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# Tidal flat-wetland systems as flood defenses: Understanding biogeomorphic controls

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ABSTRACT

Coastal managers worldwide increasingly recognize the importance of conservation and restoration of natural coastal ecosystems. This ensures coastal resilience and provision of essential ecosystem services, such as wave attenuation reducing coastal flooding and erosion. In the continuum from unvegetated tidal flats to salt marshes and mangroves, fundamental physical controls as well as biotic interactions, and feedbacks among them, determine morphology and vegetation distribution. Although these processes are well described in established literature, this information is rarely applied to understanding the role of these ecosystems as coastal defense. The focus is often on specific elements of the complex system, such as vegetation structure and cover, rather than on their complex natural dynamics. This review examines whether and how the dynamic nature of tidal flat - wetlands systems contributes to, or detracts from, their role in coastal defense. It discusses how the characteristics of the system adjust to external forcing and how these adjustments affect ecosystem services. It also considers how human interventions can take advantage of natural processes to enhance or accelerate achievement of natural coastal defense.

## 1. Introduction

There is a broad consensus among coastal managers concerning the importance of conserving or restoring natural systems. This contributes to coastal resilience and ecosystem service provision, such as the attenuation of waves to reduce coastal flooding and marginal erosion (e.g., Spalding et al., 2014 and references therein). Governmental assessments and formal planning procedures (e.g. State of Queensland, 2012; European Environment Agency, 2015; National Science and Technology Council, 2015) increasingly respond to calls for a more holistic appreciation of 'natural infrastructure' in coastal decision making, which for some time have been coming from scientific and non-governmental sources (e.g., Shepard et al., 2011; Beck et al., 2012; Sutton-Grier et al., 2015). The role of natural system features in providing protection for coastal communities gains traction after each natural flooding disaster. The 2004 Indian Ocean tsunami, Hurricane Katrina in 2005 on the northern Gulf of Mexico, and 'superstorm' Sandy in the north-east US have each prompted serious examination of the potential 'bioshield' effect of coastal wetlands by less traditional advocates, such as governments or the insurance industry (Broadhead and Leslie, 2007; Bridges et al., 2013; Narayan et al., 2017).

Scientists have sought to provide data and models to inform the integration of natural features into coastal risk reduction planning. Potential functionality has been quantified for many systems, e.g., by multiple studies of wave attenuation by coastal vegetation (e.g., Quartel et al., 2007; Horstman et al., 2014; Möller et al., 2014; Foster-Martinez et al., 2018). These direct measurements illustrate that wetlands reduce impacts of waves, while other studies show the potential of systems to reduce economic damage (Narayan et al., 2016; Barbier et al., 2013). For storm surges and extreme events, there have been fewer direct field measurements (Stark et al., 2015; Paquier et al., 2017) but numerical modeling approaches have identified key factors influencing storm surge and wave attenuation (e.g., Loder et al., 2009; Sheng et al., 2012; Marsooli et al., 2016).

In addition to a potential protective role under extreme conditions, several recent global assessments have linked adjacent natural habitats

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to the sustainability of coastal cities and infrastructure (Arkema et al., 2013; Temmerman et al., 2013) in the light of future relative sea-level rise. These generalized studies clarify the need to adapt at large scales (Hinkel et al., 2010; Hallegatte et al., 2013). However, planning and implementation of adaptation measures, especially those that include natural features, require detailed consideration of project objectives and local conditions – past, present and future. Elliot et al. (2016) note several case studies where 'ecoengineering' outcomes have not been as expected, pointing to the need to follow the 10-tenets identified by Barnard and Elliott (2015) that include engineering, environmental, economic as well as socio-political factors.

Decades of scientific research on the processes that control the form and function of coastal wetlands provide a solid foundation for understanding and potentially enhancing the protective role of these systems. Well established literature on salt marshes (e.g., Beeftink, 1966; Chapman, 1960), mangroves (e.g., Thom, et al., 1975) and associated unvegetated tidal flats (Postma, 1967) recognizes both the fundamental physical controls as well as biotic interactions that determine form, vegetation distribution and feedbacks between them. Detailed field measurements of processes and novel modeling approaches have enabled process-based simulation of these interactions and the prediction of patterns of change decades into the future. The predictions include the potential effects of changes in external forcing, such as sea-level rise and sediment supply, on coastal wetland systems. This rapidly developing area of study is mostly focused on understanding the fundamental controls and dynamics of the systems. In contrast, coastal design manuals (e.g., Coulbourne et al., 2011; CPRA, 2015) used by agencies in developing measures to reduce flood risk, often expect certainty (or a high degree of confidence) regarding feature performance. However, more recent guidance documents identify system dynamics as a key consideration (van Wesenbeeck et al., 2017b).

Larger scale application of nature-based flood defense is hampered by a perceived lack of knowledge regarding their usefulness and their sustainability. Quantification of their actual benefits for flood risk reduction and of their dynamic character and strength in the face of extreme events is particularly challenging (Bouma et al., 2014). This is compounded by the absence of generally accepted comprehensive design guidelines. Although the same factors should also be considered in traditional coastal protection design, this is, surprisingly, not always the case (e.g., Mai et al., 2009). Nonetheless, uncertainty is perceived to make nature-based flood defenses less reliable despite potentially lower cost compared to traditional coastal risk reduction measures. Moreover, the focus of wetland nature-based defenses is often on the vegetated wetlands themselves rather than on the entire coastal setting within which coastal wetlands have evolved and are sustained. This setting includes unvegetated tidal flats that also contribute independently to the defense function.

This review examines whether and how the dynamic nature of tidal flat - wetland systems contributes to, or detracts from, their role in coastal defense. It discusses how the characteristics of the system adjust to external forcing, and how these dynamics and management measures enhance the flood defense role. The following questions will guide the discussion:

- How do the changing characteristics of tidal flat-wetland systems influence their role as natural defenses?
- How can management interventions take advantage of natural processes to enhance or accelerate achievement of natural defense functions?

There have been several recent extensive reviews of various aspects of tidal flat-wetland systems including tidal flat morphodynamics (Friedrichs, 2011), advances in modeling (Fagherazzi et al., 2012) and coastal protection by mangroves (Marois and Mitsch, 2015). The purpose here is not to repeat these syntheses but to focus on understanding of gradual transformation or rapid change. The basic question is whether and how these dynamics contribute to the flood defense function. While coastal wetlands are globally diverse, there are some common features which can be used to characterize their morphodynamics. A typology is used here to provide a framework for the evaluation of different types of human interventions.

