated the year 1870 versus the present, indicating that combined wetland reclamation around the bay, channel deepening within the bay, and tidal inlet widening have resulted in increased storm surge levels within the bay. Numerical experiments of separate effects revealed that increasing the wetland area surrounding the bay to its historical extent reduced surge levels by 13% (∼30 cm), while increasing wetland area in the bay center reduced surge levels by only 2% (**Figure 3**).

In summary, empirical data and modeling studies have demonstrated considerable storm surge height reduction by large (at least 10 km wide), continuous marshes and mangroves with few or small channels, while smaller or discontinuous marshes and mangroves do not provide significant storm surge height reduction. Furthermore, wetlands are most effective for storm surge attenuation when they are located farther inland along narrow reaches of funnel-shaped estuarine channels. Under these circumstances, wetlands are especially effective during moderate storm surges but less so for extreme storms that persist for more than a day.

3.3. Erosion Risk Mitigation

Mitigation of waves and currents by aboveground vegetation structures reduces bed shear stresses in marshes and mangroves compared with unvegetated mudflats (e.g., Norris et al. 2021, Vandenbruwaene et al. 2015). Observed bed shear stresses by the combined action of currents and waves were five times lower in a salt marsh (0.06 N/m^2) than on an adjacent mudflat (0.27 N/m^2) in the Yangtze Delta (Shi et al. 2012). Additionally, the shear strength of (often fine-grained) tidal wetland substrates increases substantially with belowground biomass (Brooks et al. 2021, Cahoon et al. 2021), resulting in shear strengths that are three to four times greater in densely vegetated interior marshes than at marsh edges (Gillen et al. 2021). Critical bed shear stresses for erosion increased from 0.26–0.58 N/m^2 for a bare intertidal mudflat (Andersen et al. 2007) to 1.9–4.3 N/m^2 for (reclaimed) salt marshes (Watts et al. 2003). Erosion thresholds of intertidal mudflats and wetlands also increase with the presence of biofilms and with reduced water contents following longer exposure to (warmer) air (Nguyen et al. 2020). Willemsen et al. (2022) observed a maximum event-based vertical bed level variability of 12 mm in salt marshes across the Netherlands, while this variability was consistently greater at the lower-elevation bare tidal flats in front. Pennings et al. (2021) observed that increased attenuation of aboveground hydrodynamics combined with greater root biomass makes mangroves more effective at reducing storm erosion than salt marsh plants.

Apart from vertical erosion, tidal wetland vegetation also significantly reduces lateral coastal erosion (retreat). Adjacent experimental plots without and with mangrove vegetation showed 9.7 m and 1.6 m of lateral erosion, respectively, over a period of 19 months that included a hurricane (Pennings et al. 2021). On a larger scale, the presence of mangroves was associated with coastal extension over a 30-year period in southern Thailand, whereas mangrove removal was linked to coastal retreat (Thampanya et al. 2006). Likewise, lateral erosion rates in a salt marsh were 48% greater in plots where above- and belowground biomass was removed, whereas no significant difference was observed if only aboveground biomass was removed (Silliman et al. 2019). Despite these clear relations between reduced coastal erosion and belowground biomass, a mechanistic understanding of the contribution of plant roots to wetland stability is still lacking (Bouma et al. 2014). Importantly, long-term lateral wetland erosion is dominated by frequent average conditions rather than infrequent extremes: A global analysis showed that storms and hurricanes contribute to less than 1% of long-term salt marsh lateral erosion rates (Leonardi et al. 2016).

4. PERSISTENCE OF TIDAL WETLANDS AS COASTAL STORM BUFFERS

To be effective coastal storm buffers, marshes and mangroves need to have a certain resistance and resilience to storm damage. Resistance is defined here as the capacity to avoid damage, while resilience is the capacity to recover from eventual damage (e.g., Castagno et al. 2021). As reviewed below and summarized in **Figure 4**, healthy tidal wetlands often exhibit high resistance and resilience to storm impacts, making them persistent in time. Not only are contemporary observations indicative (e.g., Armitage et al. 2020, Castagno et al. 2021, Spencer et al. 2016), but also the geologic record shows the persistence of tidal wetlands over millennia, even though they would have experienced many severe storms (e.g., Jones et al. 2017, Willard et al. 2003). Yet there are limits, or so-called tipping points, beyond which wetlands can temporarily or permanently degrade into bare mudflats or shallow open water (e.g., Howes et al. 2010), with consequent loss of storm wave and storm surge attenuation functionality (e.g., Temmerman et al. 2012).

4.1. Resistance to Disturbance During Storms

Tidal wetlands can experience various degrees of structural storm damage to vegetation and sediments, due to severe wind forces, hydrodynamic forces from waves and currents, and erosion. First, direct wind damage is especially reported from mangrove forests, while it is generally minimal in marshes. In a review of tropical cyclone impacts on mangroves, Krauss & Osland (2020) found that loss of foliage, breaking of branches and stems, and tip-over of trees occur almost without exception for category 3 or higher wind speeds (>178 km/h). Larger trees and their upper parts are particularly vulnerable, while damage to smaller trees and saplings can be more limited (**Figure 4*a***) because they are positioned lower and hence experience smaller wind drag forces, and shorter trees can also be submerged by the storm surge and therefore protected from wind damage (Armitage et al. 2020, Krauss & Osland 2020, Radabaugh et al. 2020). The latter would imply that their wave and surge attenuation capacity could be largely conserved during storms. Marsh species are much less vulnerable to storm damage, as they are low and more flexible. For example, as a category 5 hurricane (Michael) made landfall in 2018 in Florida, USA, healthy tidal marshes experienced little damage (2%) (Castagno et al. 2021), while a category 5 hurricane (>250 km/h) would have yielded major structural damage to mangroves (Krauss & Osland 2020).

Second, structural vegetation damage can be caused by the storm waves and currents. Most studies examine marsh plants, showing that damage by hydrodynamics can result from two main mechanisms (e.g., Silinski et al. 2016): drag forces exerted by the water flow, which can cause stem bending, folding, and eventually breakage (e.g., Rupprecht et al. 2017, Vuik et al. 2018), and uprooting of plants, through scouring of sediments around stems (e.g., Bouma et al. 2009). The magnitude of both mechanisms depends on morphological plant traits, apart from hydrodynamic