The review begins with a brief overview of biogeophysical understanding of the development of tidal flat-wetland systems and the key factors influencing morphology and vegetation. Several examples of 'cyclic change' and its impact on flood defense will be examined to illustrate dynamism at different scales. Long term prospects for flood defense are, in that respect, the most critical features. This understanding is then applied to how interventions, e.g., material placement or erosion management, can enhance the role of tidal flats and wetlands in reducing flood risk. The review concludes with discussion of factors that managers and decision makers should consider in the design of coastal risk reduction strategies.

## 2. Development of coastal tidal flat-wetland systems

The long-term development of coastal marshes and mangroves has been studied for over a century by geologists, geomorphologists, and ecologists (e.g., Thom et al., 1975; Frey and Basan, 1978; Allen, 1990; Redfield, 1965) using stratigraphic, dating and ecological reconstructions of the developmental 'stages' underlying the current biogeomorphic profile. Geological conceptual models of estuarine and delta development consider extensive tidal flats and marshes as characteristics of tide dominated systems (Coleman and Roberts, 1989; Dalrymple et al., 1992). Earlier studies (Vann, 1959; Thom, 1967) suggested that vegetation is a secondary factor in deltaic development following the formation of geomorphic features that provide inundation and drainage suitable for specific plants to occupy. Chapman (1960) saw inundation frequency as a key control on colonization by emergent plants. As Adam (1990) notes, it is appropriate to view salt marshes as 'taking advantage of sites where sediment accumulation is already occurring'. Space-for-time analyses, e.g., Pethick (1981), have confirmed lower limits of salt marsh development in relation to tidal inundation, using colonization by plants as the indicator. The initiation of mangrove colonization is similar, as discussed below, with waves and currents being important controls on the distribution and establishment of propagules. However, like marshes, once vegetation is established a complex set of interactions between biotic and physical processes control the development of morphology and vegetation patterns. These patterns are the foundation of the role of coastal wetlands as flood defenses.

While marshes are common in estuaries, in the shelter of islands or in protected bays, there are numerous examples globally of wetlands facing open coasts with extensive tidal flats that reduce wave energy sufficiently for vegetation colonization. Even in estuaries, depending on tidal range, tidal flats and wetlands show strong interdependence. The character of tidal flats has been the subject of considerable theoretical analysis. Fundamental work on sediment dynamics and tidal flows (e.g., Postma, 1967) provides mechanisms for shoreward sediment transport and sediment accumulation. Kirby (1992) identified two endmembers for cross profiles of tidal flats as either concave (net erosional) or convex (net depositional). Friedrichs and Aubrey (1996) note that stable morphology occurs when there is zero net sediment transport, expressed as a uniform distribution of maximum bottom shear stress, with a deviation from the mean resulting in net erosion or deposition. In the absence of waves on a straight shoreline, the equilibrium profile produced by tidal currents alone is convex (Fig. 1B). Wind waves promote concave cross profiles and, as tidal range increases, stronger tidal currents lead to a profile shift from concave to convex. Le Hir et al. (2000), however, conclude that such equilibrium profiles are ephemeral due to seasonal cycles of accretion and erosion. Hu et al. (2015) simulate both long-term and short-term changes in flat morphology in

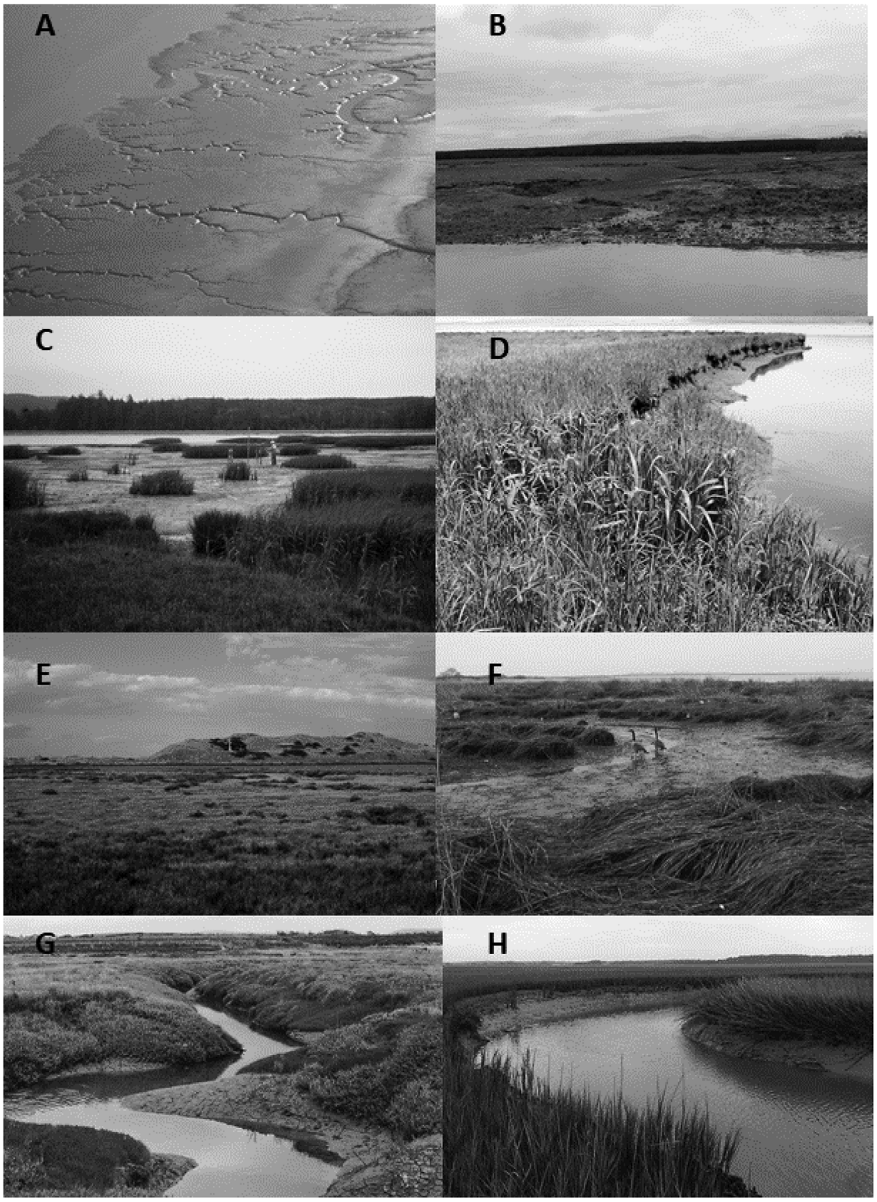

**Fig. 1.** Illustrative examples of tidal flat and marsh biogeomorphology. A. Cook Inlet, Alaska. B. Ria Formosa, Portugal. C Willapa Bay, WA, USA. D. Columbia River estuary, USA E. Scolt Head Island, Norfolk, UK. F. Jamaica Bay, NY USA. G. Wyre estuary, Lancs, UK.H Virginia Coast Reserve, USA.

response to events and other interventions, and show support for dynamic equilibrium based on bed shear stress distribution.