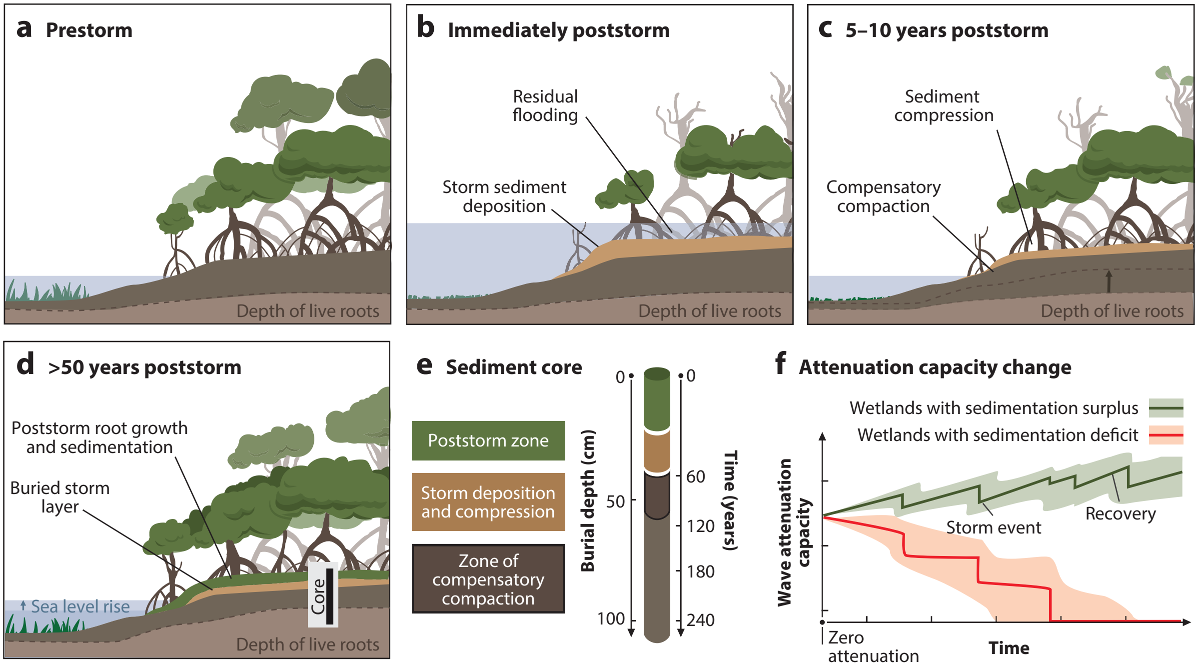

Figure 4

Summary of storm impacts on tidal wetlands, illustrated here for mangrove forests. (*a*) Prestorm wetland structure. (*b*) Immediate poststorm structural damage to vegetation (with typically more damage for tall mangrove trees and less for short marsh plants), sediment deposition, and residual flooding. (*c*) The situation 5–10 years after the storm, with partial recovery of aboveground vegetation and root growth countered by sediment compaction, as dead roots decay, organic sediments oxidize, and mineral sediments consolidate. (*d*) Continued recovery and sedimentation after more than 50 years, when a sediment core is extracted. (*e*) Sediment core depicting the storm deposit (52 years previously); approximate burial depths versus time are from Breithaupt et al. (2018). (*f*) Change in wind wave attenuation capacity in response to wetland storm damage and recovery trajectories for a wetland scenario with a sedimentation surplus relative to sea level rise (*green*) and a scenario with a sedimentation deficit (*red*). For each scenario, the indicated ranges represent potential variability in the wetland trajectory due to additional stresses from, for example, increasing storminess in certain regions, increasing temperature and droughts, and anthropogenic disturbances of sediment budgets and nutrient budgets.

and sediment conditions. Flume experiments with currents (Bouma et al. 2009) and storm waves (Schoutens et al. 2021) showed that seedlings with stiffer and thicker stems suffered from larger scouring volumes around their stems. Salt marsh species with higher frontal areas and stiffness experience increased exposure to drag forces, while more flexible species can bend and thereby reduce their frontal areas and experienced drag forces (Paul et al. 2016, Rupprecht et al. 2017). Interestingly, marsh species traits that increase the plants' capacity to cope with stress from hydrodynamic drag forces (i.e., lower stiffness and frontal plant area) at the same time decrease their capacity to attenuate wind waves and currents, implying a trade-off between hydrodynamic stress avoidance and attenuation capacity (Schoutens et al. 2020). In summary, the resistance of marsh vegetation to storm damage is often high (e.g., Castagno et al. 2021, Spencer et al. 2016).

Third, lateral erosion of wetland edges during storms can be prominent, with examples showing more lateral erosion in marshes (>5 m in Texas during Hurricane Harvey in 2017) than in adjacent mangroves (<0.5 m) (Armitage et al. 2020, Pennings et al. 2021). However, erosion in wetland interiors is often limited; for instance, salt marsh plants effectively limit marsh erosion under storm conditions (Spencer et al. 2016), and measurements within healthy mangroves in Texas in 2017 yielded little erosion from Hurricane Harvey (Pennings et al. 2021). Yet wetland erosion may occur during extreme storms, such as Hurricanes Katrina and Rita in 2005, causing erosion in 5–8% of low-salinity marsh areas in the Mississippi deltaic plain but negligible erosion in salt marshes (Howes et al. 2010).

4.2. Storm Legacies and Resilience After Storms

Storm impacts on tidal wetlands, and in particular on mangrove forests, can persist for years as legacies of past disturbance. Legacies may include, for example, downed woody debris, reduced live vegetation cover, plant deformation, new sediment depositional layers, ponding of water, and altered biogeochemical processes, which may provoke delayed plant mortality (Hanley et al. 2020, Krauss & Osland 2020, Radabaugh et al. 2020) (**Figure 4**). Tidal wetland resilience requires recovery through successional processes, which may start soon after the storm disturbance but may develop slowly. In general, while the altered structure of tidal forested wetlands, such as mangroves, can persist for years after a storm (e.g., Krauss & Osland 2020, Smith et al. 1994), the structure of herbaceous marshes can recover more quickly when erosion has not been prominent. For instance, after the passage of Hurricane Harvey in Texas in 2017, regrowth of leaves on damaged mangrove branches started within two months (Armitage et al. 2020). After Hurricane Irma in 2017, mangrove canopy cover in South Florida increased from 40% cover one to two months after the storm to 60% cover three to six months after the storm, and the cover remained the same nine months after the storm (Radabaugh et al. 2020). While only 2% of the marsh area was damaged after Hurricane Michael in Florida in 2018, recovery amounted to 16% of the damaged marsh area six months after the storm (Castagno et al. 2021). Overall, storm legacies are more a characteristic of tidal forested wetlands (which have less environmental plasticity and are reliant on regeneration after storms) than marshes (which have greater environmental plasticity and are reliant on sediment–root resistance during storms) (Friess et al. 2012).

One particular storm legacy, often prevailing in both mangroves and marshes, is the deposition of an anomalously thick sediment layer. Sediments, originating from adjacent subtidal or intertidal locations, are typically brought into motion by large storm-generated waves and subsequently transported by the storm surge deep into wetlands, where the sediments are deposited (**Figure 4*b***). While too much sedimentation can cause plant mortality (e.g., Cahoon et al. 2003, Walters & Kirwan 2016), storm deposits have been widely considered important in contributing to wetland sediment accretion and sustaining plant productivity in response to sea level rise (e.g., Baustian & Mendelssohn 2015, Castaneda-Moya et al. 2020, Walters & Kirwan 2016). Indeed, accretion of mineral and organic sediments is a key mechanism through which tidal wetlands can build up elevation and compensate for the stress on wetland plants induced by sea level rise, while in the case of insufficient sediment accretion, sea level rise can lead in the long term to wetland submergence and die-off (e.g., Breithaupt et al. 2018, Coleman et al. 2022, Lovelock et al. 2015). For instance, Williams & Flanagan (2009) demonstrated that storm sedimentation contributed up to two-thirds of the sedimentation required to maintain long-term buildup of wetland elevation with sea level rise in Louisiana. Likewise, the thickness of sediment deposits from Hurricane Wilma in 2005 along five small Everglades rivers (Florida, USA) averaged 6 cm at the mouths of these rivers and decreased by 3 mm per kilometer of distance from open water, up to a distance of 16 km inland, beyond which sedimentation was no longer evident (Smith et al. 2009). However, Cahoon (2006) cautioned that an immediate rise in wetland elevation through storm sedimentation may be countered by longer-term processes after the storm, such as sediment erosion, compaction of the storm deposit through self-weight consolidation, compaction of underlying sediments, root decomposition, and associated shallow soil subsidence (**Figure 4*c*,*d***). These adjustments can be fully compensatory in organic-rich wetland sediments: in some locations, elevation changes return to prestorm trajectories 5–10 years after storm deposits, with only a temporary inflection of the elevation trajectory (Feher et al. 2020). Such compensatory sediment elevation change would not be easily discernible within the geologic record. For instance, in a sediment core taken 52 years after a storm, a 7-cm storm sediment layer at a core depth of 25 cm would not have provided 7 cm of elevation increase to that wetland 52 years ago (**Figure 4*e***, based on data from Breithaupt et al. 2018).