Colonization by emergent plants in the upper intertidal requires the threshold conditions for vegetation establishment to be exceeded. These vary among species. Krauss et al. (2008) describe physiological tolerances for individual mangrove plants and Friess et al. (2012) describe the traits of key pioneer plants for marshes and mangroves, identifying genus *Salicornia* and genus *Spartina* as common for marshes. Friess et al. also note that clonal spreading (Fig. 1C) may explain the greater ability of *Spartina* spp. to withstand tidal inundation and water movement. Wiehe (1935) found that seeds of *Salicornia europea* required 2–3 days without tidal inundation to establish and observed evidence of tidal ‘dragging’ of dead seedlings. Callaway and Josselyn (1992) showed that invasive *Spartina alterniflora* colonized flats at lower elevations than native *Spartina foliosa* in San Francisco Bay, implying species-specific threshold tolerance. Threshold conditions for the establishment of different mangrove species have been identified in many systems (e.g., Ellison and Farnsworth, 1993; McKee, 1995; Delgado et al., 2001; Thampanya et al., 2002; Balke et al., 2013). An interesting outcome of these studies is that hydrodynamic processes and their influence on dispersal have an important role in establishment. This is a move away from the earlier concept that patterns of mangrove vegetation are a result of their physiological tolerance of a narrow set of conditions, governed by substrate type and physiography. In areas with large tidal ranges, the ‘tidal sorting hypothesis’ (Rabinowitz, 1978) has been proposed to explain colonization patterns. Species with larger propagules are better able to establish in deeper water but their landward dispersal is limited by shallow water on the upper tidal flat where species with smaller propagules are favored. In areas with smaller tidal range, however, other factors such as seasonal freshwater inflow may limit the role of tide in dispersal, e.g., Sousa et al. (2007). Further, Balke et al. (2011) used a controlled flume experiment to show that roots 2 cm long were needed to anchor propagules in the sediment and prevent floating during inundation. Longer roots were needed to withstand shear stress from waves and currents, indicating that successful colonization required both appropriate biotic and abiotic conditions.

Once emergent vegetation is established, interaction between plants and physical processes controls further development. Sanchez et al.

(2001) document a logarithmic relationship between vegetation patch diameter and cumulative sediment accretion height in the patch for *Spartina maritima* in NW Spain. High accretion rates in new vegetation patches have been frequently observed in relation to invasions by *Spartina* sp. occupying lower elevations on the tidal flats than native vegetation (Ward et al., 2003). *Spartina* has even been specifically introduced to promote accretion (Chen et al., 2007). Studies of pioneer vegetation patches often appear to be randomly spaced (van Wesenbeeck et al., 2008a). Patches can also increase flow velocities in intervening non-vegetated areas (Temmerman et al., 2005) and expanding patches can channelize flow sufficiently to result in creek formation (Temmerman et al., 2007). In a detailed experimental study, Vandenbruwaene et al. (2011) found incoming flow to be a key control on the merger of patches. Patch expansion increased the flow velocity between patches, but in sheltered areas the accelerated flow could be insufficient for erosion allowing patch coalescence.

Within the vegetated platform, coastal wetlands are rarely homogenous (Fig. 1) with common features including tidal drainage channels or creeks, marshes and mangroves as well as tidal flats. The dynamics of these channels, their velocity characteristics and role in sediment and other constituent flux to and from the marsh surface has been the subject of extensive study in different systems (see Friedrichs and Perry, 2001; Lawrence et al., 2004; D'Alpaos et al., 2007). The initiation of tidal creeks in marshes can be a legacy of shallow creeks in the tidal flat or of patterns in pioneer vegetation colonization (e.g., French and Stoddart, 1992; Vandenbruwaene et al., 2013; Marani et al., 2006) or they may be a legacy of terrestrial drainage in submerging systems (e.g., Gardner and Bohn, 1980). Schwarz et al. (2014) note that vegetation may either stabilize existing channels or initiate channel formation depending on the depth of tidal flat channels and their efficiency in conveying tidal flows. In some settings, the role of burrowing fauna in creating favorable conditions for creek initiation and extension has been identified (e.g., Escapa et al., 2007). This potentially explains observations of rapid headward creek extension in high marsh systems with low creek shear stress (Hughes et al., 2009), although gradual submergence has also been suggested as a causative factor. Many authors have observed the relative stability of tidal creek systems once established (Ashley and Zeff, 1988; Novakowski et al., 2004) even where marginal erosion of creek banks is measurable (Gabet, 1998).

Across the vegetated platform there is also morphological variability. Chapman (1960) described the presence of ‘salt pans’ as typical of coastal marshes. The origins of these ‘small, shallow pools’ (Pethick, 1984) include channel collapse, shading by adjacent vegetation or wrack, lack of initial vegetation colonization, surficial scour or bird foraging (Yapp et al., 1917; Boston, 1983; Tolley and Christian, 1999). While earlier authors considered these features as relatively stable once formed, recent studies (Wilson et al., 2009; Schepers et al., 2017) show they can be dynamic features but do not necessarily signify degradation. They provide topographic and vegetation variations and are common in high marsh areas (especially due to stranded wrack deposits following storms). Pans have not been reported in mangroves although canopy gaps associated with storm disturbances are common (Jimenez et al., 1985). However, topographic variation within otherwise homogenous swamps can be associated with fallen trees (Krauss et al., 2005) and mud mounds associated with burrowing crabs (Minchinton, 2001).