Biogeochemical impacts are another type of storm legacy that may have negative or positive effects on wetland resilience. On the one hand, ponding of water due to residual flooding may induce increased salinity and bacterial production of phytotoxic sulfides in the water-logged, oxygen-poor sediments, and thereby cause delayed plant mortality (e.g., Himmelstein et al. 2021, Swarzenski et al. 2008). On the other hand, nutrient-limited tidal wetlands can benefit from nitrogen and phosphorus supplied by the storm surge deposits, while sedimentation can also yield more oxidized soils and reduce phytotoxic sulfides, and thus facilitate marsh recovery (Baustian & Mendelssohn 2015, Castaneda-Moya et al. 2020). The degree of benefit relates to the antecedent biogeochemical state of the tidal wetland soil. For example, mangroves and marshes in the Everglades benefit from storm-redistributed phosphorus subsidies to offset phosphorus limitation (Smith et al. 2009). However, deposition of nitrogen and phosphorus to tidal wetland soil surfaces would need to benefit macrophytes through uptake, the capacity of which would relate inversely to structural damage. Accordingly, Castaneda-Moya et al. (2020) showed that total phosphorus was higher three months after Hurricane Wilma in Everglades mangroves, which was related to a simultaneous increase in soil pore water inorganic phosphorus and provided evidence that phosphorus was taken up by residual mangrove trees after the storm.

Finally, we highlight that wetland vulnerability to storm impacts interacts with other stress and disturbance factors, which may originate from climate change and human disturbances. For instance, wetlands with a long-term sediment accretion deficit relative to mean sea level rise may experience gradually increasing tidal inundations, leading to more stress on the wetland vegetation and gradual wetland degradation (e.g., Lovelock et al. 2015). In such a case, wetland resistance and resilience to recurring storm impacts may gradually decrease until a storm ultimately causes so much damage that recovery is inhibited (**Figure 4*f***). Additional stresses from, for example, increasing storminess in certain regions, increasing temperature and droughts, and anthropogenic disturbances of sediment and nutrient supply to tidal wetlands may further enhance wetland vulnerability to individual storm impacts (Friess et al. 2019, Krauss & Osland 2020). For example, Taillie et al. (2020) indicated that during the 2017 hurricane season, mangrove wetlands in the Caribbean region appeared to be more susceptible to wind events with lower wind speeds than previously documented, likely due to greater realized environmental stress under multiple scenarios of land use change.

5. NATURE-BASED COASTAL DEFENSE

Several reviews have called for the conservation and restoration of coastal wetlands as a nature-based approach to mitigate storm impacts (e.g., Morris et al. 2018, Narayan et al. 2016, Sutton-Grier et al. 2015, Temmerman et al. 2013). Whereas engineered coastal defense structures (e.g., seawalls, dikes, and levees) remain important for reducing flood risks in inhabited lowlands, nature-based flood risk mitigation by marshes and mangroves is regarded as a cost-effective complementary strategy (e.g., Narayan et al. 2016, van Zelst et al. 2021), with a certain capacity to be self-sustaining with ongoing sea level rise through the natural process of wetland sediment accretion (e.g., Coleman et al. 2022, Lovelock et al. 2015). Nature-based approaches also provide additional societal benefits, such as carbon sequestration, water quality regulation, fishery production, recreation, and biodiversity (e.g., Barbier et al. 2011).

Coastal wetlands can be integrated with engineered structures, creating vegetated foreshores that reduce wave impacts on and required crest heights of these structures (Bouma et al. 2014, Schoonees et al. 2019, Vuik et al. 2016). Additionally, analysis of historic dike breaches in the Netherlands has shown that narrow marshes in front of dikes, although not effective in preventing dike breaching, can serve as an extra natural dike that is more resistant to erosion than the engineered dike is to dike breaching; as a result, marshes in front of a breached dike reduce the flood discharge through the dike breach and hence limit the growth of the breach dimensions and reduce the speed of flooding, flood depth, and potential damage in the low-lying inhabited land behind the dike (Zhu et al. 2020). Apart from the Netherlands, this contribution of wetlands to flood risk mitigation is particularly relevant to coasts where sediment accretion has raised the wetland elevation up to one or several meters higher than the embanked hinterland, such as in the Firth of Thames in New Zealand (Horstman et al. 2018) or the Ganges–Brahmaputra Delta in Bangladesh (Auerbach et al. 2015).

The flood protection functionality of coastal wetlands depends significantly on the wave conditions, with storm surges requiring much greater wetland widths than ordinary wind waves for substantial risk reduction (**Figure 2**). Further challenges relate to our limited understanding of temporal variations and trends in coastal wetland persistence (width, elevation, and vegetation density) in response to human interference (e.g., Friess et al. 2019), storm damage (e.g., Hanley et al. 2020, Krauss & Osland 2020), and sea level rise (e.g., Coleman et al. 2022, Lovelock et al. 2015) (**Figure 4**). These fundamental differences in the functionality and persistence of coastal wetlands compared with those of traditional engineered structures require a shift toward adaptive coastal management strategies that combine flood risk mitigation with accommodating resilient marshes and mangroves (Gijsman et al. 2021).

First, adaptive coastal management should maintain or create suitable accommodation space for coastal wetlands in upper intertidal areas, where tidal inundation and wave activity are not impeding seedling establishment (Bouma et al. 2014, Hu et al. 2021). Intertidal accommodation space can dynamically change, either expanding with sustained sediment supply (e.g., following land use changes) (Ladd et al. 2019, Roner et al. 2021) or shrinking (e.g., due to eroding impacts of storms and waves) (Leonardi et al. 2018, Thampanya et al. 2006). Wetland erosion may be moderated naturally by subtidal coral reefs and seagrass meadows (Carr et al. 2018, Guannel et al. 2016). Alternatively, artificial permeable structures or double-dike systems (e.g., managed dike realignment) can be put in place to ameliorate hydrodynamic energy and increase sedimentation to create or gain wetland accommodation space (Willemsen et al. 2022, Zhu et al. 2020).

Second, adaptive coastal management should support the sediment trapping and vegetation growth that provide coastal wetlands with their natural resilience to environmental change. Vertical accretion in wetlands can be limited compared with predicted rates of sea level rise (e.g., Lovelock et al. 2015), and dwindling sediment supplies further reduce wetland accretion rates (e.g., Coleman et al. 2022, Friess et al. 2019), which may induce a gradual decline and retreat of coastal wetlands (e.g., Ladd et al. 2019). This process may be exacerbated by engineered structures placed in front of coastal wetlands to reduce wave impacts, which may reduce wetland resilience by limiting sediment supplies (van Zelst et al. 2021).