As marsh platforms develop and increase in elevation following vegetation colonization, the system interactions between tidal flat and wetland become more complex. The resulting bimodal distribution of plant biomass and elevation within the system suggests that vegetated patches vs. non-vegetated flats represent alternate stable states (Wang and Temmerman, 2013). However, van Wesenbeeck et al. (2008b) note that, in the long-term, vegetation patches in the Western Scheldt tend to coalesce to form a vegetated marsh platform representing a single more stable state. Rapid accretion following vegetation colonization can also lead to an elevation difference at the vegetated-unvegetated margin, which is then subject to wave attack and erosion, often forming a cliff at the seaward limit of the vegetation (Fig. 1D). Cliffs have been observed in many NW European systems (e.g., van Eerdt, 1985; Pringle, 1995; Pedersen and Bartholdy, 2007; Allen and Haslett, 2014), and as discussed in later sections, cliffs at the marsh-tidal flat interface can contribute to wave energy dissipation. Koppel et al. (2005) found expanding patches of pioneer vegetation in front of eroding cliffs, inferring that the erosion at the marsh edge is part of an intrinsic process of cyclic rejuvenation rather than a sign of changes in external forcing by waves and currents. van der Wal et al. (2008) also note the presence of both tussocks and cliffs, and showed that expansion of tussocks was associated with a decrease in cliff retreat rate. Others have documented apparent cyclic sediment exchanges between tidal flat and marsh surface (e.g., Bouma et al., 2016; Pethick, 1992), while on other marsh margins eroded sediment is removed by waves and currents (Marani et al., 2011) or can be transferred to maintain elevation at the marsh edge (Reed, 1988). Locally, erosion can proceed until overconsolidated muds are encountered resulting in complex bare mud topography (Greensmith and Tucker, 1966; Möller and Spencer, 2002). Mariotti and Fagherazzi (2013) consider the fate of eroded sediment as an important control on whether cliff development is autocyclic, but also identify the role of sediment supply to the marsh/tidal flat system. Francalanci et al. (2013) developed a conceptual model based on flume experiments, with block failure at the marsh edge providing for either continued erosion or the development of a new stable state. Several authors address the issue of initiation of cliff formation (Cox et al., 2003; Houser, 2010), and some point to the importance of storms including Koppel et al. (2005). Houwing (2000) documented storm erosion at the border of the tidal flat and pioneer zone in the Wadden Sea. van der Wal and Pye (2004) examined the initiation of periods of marsh margin retreat in Essex, and found that changes in wave/climatic forcing were important. Marsh margin erosion occurred in response to changes in surrounding coastal configuration until a new state of equilibrium is established (Pethick, 1993). In summary, morphological change at the marsh margin is related to time scale, with episodic or decadal scale external forcing superimposed on mobilization of sediments by waves and tides, mediated by vegetation effects in trapping sediment and binding soils.

## 3. Long term controls of system character and dynamics

The development and dynamics of the features outlined above are controlled at macro temporal and spatial scales by external forcings. This section summarizes system responses to external factors, as a foundation for understanding long-term change in flood defense functions. Pethick (1993) considered tidal flat-marsh systems as interacting parts mutually adjusting to external changes in sediment supply, wave forcing and sea-level rise. This is similar to the concept of coupled shoreface-beach systems used in the study of morphodynamics of sandy coasts (Masselink et al., 2006). The role of external factors in the biogeomorphic character of tidal flat-wetland systems is important for flood defense functionality. It defines long-term cycles or trends and determines factors such as elevation in the intertidal, vegetation type/coverage and the nature of within-system features such as cliffs and channels.

Since Redfield (1965) showed that salt marshes respond to changes in millennial-century scale changes in sea level, many authors have discussed whether and how coastal wetlands can keep pace with relative sea-level rise (e.g., Stevenson et al., 1986; Allen, 1990; Reed, 1995; Cahoon et al., 1995a; Kirwan and Murray, 2008; Fagherazzi, 2013). The balance between surface elevation change and relative sea-level rise is considered a key control on long-term marsh survival. However, there is also an important horizontal component to the long-term survival of marshes. Field measurements, numerical modeling and laboratory studies have noted the importance of marginal erosion of marshes in determining areal extent (e.g., Marani et al., 2011; Mariotti and Fagherazzi, 2013; Bendoni et al., 2016; Francalanci et al., 2013) but in some areas, as outlined above, marginal erosion of marshes is a cyclic phenomenon. Within estuaries, when subtidal channel positions are fixed, increasing accretion of the marsh platform results in steepening of the tidal flat-marsh profile unless the wetland can move landward and the overall profile widens. Steepening of the profile spatially concentrates wave attack and increases the likelihood of edge erosion (Bouma et al., 2014; Kirwan et al., 2016).

Sediment supply to tidal flat-wetland systems is rarely constant. Storms mobilize sediment and provide for high water levels that enable sediment deposition over wide areas high in the tidal frame (e.g., Cahoon et al., 1995b; Yang et al., 2003; Bartholdy et al., 2004; Kim et al., 2011; Schuerch et al., 2012; Tweel and Turner, 2012). Natural cycles and larger scale processes can also influence the status of marshes and mangroves. The 10–40 km long shore-attached mudbanks that move along the coast of French Guiana at rates averaging 1.5 km/yr (Wells and Coleman, 1981) are a good example. Wave damping by offshore unconsolidated mud banks favours mangrove colonization even on the open coast. As fluid mud is moved onto the tidal flat by coastal set up and flood tide (Allison and Lee, 2004), the existing mangrove stand spreads shoreward (Gensac et al., 2011). Pioneer colonization also occurs, often facilitated by desiccation cracking on the upper intertidal (Gedan et al., 2011). However, as mudbank migration continues, wave attack increases leading to erosion of the mangroves during the interbank period.

Sediment supply can be limited within estuaries by human interventions including upstream dam impacts (Yang et al., 2006), land reclamation (van Maren et al., 2016) and channel deepening (Kerner, 2007). However, in some estuaries, dredging has increased suspended sediment concentration (van Maren et al., 2015) and promoted heightening and steepening of intertidal flats (de Vet et al., 2017) which has led to the development of new marshes on previously unvegetated flats in the Western Scheldt. Such anthropogenic influences on sediment supply and distribution can be indirect consequences of interventions with very different aims. There are currently few examples of deliberate interventions to enhance sediment supply to tidal flat-wetland systems at a large scale (see examples discussed later in this paper). Thus, while in many cases coastal wetlands can survive at least moderate rates of sea-level rise, especially where migration inland is possible (Kirwan et al., 2016), on longer time scales the existence of marshes with limited sediment supply is threatened.

In addition to sea-level rise, climate change can influence on the presence of different species of halophytes (see Adam (1990) for review of the role of climate) including the separation of salt marsh and mangrove species based on winter climate tolerance. Warmer winter temperatures that lead to reductions in the intensity of freeze events could result in a shift from marsh vegetation to mangrove forests. Osland et al. (2013) demonstrate the potential for substantial poleward shift in mangroves at the expense of marshes along the north Gulf of Mexico with a modest change in winter freezes, and Raabe et al. (2012) document that in Tampa Bay marsh-to-mangrove ratio has changed from 86:14 to 25:75 since the 1870s. However, Saintilan and Williams (1999) document more complex patterns of landward migration of mangroves into salt marsh areas. Other factors such as local changes in nutrient level or propagule dispersal may be involved. In general, climate exerts overall control on large-scale distributions, but interaction between multiple physical factors within marsh and mangrove systems influence specific vegetation distribution patterns (e.g., Woodroffe, 1982; Kim et al., 2010).