Effective adaptive management and the use of nature-based solutions require a balancing act between providing the right conditions for wetland persistence and mitigating flood risk (Gijsman et al. 2021, van Zelst et al. 2021). Increasing anthropogenic impacts on coastal systems are severely compromising the natural resilience of marshes and mangroves, which is required to keep up with sea level rise and recover from storm damage (**Figure 4**). Negative impacts of engineered structures on wetland resilience emphasize the need for accommodation space in an era of ongoing coastal development (Friess et al. 2019), while unsuitable hydrodynamic conditions are also a common cause of failed planting efforts to aid wetland persistence (Primavera & Esteban 2008). Hence, successful implementation of nature-based coastal defense requires continued wetland monitoring, modeling of possible wetland development and response pathways, and adaptive management to accommodate wetland resilience (Gijsman et al. 2021).

6. SYNTHESIS AND DIRECTIONS FOR FUTURE RESEARCH

A growing body of evidence demonstrates under which conditions and to what extent tidal wetlands can or cannot contribute to mitigation of coastal storm impacts. Yet developing a universal understanding of the ability of wetlands to protect coastlines remains a substantial challenge. Below, we attempt to summarize the main insights, provide critical reflections, and propose potential directions for future research, on three levels.

6.1. How Effective Are Tidal Wetlands in Protecting Against Different Components of Coastal Storm Impacts?

Presently, the evidence is furthest developed for short-period wind wave attenuation by marshes and mangroves. Even under storm wave conditions, wetlands are effective wave energy dissipators over distances on the order of one to several hundred meters (**Figure 1**). Also, erosion during storms and average conditions is limited by wetlands, with potentially some erosion impact in the first few meters near the seaward wetland edge, while storms usually induce increased sediment accretion rather than erosion in wetland interiors. However, when it comes to reduction of storm surge levels, wetlands are much less effective or not effective at all, depending on properties of the storm, wetland ecosystem, and larger-scale coastal landscape geometry. Only when storm surges propagate through continuous wetlands that are several thousand to tens of thousands of meters wide can wetlands have a considerable effect on storm surge attenuation (**Figure 2**). However, this attenuation is lost when there are large or many tidal channels running through the wetland, which facilitates effective landward storm surge propagation (**Figure 2**). Further research is needed to quantify the conditions under which storm surge attenuation by wetlands can be effective or not, including more observational data from water level sensors deployed during storm surges to verify model outcomes. In particular, our understanding of the dependency of surge attenuation on the larger-scale coastal landscape setting is only fragmentary (**Figure 3**). Integration of storm surge monitoring and hydrodynamic modeling in different types of wetland-holding coastal systems (deltas, estuaries, and back-barrier lagoons of different dimensions and geometries) is key and should cover interactions between the larger system scale and smaller within-wetland scale, both of which are relevant for the propagation of surges.

6.2. How Will the Functionality and Persistence of Tidal Wetlands as Coastal Storm Buffers Evolve in the Long Term in Response to Global Change Factors?

The functionality of tidal wetlands as storm buffers critically depends on the storm disturbance–recovery dynamics of the ecosystem (**Figure 4**). Many case studies have shown that established wetlands have a rather strong resistance and resilience to storm erosion impacts, although mangrove vegetation is generally more vulnerable to structural wind damage than marsh vegetation. A critical question is how the storm disturbance–recovery dynamics are affected by global changes that induce additional stresses or disturbances, such as sea level rise, increasing storminess in certain regions, increasing temperature and droughts, and anthropogenic impacts on sediment budgets and nutrient loading. These combined stresses may induce slower ecosystem recovery after a storm disturbance and may eventually push the ecosystem trajectory beyond a tipping point, after which wetlands degrade and convert to permanent bare mudflats or shallow water. Determining the response of tidal wetland trajectories to multiple stressors affected by global change represents a major challenge in tidal wetland research that to date remains largely unresolved. This question implies that a key uncertainty exists regarding the long-term functionality and persistence of tidal wetlands as coastal storm buffers.

6.3. How Should We Deal with Uncertainty in the Implementation and Long-Term Reliability of Nature-Based Coastal Defenses?

Coastal defense systems, traditionally consisting of engineered defense structures such as seawalls, dikes, or levees, benefit considerably from wetland conservation, restoration, or creation in front of these structures. Observations and quantitative analyses have shown that wetland foreshores can substantially reduce the wave loading on flood defense structures, thereby reducing the risks and impacts of structural failure of flood defense structures during storms. This knowledge has recently opened a new field of research in ecological engineering, exploring the potential and limitations of a diversity of so-called hybrid and nature-based approaches to coastal defense. However, several key uncertainties are still limiting the broad-scale implementation of hybrid and nature-based coastal defense. For example, we are unsure about the critical environmental and biological conditions under which new tidal wetlands can be created and can persist over time for nature-based coastal defense, which adaptive management strategies can be applied to promote the successful establishment of new wetlands, how fast new wetlands can develop their functionality in terms of wave attenuation and erosion mitigation, through which management practices we can optimize the functionality and persistence of existing wetlands as protective storm buffers, and what societal and governance structures are needed for implementation of nature-based coastal defense. To advance fundamental understanding of the functionality and persistence of tidal wetlands as coastal storm buffers and to evolve toward successful implementation and management of adaptive nature-based coastal defense systems, further integration of research programs is needed, combining ecological, hydrodynamic, geomorphological, biogeochemical, engineering, socioeconomic, and governance expertise.

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LITERATURE CITED

Andersen TJ, Fredsoe J, Pejrup M. 2007. In situ estimation of erosion and deposition thresholds by Acoustic Doppler Velocimeter (ADV). *Estuar. Coast. Shelf Sci.* 75:327–36

Anderson ME, Smith JM. 2014. Wave attenuation by flexible, idealized salt marsh vegetation. *Coast. Eng.* 83:82–92

Arkema KK, Guannel G, Verutes G, Wood SA, Guerry A, et al. 2013. Coastal habitats shield people and property from sea-level rise and storms. *Nat. Clim. Change* 3:913–18

Armitage AR, Weaver CA, Kominoski JS, Pennings SC. 2020. Resistance to hurricane effects varies among wetland vegetation types in the marsh-mangrove ecotone. *Estuaries Coasts* 43:960–70

Auerbach LW, Goodbred SL, Mondal DR, Wilson CA, Ahmed KR, et al. 2015. Flood risk of natural and embanked landscapes on the Ganges-Brahmaputra tidal delta plain. *Nat. Clim. Change* 5:153–57

Augustin LN, Irish JL, Lynett P. 2009. Laboratory and numerical studies of wave damping by emergent and near-emergent wetland vegetation. *Coast. Eng.* 56:332–40

Baird AH, Bhalla RS, Kerr AM, Pelkevy NW, Srinivas V. 2009. Do mangroves provide an effective barrier to storm surges? *PNAS* 106:E111

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR. 2011. The value of estuarine and coastal ecosystem services. *Ecol. Monogr.* 81:169–93

Baustian JJ, Mendelssohn IA. 2015. Hurricane-induced sedimentation improves marsh resilience and vegetation vigor under high rates of relative sea level rise. *Wetlands* 35:795–802

Borsje BW, Van Wesenbeeck BK, Dekker F, Paalvast P, Bouma TJ, et al. 2011. How ecological engineering can serve in coastal protection. *Ecol. Eng.* 37:113–22

Bouma TJ, Friedrichs M, Klaassen P, van Wesenbeeck BK, Brun FG, et al. 2009. Effects of shoot stiffness, shoot size and current velocity on scouring sediment from around seedlings and propagules. *Mar. Ecol. Prog. Ser.* 388:293–97

Bouma TJ, van Belzen J, Balke T, Zhu ZC, Airoldi L, et al. 2014. Identifying knowledge gaps hampering application of intertidal habitats in coastal protection: opportunities and steps to take. *Coast. Eng.* 87:147–57

Breithaupt JL, Smoak JM, Byrne RH, Waters MN, Moyer RP, Sanders CJ. 2018. Avoiding timescale bias in assessments of coastal wetland vertical change. *Limnol. Oceanogr.* 63:S477–95

Brooks H, Möller I, Carr S, Chirol C, Christie E, et al. 2021. Resistance of salt marsh substrates to near-instantaneous hydrodynamic forcing. *Earth Surf. Process. Landf.* 46:67–88

Cahoon DR. 2006. A review of major storm impacts on coastal wetland elevations. *Estuaries Coasts* 29:889–98

Cahoon DR, Hensel P, Rybczyk J, McKee KL, Proffitt CE, Perez BC. 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *J. Ecol.* 91:1093–105

Cahoon DR, McKee KL, Morris JT. 2021. How plants influence resilience of salt marsh and mangrove wetlands to sea-level rise. *Estuaries Coasts* 44:883–98

Carr J, Mariotti G, Fahgerazzi S, McGlathery K, Wiberg P. 2018. Exploring the impacts of seagrass on coupled marsh-tidal flat morphodynamics. *Front. Environ. Sci.* 6:92

Cassalho F, Miesse TW, de Lima AD, Khalid A, Ferreira CM, Sutton-Grier AE. 2021. Coastal wetlands exposure to storm surge and waves in the Albemarle-Pamlico estuarine system during extreme events. *Wetlands* 41:49

Castagno KA, Tomiczek T, Shepard CC, Beck MW, Bowden AA, et al. 2021. Resistance, resilience, and recovery of salt marshes in the Florida Panhandle following Hurricane Michael. *Sci. Rep.* 11:20381

Castaneda-Moya E, Rivera-Monroy VH, Chambers RM, Zhao XC, Lamb-Wotton L, et al. 2020. Hurricanes fertilize mangrove forests in the Gulf of Mexico (Florida Everglades, USA). *PNAS* 117:4831–41

Chen H, Ni Y, Li YL, Liu F, Ou SY, et al. 2018. Deriving vegetation drag coefficients in combined wave-current flows by calibration and direct measurement methods. *Adv. Water Res.* 122:217–27

Chen Q, Li YP, Kelly DM, Zhang KQ, Zachry B, Rhome J. 2021. Improved modeling of the role of mangroves in storm surge attenuation. *Estuar. Coast. Shelf Sci.* 260:107515

Coleman DJ, Schuerch M, Temmerman S, Guntenspergen G, Smith CG, Kirwan ML. 2022. Reconciling models and measurements of marsh vulnerability to sea level rise. *Limnol. Oceanogr. Lett.* 7:140–49

Costanza R, Perez-Maqueo O, Martinez ML, Sutton P, Anderson SJ, Mulder K. 2008. The value of coastal wetlands for hurricane protection. *Ambio* 37:241–48

Dahdouh-Guebas F, Koedam N. 2006. Coastal vegetation and the Asian tsunami. *Science* 311:37–38

Dalrymple RA, Kirby JT, Hwang PA. 1984. Wave diffraction due to areas of energy dissipation. *J. Waterw. Port Coast. Ocean Eng.* 110:67–79

Danielsen F, Sorensen K, Olwig MF, Selvam V, Parish F, et al. 2005. The Asian tsunami: a protective role for coastal vegetation. *Science* 310:643

Das S, Vincent JR. 2009. Mangroves protected villages and reduced death toll during Indian super cyclone. *PNAS* 106:7357–60

Day JW, Boesch DF, Clairain EJ, Kemp GP, Laska SB, et al. 2007. Restoration of the Mississippi Delta: lessons from Hurricanes Katrina and Rita. *Science* 315:1679–84

Deb M, Ferreira CM. 2017. Potential impacts of the Sunderban mangrove degradation on future coastal flooding in Bangladesh. *J. Hydroenviron. Res.* 17:30–46

del Valle A, Eriksson M, Ishizawa OA, Miranda JJ. 2020. Mangroves protect coastal economic activity from hurricanes. *PNAS* 117:265–70

Fairchild TP, Bennett WG, Smith G, Day B, Skov MW, et al. 2021. Coastal wetlands mitigate storm flooding and associated costs in estuaries. *Environ. Res. Lett.* 16:074034

Feagin RA, Mukherjee N, Shanker K, Baird AH, Cinner J, et al. 2010. Shelter from the storm? Use and misuse of coastal vegetation bioshields for managing natural disasters. *Conserv. Lett.* 3:1–11

Feher LC, Osland MJ, Anderson GH, Vervaeke WC, Krauss KW, et al. 2020. The long-term effects of Hurricanes Wilma and Irma on soil elevation change in Everglades mangrove forests. *Ecosystems* 23:917–31

Ferrario F, Beck MW, Storlazzi CD, Micheli F, Shepard CC, Airoldi L. 2014. The effectiveness of coral reefs for coastal hazard risk reduction and adaptation. *Nat. Commun.* 5:3794

Foster-Martinez MR, Lacy JR, Ferner MC, Varian EA. 2018. Wave attenuation across a tidal marsh in San Francisco Bay. *Coast. Eng.* 136:26–40

Friess DA, Krauss KW, Horstman EM, Balke T, Bouma TJ, et al. 2012. Are all intertidal wetlands naturally created equal? Bottlenecks, thresholds and knowledge gaps to mangrove and saltmarsh ecosystems. *Biol. Rev.* 87:346–66

Friess DA, Rogers K, Lovelock CE, Krauss KW, Hamilton SE, et al. 2019. The state of the world's mangrove forests: past, present, and future. *Annu. Rev. Environ. Resour.* 44:89–115

Garzon JL, Maza M, Ferreira CM, Lara JL, Losada IJ. 2019a. Wave attenuation by *Spartina* saltmarshes in the Chesapeake Bay under storm surge conditions. *J. Geophys. Res. Oceans* 124:5220–43

Garzon JL, Miesse T, Ferreira CM. 2019b. Field-based numerical model investigation of wave propagation across marshes in the Chesapeake Bay under storm conditions. *Coast. Eng.* 146:32–46

Gedan KB, Kirwan ML, Wolanski E, Barbier EB, Silliman BR. 2011. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. *Clim. Change* 106:7–29

Gijsman R, Horstman EM, van der Wal D, Friess DA, Swales A, Wijnberg KM. 2021. Nature-based engineering: a review on reducing coastal flood risk with mangroves. *Front. Mar. Sci.* 8:702412

Gillen MN, Messerschmidt TC, Kirwan ML. 2021. Biophysical controls of marsh soil shear strength along an estuarine salinity gradient. *Earth Surf. Dyn.* 9:413–21

Glass EM, Garzon JL, Lawler S, Paquier E, Ferreira CM. 2018. Potential of marshes to attenuate storm surge water level in the Chesapeake Bay. *Limnol. Oceanogr.* 63:951–67

Guannel G, Arkema K, Ruggiero P, Verutes G. 2016. The power of three: coral reefs, seagrasses and mangroves protect coastal regions and increase their resilience. *PLOS ONE* 11:e0158094

Haddad J, Lawler S, Ferreira CM. 2016. Assessing the relevance of wetlands for storm surge protection: a coupled hydrodynamic and geospatial framework. *Nat. Hazards* 80:839–61