## 4. Contributions to flood risk reduction

Tidal flat-wetland systems can mitigate flood risk by several mechanisms. Due to bathymetric influences and friction, they can alter surge propagation, attenuate waves and reduce current velocity. Emergent canopy wetlands limit the transfer of wind momentum to the water column (Wamsley et al., 2010). Attenuation of waves by tidal flat-wetland systems can potentially reduce:

- direct wave attack on otherwise unprotected coastal infrastructure, limiting damages or reducing the need for armoring or reinforcement
- wave run-up on levees or other protection structures, limiting overtopping that can both directly reduce flooding and the need to armor the dry-side of structures
- erosion of earthen levees at the landward side of the flat-wetland system, increasing their reliability during events or the need for armoring on the wet-side

While these can all occur during storm events, the protection of levees from wave erosion is also important during lower magnitude events, e.g., high tides or moderate storms, when wave attack on structures would otherwise need to be mitigated. Alternatively, wrack generated from adjacent wetlands during storms and stranded on grass covered levees, could result in die-back of protective vegetation cover (e.g., Valiela and Reitsma, 1995) on the levees unless specific management actions are taken (US Army Corps of Engineers, 2014).

### *4.1. Reduction of surge height during storms*

Reduction of surge height has been studied using hydrodynamics theory, field observations and numerical modeling. The height of storm surges is a complex function of bathymetry, duration of persistent winds, propagation speed and angle of the storm, presence of vegetation, and other factors (Resio and Westerink, 2008). Although effects of coastal vegetation on reducing storm surges have been documented in historic cases (described below), the effects vary and cannot be reduced to a single 'reduction factor' of storm surges by coastal wetlands.

There have been few direct observations of surge attenuation across wetland dominated coasts. Williams et al. (2007) provide anecdotal evidence of mangroves near Cairns protecting local infrastructure during Cyclone Larry. Wamsley et al. (2010) analyzed measurements during Hurricane Rita in 2005 in Louisiana and Texas by McGee et al. (2006) and found that measured surge attenuation rates varied from 1m per 25 km to 1m per 4 km. A similar range (1m per 6 km to 1m per 23 km) was reported by Krauss et al. (2009) for two hurricanes in Florida. Paquier et al. (2017) measured a downward slope in water surface elevation (i.e., higher seaward and lower landward) across a relatively narrow marsh in Chesapeake Bay during storms and found a strong interaction among wave attenuation, wave setup and water surface slope. In the Western Scheldt estuary, Stark et al. (2015) measured tidal propagation through a marsh for several flood events, including two storm surges. Calculated attenuation rates were up to 1 m per 1.4 km across the marsh platform and 1m per 20 km through the channels; however, the authors note that many of the transects were very short (< 100m).

Due to the difficulty of collecting field data that is in line with the path of the storm and devoid of influence of other features such as roads, exploration of the effect of wetlands on storm surge has largely been restricted to modeling studies. Ferreira et al. (2014) isolated the effects of land cover by using different data sources for cover to drive simulations of surge associated with Hurricane Brett and a number of synthetic storms. They found uncertainty of approximately 7% of the surge value associated with land cover variations tested, but the study did not consider wave effects. Loder et al. (2009) also examined the effects on surge without waves using an idealized experimental model grid within which they simulated changes in bottom friction, elevation and wetland continuity. While Loder et al. did not relate bottom friction directly to specific vegetation types or landscape factors, they found vegetation-induced bottom friction decreased storm surge levels for peak surges < 2m. Effects of wetlands on storm surges were found to depend strongly on the specifics of the storm. This point was reiterated by Wamsley et al. (2010), who simulated storm surge and wave propagation across wetlands and bays in coastal Louisiana. They found surge attenuation rates ranged from 1m per 50 km to 1m per 6 km with the variations due to landscape character (including bathymetry and wetland type), and storm characteristics including size, speed, track and intensity. Zhang et al. (2012), using model simulations, found higher surge attenuation rates for Hurricane Wilma in South Florida mangroves (1m per 5 km to 1m per 2 km) but also identified a strong dependency of attenuation on storm intensity and speed.

In the Western Scheldt, Smolders et al. (2015) used numerical modeling to examine the influence of different wetland configurations on along estuary attenuation of storm tides. They found a larger wetland surface area increased attenuation along the estuary, but the relation was non-linear with a threshold beyond which increasing area did not result in further attenuation.

### 4.2. *Wave attenuation*

For many coasts, moderate magnitude but high frequency storm events produce waves that cause erosion or threaten coastal defenses. Many studies have examined the role of vegetation in contributing drag and attenuating waves. These include detailed small-scale laboratory studies of idealized stems and their properties such as flexibility and structure (Bouma et al., 2005; Feagin et al., 2009; Smith and Anderson, 2014) and field studies through monospecific or diverse vegetation stands (see summary in Horstman et al. (2014) and more recent work by Mullarney et al. (2017); Norris et al. (2017); Foster-Martinez et al. (2018)). These investigations affirmed that attenuation of wind waves by wetland vegetation is related to factors such as stiffness, plant biomass and height. Horstman et al. (2014) found strong positive relationships between volumetric vegetation density and the rate of wave attenuation in mangrove stands. They attributed the energy loss mostly to vegetation drag rather than bottom friction or viscous dissipation, as ‘the attenuation rates were smallest on the bare tidal flats and significantly increased inside the mangrove vegetation’. Bouma et al. (2010) reported essentially the same result for salt marshes. Wave attenuation by two species with very different growth characteristics was explained by a common function of above-ground biomass, which is equivalent to volumetric density.

Wave damping by vegetation has been incorporated into wave models such as SWAN (Suzuki et al., 2012), XBeach (Roelvink et al., 2009), STWAVE (Anderson and Smith, 2015) and MDO (Marsooli et al., 2017). The formulations of Mendez and Losada (2004), refinements of the basic equations of Dalrymple et al. (1984), are commonly used for shorter waves. XBeach adds a compatible formulation based on the orbital velocity that is resolved for infragravity waves in the model. Wave damping by vegetation depends on both hydraulic conditions, such as water depth and height of incoming waves, and vegetation characteristics, such as vegetation height, density, diameter and flexibility. Vegetation character is commonly represented by a drag coefficient used as a calibration parameter in practical applications. van Wesenbeeck et al. (2017a), using SWAN, show that higher waves are dampened much faster than lower waves. Thus, a wide range of incoming wave heights results in a narrow range of wave height after passing through a vegetated stand. In addition, damping strongly depends on the length of the incoming waves as waves with larger periods need longer distance to travel through vegetation for substantial dampening. van Rooijen et al. (2016) used XBeach to consider infragravity waves and non-linear intrawave interactions. Their study shows that coastal vegetation may have a significant effect on reducing coastal wave setup.