Hanley ME, Bouma TJ, Mossman HL. 2020. The gathering storm: optimizing management of coastal ecosystems in the face of a climate-driven threat. *Ann. Bot.* 125:197–212

Henderson SM, Norris BK, Mullarney JC, Bryan KR. 2017. Wave-frequency flows within a near-bed vegetation canopy. *Cont. Shelf Res.* 147:91–101

Himmelstein J, Vinent OD, Temmerman S, Kirwan ML. 2021. Mechanisms of pond expansion in a rapidly submerging marsh. *Front. Mar. Sci.* 8:704768

Hochard JP, Barbier EB, Hamilton SE. 2021. Mangroves and coastal topography create economic "safe havens" from tropical storms. *Sci. Rep.* 11:15359

Hochard JP, Hamilton S, Barbier EB. 2019. Mangroves shelter coastal economic activity from cyclones. *PNAS* 116:12232–37

Horstman EM, Bryan KR, Mullarney JC. 2021. Drag variations, tidal asymmetry and tidal range changes in a mangrove creek system. *Earth Surf. Process. Landf.* 46:1828–46

Horstman EM, Dohmen-Janssen CM, Narra PMF, van den Berg NJF, Siemerink M, Hulscher S. 2014. Wave attenuation in mangroves: a quantitative approach to field observations. *Coast. Eng.* 94:47–62

Horstman EM, Lundquist CJ, Bryan KR, Bulmer RH, Mullarney JC, Stokes DJ. 2018. The dynamics of expanding mangroves in New Zealand. In *Threats to Mangrove Forests: Hazards, Vulnerability, and Management*, ed. C Makowski, CW Finkl, pp. 23–51. Cham, Switz.: Springer

Howes NC, Fitzgerald DM, Hughes ZJ, Georgiou IY, Kulp MA, et al. 2010. Hurricane-induced failure of low salinity wetlands. *PNAS* 107:14014–19

Hu KL, Chen Q, Wang HQ. 2015. A numerical study of vegetation impact on reducing storm surge by wetlands in a semi-enclosed estuary. *Coast. Eng.* 95:66–76

Hu Z, Borsje BW, Belzen J, Willemsen PWJM, Wang H, et al. 2021. Mechanistic modeling of marsh seedling establishment provides a positive outlook for coastal wetland restoration under global climate change. *Geophys. Res. Lett.* 48:e2021GL095596

Hu Z, Suzuki T, Zitman T, Uittewaal W, Stive M. 2014. Laboratory study on wave dissipation by vegetation in combined current-wave flow. *Coast. Eng.* 88:131–42

Jadhav RS, Chen Q, Smith JM. 2013. Spectral distribution of wave energy dissipation by salt marsh vegetation. *Coast. Eng.* 77:99–107

Jones MC, Bernhardt CE, Krauss KW, Noe GB. 2017. The impact of late Holocene land use change, climate variability, and sea level rise on carbon storage in tidal freshwater wetlands on the southeastern United States coastal plain. *J. Geophys. Res. Biogeosci.* 122:3126–41

Kathiresan K, Rajendran N. 2005. Coastal mangrove forests mitigated tsunami. *Estuar. Coast. Shelf Sci.* 65:601–6

Kerr AM, Baird AH, Campbell SJ. 2006. Comments on "Coastal mangrove forests mitigated tsunami" by K. Kathiresan and N. Rajendran [Estuar. Coast. Shelf Sci. 65 (2005) 601–606]. *Estuar. Coast. Shelf Sci.* 67:539–41

Kiesel J, Schuerch M, Möller I, Spencer T, Vafeidis A. 2019. Attenuation of high water levels over restored saltmarshes can be limited. Insights from Freiston Shore, Lincolnshire, UK. *Ecol. Eng.* 136:89–100

Kobayashi N, Raichle AW, Asano T. 1993. Wave attenuation by vegetation. *J. Waterw. Port Coast. Ocean Eng.* 119:30–48

Krauss KW, Doyle TW, Doyle TJ, Swarzenski CM, From AS, et al. 2009. Water level observations in mangrove swamps during two hurricanes in Florida. *Wetlands* 29:142–49

Krauss KW, Osland MJ. 2020. Tropical cyclones and the organization of mangrove forests: a review. *Ann. Bot.* 125:213–34

Ladd CJT, Duggan-Edwards MF, Bouma TJ, Pagès JF, Skov MW. 2019. Sediment supply explains long-term and large-scale patterns in salt marsh lateral expansion and erosion. *Geophys. Res. Lett.* 46:11178–87

Lawler S, Haddad J, Ferreira CM. 2016. Sensitivity considerations and the impact of spatial scaling for storm surge modeling in wetlands of the Mid-Atlantic region. *Ocean Coast. Manag.* 134:226–38

Leonardi N, Camacina I, Donatelli C, Ganju NK, Plater AJ, et al. 2018. Dynamic interactions between coastal storms and salt marshes: a review. *Geomorphology* 301:92–107

Leonardi N, Ganju NK, Fagherazzi S. 2016. A linear relationship between wave power and erosion determines salt-marsh resilience to violent storms and hurricanes. *PNAS* 113:64–68

Lin N, Emanuel K, Oppenheimer M, Vanmarcke E. 2012. Physically based assessment of hurricane surge threat under climate change. *Nat. Clim. Change* 2:462–67

Liu HQ, Zhang KQ, Li YP, Xie L. 2013. Numerical study of the sensitivity of mangroves in reducing storm surge and flooding to hurricane characteristics in southern Florida. *Cont. Shelf Res.* 64:51–65

Liu X, Wang YB, Costanza R, Kubiszewski I, Xu N, et al. 2019. The value of China's coastal wetlands and seawalls for storm protection. *Ecosyst. Serv.* 36:100905

Loder NM, Irish JL, Cialone MA, Wamsley TV. 2009. Sensitivity of hurricane surge to morphological parameters of coastal wetlands. *Estuar. Coast. Shelf Sci.* 84:625–36

Lovelace JK. 1994. *Storm-tide elevations produced by Hurricane Andrew along the Louisiana coast, August 25–27, 1992.* Open-File Rep. 94-371, US Geol. Surv., Baton Rouge, LA

Lovelock CE, Cahoon DR, Friess DA, Guntenspergen GR, Krauss KW, et al. 2015. The vulnerability of Indo-Pacific mangrove forests to sea-level rise. *Nature* 526:559–63

Luhar M, Nepf HM. 2016. Wave-induced dynamics of flexible blades. *J. Fluids Struct.* 61:20–41

Marsooli R, Orton PM, Georgas N, Blumberg AF. 2016. Three-dimensional hydrodynamic modeling of coastal flood mitigation by wetlands. *Coast. Eng.* 111:83–94

Marsooli R, Orton PM, Mellor G. 2017. Modeling wave attenuation by salt marshes in Jamaica Bay, New York, using a new rapid wave model. *J. Geophys. Res. Oceans* 122:5689–707

Maza M, Lara JL, Losada IJ. 2019. Experimental analysis of wave attenuation and drag forces in a realistic fringe *Rhizophora* mangrove forest. *Adv. Water Res.* 131:103376

Maza M, Lara JL, Losada IJ. 2021. Predicting the evolution of coastal protection service with mangrove forest age. *Coast. Eng.* 168:103922

Mazda Y, Magi M, Ikeda Y, Kurokawa T, Asano T. 2006. Wave reduction in a mangrove forest dominated by Sonneratia sp. *Wetl. Ecol. Manag.* 14:365–78