Differential wave damping for short and long waves has also been observed in the field (Phan et al., 2014; Horstman et al., 2014), demonstrating that use of a single coefficient for fraction of wave height lost per m of marsh or mangrove may lead to recommendations of too narrow vegetation belts seaward of coastal protection works. Long waves carry most of the incoming wave energy. If these waves are insufficiently attenuated, they will reflect on the sea wall and cause a local peak in wave dissipation. Phan et al. (2014) also stress the interaction between the long waves and the geomorphology of the coastal area. While long waves can move sediment to the interior of a mangrove stand, they may also be a prime factor inhibiting net sedimentation. Constructing a levee too close to the sea can alter the role of long waves in sediment distribution and lead to mangrove loss in the long term.

Until recently, limited observations were available of either marshes or mangroves subjected to high waves moving across deeply inundated wetlands, i.e., in extreme storm conditions. Möller et al. (2014) conducted a flume study simulating storm waves and showed wave dissipation can still reach 20% over a 40m distance even in water depths typically found during storm conditions. Through comparison with a mowed section, they found that 60% of the change was due to vegetation. However, even this large flume experiment did not allow the simulations of wave heights as specified in the design criteria for dikes in The Netherlands (Vuik et al., 2016). These authors complement field studies with a calibrated version of the SWAN model. They found that vegetation dissipates significant fractions of wave energy well before wave breaking starts, shifting the main energy dissipation mechanism from intense and locally-focused breaking to diffuse dissipation over the vegetation. This study identifies two contributions from vegetation to the attenuation of waves: direct attenuation leading to a diffuse spreading of wave energy dissipation, and indirect effects through the maintenance of a gently sloping and relatively high bathymetry that also significantly contributes to wave attenuation. Without the effect of vegetation on the pattern of wave breaking and stabilizing sediment, such bathymetry would not be stable and the wave energy dissipation would be very different. A comparison between unvegetated and vegetated foreshore effects on wave energy during storm conditions, is given in Fig. 2.

The limited effect of vegetation on reducing the height of storm surges reflects the same phenomenon, as storm surges are extremely long waves (order $10^5$–$10^6$ m). Surge interactions with vegetation are similar to the interaction with tidal currents. Drag forces will slow surge propagation down locally and lead to increased height of the surge seaward of the vegetation. However, if the surge is sustained for a long period, vegetation will have little influence on surge height near the coast eventually. Therefore, vegetation can be a significant factor in the evolution of the surge, but any simplification in terms of an attenuation factor becomes approximate at best, and an inadequate basis for risk reduction measures.

### 4.3. *Effects of local topographic features*

Many researchers identify the need for wide stands of vegetation for effective defense (Bao, 2011; Mariotti and Fagherazzi, 2013; Bouma et al., 2014) but few consider the natural dynamics of those systems and how specific features, beyond vegetation, influence their flood defense function. One of the most dynamic parts of the tidal flat-wetland system is the transition from unvegetated to vegetated zones, which can include marsh cliffs and tussocks or hummocky vegetation. Yang et al. (2012) measured waves seaward, within and landward of a tussock of *Spartina alterniflora* on a macrotidal tidal flat in China and note that wave height over the tidal flat on the landward side of the marsh tussock tended to be lower than that on the seaward side. However, wave height landward of the tussock was greater than that recorded over the marsh tussock itself. At a larger scale, Yang and Irish (2017) conducted laboratory studies of marsh mounds, dynamically similar to those constructed near Snake Island in Galveston Bay, Texas (https://galvbay.org/how-we-protect-the-bay/on-the-ground/snake-island-restoration-project/). They found complex interactions among mound spacing and water depth influenced wave height, with closer mounds and shallower depths producing a greater overall reduction in wave height. They also found that mound-channel bathymetry is a more important factor in

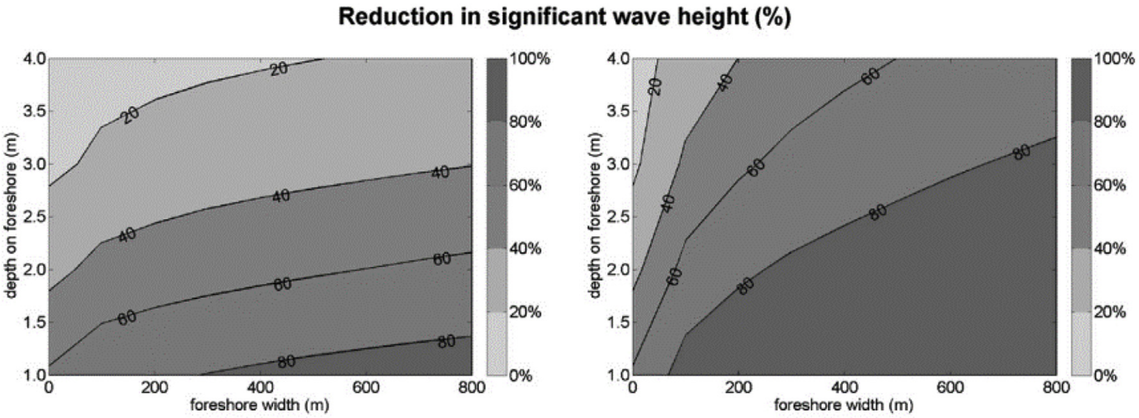

**Fig. 2.** Reduction in significant wave height along a foreshore transect of varying width, without (left) and with (right) vegetation modeled using SWAN. Severe storm conditions were simulated. Parameters for the vegetation were calibrated on Spartina vegetation in winter (from Vuik et al., 2016).

reducing wave height than vegetation. However, without vegetation the small-sized mounds in this model study would probably not be morphologically stable, and thus there is an indirect effect of vegetation via sediment stabilization. The mound-channel bathymetry used in the Yang and Irish experimental study of wave height is analogous to the role of wetland complexity that Loder et al. (2009) and Barbier et al. (2013) found as an important influence on storm surge.

Shore parallel variations or within wetland features, such as creeks and drainage channels, surface pans and local within-system topography (Fig. 4), can also influence flood defense function. The relationship between marsh channels and marsh platform areas influencing storm tide attenuation within a marsh was explored by Stark et al. (2016) using field measurements in the Western Scheldt. They found that maximum attenuation occurred along narrow channel transects with wide marsh platforms, with lower attenuation rates along wider channels with smaller marsh platforms. In Essex, UK, Möller and Spencer (2002) measured changes in wave height across both cliffed and ramped profiles along the same shoreline. They found average wave height increased immediately seaward of a 1.5m cliff but that marsh edge wave energy dissipation is twice as high at the cliffed site than at the smoother ramped transition site. They attribute this change to interaction among wave energy reflection by the cliff face, wave shoaling (i.e., an increase in wave height due to a sudden decrease in water depths), and dissipation due to surface roughness. The effect of the cliff morphology on wave attenuation also dominated seasonal changes in vegetation characteristics.