McGee BD, Goree BB, Tollet RW, Woodward BK, Kress WH. 2006. *Hurricane Rita surge data, Southwestern Louisiana and Southeastern Texas, September-November 2005*. Data Ser. Rep. 220, US Geol. Surv., Baton Rouge, LA

Mendez FJ, Losada IJ. 2004. An empirical model to estimate the propagation of random breaking and non-breaking waves over vegetation fields. *Coast. Eng.* 51:103–18

Menendez P, Losada IJ, Torres-Ortega S, Narayan S, Beck MW. 2020. The global flood protection benefits of mangroves. *Sci. Rep.* 10:4404

Möller I, Kudella M, Rupprecht F, Spencer T, Paul M, et al. 2014. Wave attenuation over coastal salt marshes under storm surge conditions. *Nat. Geosci.* 7:727–31

Montgomery JM, Bryan KR, Horstman EM, Mullarney JC. 2018. Attenuation of tides and surges by mangroves: contrasting case studies from New Zealand. *Water* 10:1119

Montgomery JM, Bryan KR, Mullarney JC, Horstman EM. 2019. Attenuation of storm surges by coastal mangroves. *Geophys. Res. Lett.* 46:2680–89

Morris RL, Konlechner TM, Ghisalberti M, Swearer SE. 2018. From grey to green: efficacy of eco-engineering solutions for nature-based coastal defence. *Glob. Change Biol.* 24:1827–42

Mullarney JC, Henderson SM. 2010. Wave-forced motion of submerged single-stem vegetation. *J. Geophys. Res. Oceans* 115:C12061

Mullarney JC, Henderson SM. 2018. Flows within marine vegetation canopies. In *Advances in Coastal Hydraulics*, ed. V Panchang, J Kaihatu, pp. 1–46. Singapore: World Sci.

Narayan S, Beck MW, Reguero BG, Losada IJ, van Wesenbeeck B, et al. 2016. The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PLOS ONE* 11:e0154735

Neumann B, Vafeidis AT, Zimmermann J, Nicholls RJ. 2015. Future coastal population growth and exposure to sea-level rise and coastal flooding – a global assessment. *PLOS ONE* 10:e0131375

Nguyen HM, Bryan KR, Pilditch CA. 2020. The effect of long-term aerial exposure on intertidal mudflat erodibility. *Earth Surf. Process. Landf.* 45:3623–38

Norris BK, Mullarney JC, Bryan KR, Henderson SM. 2021. Relating millimeter-scale turbulence to meter-scale subtidal erosion and accretion across the fringe of a coastal mangrove forest. *Earth Surf. Process. Landf.* 46:573–92

Orton PM, Sanderson EW, Talke SA, Giampieri M, MacManus K. 2020. Storm tide amplification and habitat changes due to urbanization of a lagoonal estuary. *Nat. Hazards Earth Syst. Sci.* 20:2415–32

Oteman B, Morris EP, Peralta G, Bouma TJ, van der Wal D. 2019. Using remote sensing to identify drivers behind spatial patterns in the bio-physical properties of a saltmarsh pioneer. *Remote Sens.* 11:511

Paul M, Amos CL. 2011. Spatial and seasonal variation in wave attenuation over *Zostera noltii*. *J. Geophys. Res. Oceans* 116:C08019

Paul M, Rupprecht F, Möller I, Bouma TJ, Spencer T, et al. 2016. Plant stiffness and biomass as drivers for drag forces under extreme wave loading: a flume study on mimics. *Coast. Eng.* 117:70–78

Pennings SC, Glazner RM, Hughes ZJ, Kominoski JS, Armitage AR. 2021. Effects of mangrove cover on coastal erosion during a hurricane in Texas, USA. *Ecology* 102:e03309

Phan KL, Stive MJF, Zijlema M, Truong HS, Aarninkhof SGJ. 2019. The effects of wave non-linearity on wave attenuation by vegetation. *Coast. Eng.* 147:63–74

Pinsky ML, Guannel G, Arkema KK. 2013. Quantifying wave attenuation to inform coastal habitat conservation. *Ecosphere* 4:1–16

Primavera JH, Esteban JMA. 2008. A review of mangrove rehabilitation in the Philippines: successes, failures and future prospects. *Wetl. Ecol. Manag.* 16:345–58

Quartel S, Kroon A, Augustinus P, Van Santen P, Tri NH. 2007. Wave attenuation in coastal mangroves in the Red River Delta, Vietnam. *J. Asian Earth Sci.* 29:576–84

Radabaugh KR, Moyer RP, Chappel AR, Dontis EE, Russo CE, et al. 2020. Mangrove damage, delayed mortality, and early recovery following Hurricane Irma at two landfall sites in southwest Florida, USA. *Estuaries Coasts* 43:1104–18

Resio DT, Westerink JJ. 2008. Modelling the physics of storm surges. *Phys. Today* 61:33–38

Riffe KC, Henderson SM, Mullarney JC. 2011. Wave dissipation by flexible vegetation. *Geophys. Res. Lett.* 38:L18607

Roner M, Ghinassi M, Finotello A, Bertini A, Combourieu-Nebout N, et al. 2021. Detecting the delayed signatures of changing sediment supply in salt-marsh landscapes: the case of the Venice Lagoon (Italy). *Front. Mar. Sci.* 8:742603

Rupprecht F, Möller I, Paul M, Kudella M, Spencer T, et al. 2017. Vegetation-wave interactions in salt marshes under storm surge conditions. *Ecol. Eng.* 100:301–15

Sanchez-Gonzalez JF, Sanchez-Rojas V, Memos CD. 2011. Wave attenuation due to *Posidonia oceanica* meadows. *J. Hydraul. Res.* 49:503–14

Schoonees T, Mancheno AG, Scheres B, Houma TJ, Silva R, et al. 2019. Hard structures for coastal protection, towards greener designs. *Estuaries Coasts* 42:1709–29

Schoutens K, Heuner M, Fuchs E, Minden V, Schulte-Ostermann T, et al. 2020. Nature-based shoreline protection by tidal marsh plants depends on trade-offs between avoidance and attenuation of hydrodynamic forces. *Estuar. Coast. Shelf Sci.* 236:106645

Schoutens K, Heuner M, Minden V, Schulte Ostermann T, Silinski A, et al. 2019. How effective are tidal marshes as nature-based shoreline protection throughout seasons? *Limnol. Oceanogr.* 64:1750–62

Schoutens K, Reents S, Nolte S, Evans B, Paul M, et al. 2021. Survival of the thickest? Impacts of extreme wave-forcing on marsh seedlings are mediated by species morphology. *Limnol. Oceanogr.* 66:2936–51

Sheng YP, Lapetina A, Ma GF. 2012. The reduction of storm surge by vegetation canopies: three-dimensional simulations. *Geophys. Res. Lett.* 39:L20601

Sheng YP, Rivera-Nieves AA, Zou RZ, Paramygin VA. 2021. Role of wetlands in reducing structural loss is highly dependent on characteristics of storms and local wetland and structure conditions. *Sci. Rep.* 11:5237

Shi BW, Yang SL, Wang YP, Bouma TJ, Zhu Q. 2012. Relating accretion and erosion at an exposed tidal wetland to the bottom shear stress of combined current–wave action. *Geomorphology* 138:380–89

Silinski A, Heuner M, Troch P, Puijalon S, Bouma TJ, et al. 2016. Effects of contrasting wave conditions on scour and drag on pioneer tidal marsh plants. *Geomorphology* 255:49–62