The configuration and vertical dimension of transitions in water depth and roughness associated with creeks, vegetation changes, surface features and the flat-wetland transition zone need to be considered in site specific evaluation of natural defense functions. The character, and thus the influence on flood defense function, can change over time due to external forcing such as sediment supply and sea-level rise or as a result of interventions designed to enhance or sustain natural defenses.

## 5. Typology of tidal flat-wetland system

To provide a framework for thinking about how changes in tidal flat-wetland character, beyond the details of vegetation type and structure, influence their flood defense function three morphodynamic types are characterized (Fig. 3). Type A represents a profile where vegetation is gradually extending over the gently sloping flat with no distinct topographic margin, although clumps of colonizing vegetation will be associated with local increases in topography. Both flats and vegetated marsh areas are increasing in elevation with adequate sediment supply that enables accumulation of sediment in both vegetated and unvegetated zones. Over time, the extent of vegetation cover increases but the character of the transition zone remains consistent on a prograding coast. Type B characterizes conditions where cliffs develop at the seaward edge of the marsh, with sediment from collapsed blocks of consolidated marsh being retained in the upper bare flats and providing a foundation for vegetation colonization and renewed progradation. The elevation of the tidal flat as a whole remains relatively stable outside of the transition zone. In the marsh, elevation increases due to both sediment deposition and organic accumulation, maintaining a steep gradient between marsh and tidal flat that enables the initiation of the cliff erosion/vegetation colonization cycle (Koppel et al., 2005). For Type B, there are cyclic changes in the position and form of the seaward marsh margin over time. For Type C, the landward margin of the marsh is characterized by a steep eroding cliff and eroded material is not retained in the upper intertidal. The tidal flat is erosional or at least not increasing in elevation resulting in the positive feedback of increased fetch, depth and marsh retreat noted in many modeling

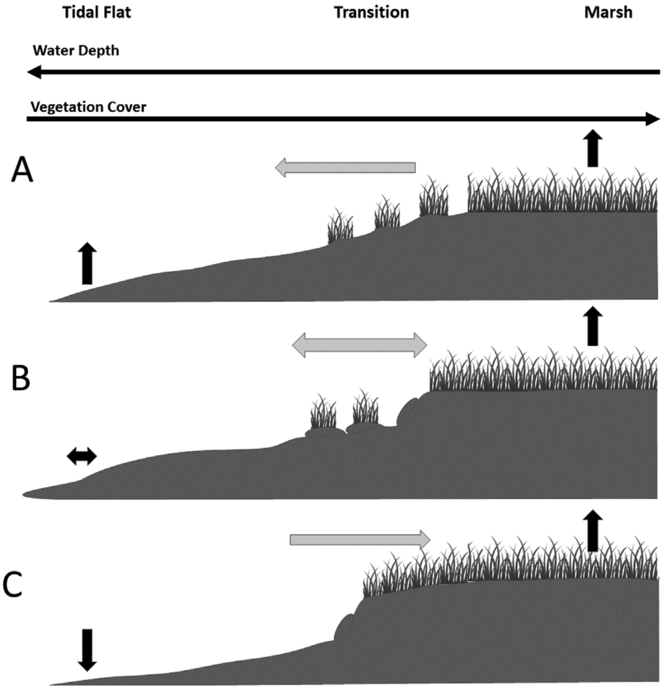

**Fig. 3.** Typology of tidal flat –wetland systems that reflects varied geomorphic contexts. A. Prograding marsh and accreting tidal flats. B. Marsh cliff with rejuvenation and dynamic tidal flat, C. Retreating marsh and eroding tidal flat. Elevation ranges and slopes are idealized and will vary according to tidal range and width available. Dark arrows indicate accretionary status of wetlands and light arrows indicate lateral growth or retreat.

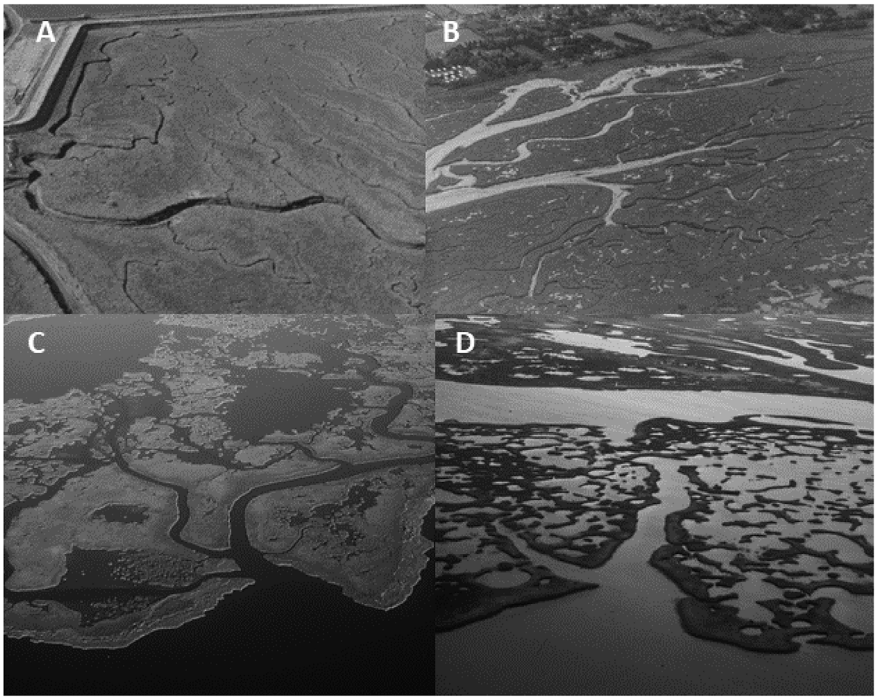

**Fig. 4.** Examples of complex planform pattern in marsh platforms. A. Dengie Peninsula, Essex UK. B. North Norfolk, USA. C. Plaquemines Parish LA USA. D. St Bernard Parish LA USA.

studies (e.g., Mariotti and Fagherazzi, 2013). In this instance the marsh retreats, potentially resulting in 'coastal squeeze' if there is sufficient ability for onshore migration at the landward margin.

## 6. Interventions

The flood defense function of each of the marsh profiles described above depends upon the specifics of morphology and vegetation types. Their current level of functionality depends upon bathymetry, vegetation, and elevation in the tidal frame. Future functionality will also be influenced by 1) sufficient sediment supply to maintain relative elevation for all profiles under sea-level rise, and 2) space to either prograde seaward (Type A) or migrate landward (Type C). Management actions that seek to maintain or enhance the flood defense function of tidal flat-wetland systems must consider these geomorphic contextual factors as well as vegetative structure.