Silliman BR, He Q, Angelini C, Smith CS, Kirwan ML, et al. 2019. Field experiments and meta-analysis reveal wetland vegetation as a crucial element in the coastal protection paradigm. *Curr. Biol.* 29:1800–6.e3

Smith JM, Bryant MA, Wamsley TV. 2016. Wetland buffers: numerical modeling of wave dissipation by vegetation. *Earth Surf. Process. Landf.* 41:847–54

Smith TJ, Anderson GH, Balentine K, Tiling G, Ward GA, Whelan KRT. 2009. Cumulative impacts of hurricanes on Florida mangrove ecosystems: sediment deposition, storm surges and vegetation. *Wetlands* 29:24–34

Smith TJ, Robblee MB, Wanless HR, Doyle TW. 1994. Mangroves, hurricanes, and lightning strikes. *BioScience* 44:256–62

Smolders S, Plancke Y, Ides S, Meire P, Temmerman S. 2015. Role of intertidal wetlands for tidal and storm tide attenuation along a confined estuary: a model study. *Nat. Hazards Earth Syst. Sci.* 15:1659–75

Spencer T, Möller I, Rupprecht F, Bouma TJ, van Wesenbeeck BK, et al. 2016. Salt marsh surface survives true-to-scale simulated storm surges. *Earth Surf. Process. Landf.* 41:543–52

Stark J, Plancke Y, Ides S, Meire P, Temmerman S. 2016. Coastal flood protection by a combined nature-based and engineering approach: modeling the effects of marsh geometry and surrounding dikes. *Estuar. Coast. Shelf Sci.* 175:34–45

Stark J, Van Oyen T, Meire P, Temmerman S. 2015. Observations of tidal and storm surge attenuation in a large tidal marsh. *Limnol. Oceanogr.* 60:1371–81
Sutton-Grier AE, Wowk K, Bamford H. 2015. Future of our coasts: the potential for natural and hybrid infrastructure to enhance the resilience of our coastal communities, economies and ecosystems. *Environ. Sci. Policy* 51:137–48
Swarzenski CM, Doyle TW, Fry B, Hargis TG. 2008. Biogeochemical response of organic-rich freshwater marshes in the Louisiana delta plain to chronic river water influx. *Biogeochemistry* 90:49–63
Taillie PJ, Roman-Cuesta R, Lagomasino D, Cifuentes-Jara M, Fatoyinbo T, et al. 2020. Widespread mangrove damage resulting from the 2017 Atlantic mega hurricane season. *Environ. Res. Lett.* 15:064010
Temmerman S, De Vries MB, Bouma TJ. 2012. Coastal marsh die-off and reduced attenuation of coastal floods: a model analysis. *Glob. Planet. Change* 92–93:267–74
Temmerman S, Meire P, Bouma TJ, Herman PMJ, Ysebaert T, De Vriend HJ. 2013. Ecosystem-based coastal defence in the face of global change. *Nature* 504:79–83
Tessler ZD, Vörösmarty CJ, Grossberg M, Gladkova I, Aizenman H, et al. 2015. Profiling risk and sustainability in coastal deltas of the world. *Science* 349:638–43
Thampanya U, Vermaat JE, Sinsakul S, Panapitukkul N. 2006. Coastal erosion and mangrove progradation of Southern Thailand. *Estuar. Coast. Shelf Sci.* 68:75–85
Tinoco RO, San Juan JE, Mullarney JC. 2020. Simplification bias: lessons from laboratory and field experiments on flow through aquatic vegetation. *Earth Surf. Process. Landf.* 45:121–43
USACE (US Army Corps Eng.). 1963. *Interim survey report, Morgan city, Louisiana and vicinity*. Ser. 63, USACE, New Orleans, LA
Van Coppenolle R, Schwartz C, Temmerman S. 2018. Contribution of mangroves and salt marshes to nature-based mitigation of coastal flood risks in major deltas of the world. *Estuaries Coasts* 41:1699–711
van Rooijen AA, McCall RT, de Vries J, van Dongeren AR, Reniers A, Roelvink JA. 2016. Modeling the effect of wave-vegetation interaction on wave setup. *J. Geophys. Res. Oceans* 121:4341–59
van Veelen TJ, Karunarathna H, Reeve DE. 2021. Modelling wave attenuation by quasi-flexible coastal vegetation. *Coast. Eng.* 164:103820
van Wesenbeeck BK, Wolters G, Antolinez JAA, Kalloe SA, Hofland B, et al. 2022. Wave attenuation through forests under extreme conditions. *Sci. Rep.* 12:1884
van Zelst VTM, Dijkstra JT, van Wesenbeeck BK, Eilander D, Morris EP, et al. 2021. Cutting the costs of coastal protection by integrating vegetation in flood defences. *Nat. Commun.* 12:6533
Vandenbruwaene W, Schwarz C, Bouma TJ, Meire P, Temmerman S. 2015. Landscape-scale flow patterns over a vegetated tidal marsh and an unvegetated tidal flat: implications for the landform properties of the intertidal floodplain. *Geomorphology* 231:40–52
Vuik V, Heo HYS, Zhu ZC, Borsje BW, Jonkman SN. 2018. Stem breakage of salt marsh vegetation under wave forcing: a field and model study. *Estuar. Coast. Shelf Sci.* 200:41–58
Vuik V, Jonkman SN, Borsje BW, Suzuki T. 2016. Nature-based flood protection: the efficiency of vegetated foreshores for reducing wave loads on coastal dikes. *Coast. Eng.* 116:42–56
Walters DC, Kirwan ML. 2016. Optimal hurricane overwash thickness for maximizing marsh resilience to sea level rise. *Ecol. Evol.* 6:2948–56
Wamsley TV, Cialone MA, Smith JM, Atkinson JH, Rosati JD. 2010. The potential of wetlands in reducing storm surge. *Ocean Eng.* 37:59–68
Watts CW, Tolhurst TJ, Black KS, Whitmore AP. 2003. In situ measurements of erosion shear stress and geotechnical shear strength of the intertidal sediments of the experimental managed realignment scheme at Tollesbury, Essex, UK. *Estuar. Coast. Shelf Sci.* 58:611–20
Willard DA, Cronin TM, Verardo S. 2003. Late-Holocene climate and ecosystem history from Chesapeake Bay sediment cores, USA. *Holocene* 13:201–14
Willemsen PWJM, Horstman EM, Bouma TJ, Baptist MJ, van Puijenbroek MEB, Borsje BW. 2022. Facilitating salt marsh restoration: the importance of event-based bed level dynamics and seasonal trends in bed level change. *Front. Mar. Sci.* 8:793235
Williams HFL, Flanagan WM. 2009. Contribution of Hurricane Rita storm surge deposition to long-term sedimentation in Louisiana coastal woodlands and marshes. *J. Coast. Res.* Spec. Issue 56:1671–75

Yin K, Xu SD, Huang WR, Liu S, Li MX. 2021. Numerical investigation of wave attenuation by coupled flexible vegetation dynamic model and XBeach wave model. *Ocean Eng.* 235:109357

Zhang KQ, Liu HQ, Li YP, Xu HZ, Shen J, et al. 2012. The role of mangroves in attenuating storm surges. *Estuar. Coast. Shelf Sci.* 102:11–23

Zhu Z, Vuik V, Visser PJ, Soens T, van Wesenbeeck B, et al. 2020. Historic storms and the hidden value of coastal wetlands for nature-based flood defence. *Nat. Sustain.* 3:853–62

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