For each of the profile types, Table 1 outlines the factors that can limit system effectiveness as flood defenses both under current conditions and in the future when they are subject to increased rates of sea-level rise. It is possible that marshes may transit from one type to another as sediment supply limits progradation or tidal flat slope, or wave climate changes. However, at any stage and with at least conceptual predictions of how the system will change in the future., the potential interventions identified in Table 1 could be used to sustain the morphological characteristics of the types.

The interventions identified in Table 1 fall into four major categories: creation of new marsh platform (usually using dredged material), enhance/increase sediment supply, limiting erosion/retaining existing sediment, and increasing width available for migration through managed realignment. Previous applications of these approaches can

**Table 1**
Limiting factors and potential interventions for current and future conditions for the three types of tidal flat-wetland systems.

| | Timeframe | Limiting Factors for Maintaining Flood Defense | Potential Interventions |
|---|---|---|---|
| Type A - Prograding marsh and accreting tidal flats | Current | Elevation in the tidal frame | Maintain net sediment supply at current rates |
| | Future | Continued/increased sediment supply for marsh/flat accretion | Maintain or increase sediment supply to levels needed to compensate for sea level rise |
| | | Assuming progradation continues an effective width can be maintained | Ensure tidal flat width/slope is kept available for progradation |
| Type B - Marsh cliff with rejuvenation and dynamic tidal flat | Current | Elevation in the tidal frame | Maintain net sediment supply at current rates |
| | | Maintenance of eroded sediment in transition zone | None (system is in apparent cyclic equilibrium) |
| | Future | Continued/increased sediment supply for marsh accretion | Maintain or increase sediment supply |
| | | Landward migration space to ensure effective width | Managed realignment |
| | | Maintenance of eroded sediment in transition zone | Limit wave energy at seaward marsh margin to current levels |
| Type C - Retreating marsh and eroding tidal flat | Current | Elevation of the system in the tidal frame | Retain sediment on the intertidal to re-establish morphological equilibrium |
| | | Maintenance of eroded sediment in transition zone | Limit wave energy at seaward marsh margin |
| | | Landward migration space to maintain width | Managed realignment |
| | Future | Increase/retain tidal flat elevation | Place/retain sediment on intertidal |
| | | Retain minimum marsh width | Construct new marsh substrate or limit wave energy at marsh margin |
| | | Landward migration space | Managed realignment |

provide important lessons learned for enhancement or maintenance.

### 6.1. Marsh platform construction

Coastal wetlands have been created using dredged material for decades and their development has been documented with some studies reporting floral and faunal characteristics similar to adjacent natural marshes (e.g., LaSalle et al., 1991) and others (Moy and Levin, 1991; Craft et al., 1999) finding that the time taken to achieve such equivalence depends on wetland type and hydrology. Studies of soils in newly created coastal wetlands estimate that decades are required for soil biogeochemistry and infaunal communities to develop to the condition of adjacent natural marshes (Edwards and Proffitt, 2003). However, given observed rapid development of vegetation cover and the opportunity to place dredged material at different heights within the tidal frame, such interventions can be used to enhance the flood defense role of coastal wetlands, even if biodiversity development lags. Most studies of created marshes are in sheltered areas but they may be subject to edge erosion in macrotidal and wave-exposed sites.

There are no field studies of wave or surge attenuation by marshes created using dredged material but modeling studies provide insight. Jamaica Bay in New York has suffered dramatic loss of coastal wetlands since 1959 (Hartig et al., 2002). Marshes have been reconstructed with dredged material in the Bay (Messaros et al., 2012). Following Hurricane Sandy, there were renewed calls for restoration of marshes to mitigate storm flooding. Orton et al. (2015) modeled the effect of ‘restoring’ the marshes to their 1897 footprint and bathymetry while leaving all other aspects of bay bathymetry at current conditions. The effects on peak water level were minimal for simulations of the Hurricane Sandy surge and of an historical storm from 1821, suggesting additional restoration may not be effective in mitigating storm flooding. The Louisiana Coastal Master Plan includes use of dredged material to create marshes and Alymov et al. (2017) numerically simulated the effect of increased elevation and altered roughness in created marshes on storm surge and waves reaching flood protection levees. A large planned marsh creation ($>$ 9500 ha with a 2015 cost of over $1.8b) reduced $H_s$ of approximately 2m from an intense hurricane by less than 0.5m. Both studies show limited effectiveness of created marshes and that marsh construction projects need to be carefully designed to contribute to flood defense. These conclusions are consistent with the discussion above on the limited effect of vegetation on high, prolonged storm surges.

### 6.2. Enhancing sediment supply

Unconfined placement of fine mineral sediments (silt and clay) within the intertidal zone on exposed foreshores allows sediment reworking by waves and currents to shape the flat-marsh system (French and Burningham, 2009). Widdows et al. (2006) document high erodibility of sediment in the few days following placement on flats in Essex, UK with surficial biota potentially playing an important role in stabilization. The fate of unconsolidated sediments placed on exposed shorelines is a key uncertainty and the Essex example suggests that exchange between tidal flat and channel may be much faster than between tidal flat and marsh. However, given concerns about the ability of marshes to keep pace with future sea-level rise, it is important to better understand how and when to place sediment to increase net sediment availability for marshes (Schoellhamer, 2011). Bever et al. (2014) used numerical models to test the fate of sediments placed in different areas of San Francisco Bay to determine whether dredged material placements adjacent to existing marshes would result in an increase in deposition rates within the marshes. Their study found that, in some areas of the Bay, natural dispersal from in-Bay placement could be effective in supplying sediment to tidal flats and marshes. The findings were very site specific but illustrate the potential for strategically enhancing sediment supply to maintain current marsh systems.

In Louisiana, efforts to increase sediment supply to maintain marshes include reconnection of sediment supplies from the Mississippi River (Allison and Meselhe, 2010; Allison et al., 2014) and reliance on physical processes within the estuary to transport sediments to marshes. This utilizes sediment size classes (e.g., fine silt and clay) that are transported as suspended load and could not be captured by dredging, Wang et al. (2014) note that, given subsidence and sea-level rise, diversions of $> 1500\,\mathrm{m}^3/\mathrm{s}$ may be needed to achieve substantial wetland benefits. Allison et al. (2017) and Yuill et al. (2016) found that within a basin, currents, waves and incoming sediment size distribution can have an important influence on whether sediments diverted from the river are retained within the receiving basin.

Predicting the fate of mobile sediment within an estuary or on an exposed foreshore is very dependent on local conditions (e.g., tidal amplitude; wave characteristics; existing morphologic features that help capture and retain suspended sediment, etc.). This has important implications for broad conceptualizations of how ‘ecosystem-based defenses’ can be maintained through the manipulation of existing estuarine processes (e.g., Temmerman et al., 2013). Even though sediment supply is a limiting factor for the long-term sustainability of many coastal marshes, enhancing that supply through direct intervention requires a detailed understanding of local process regimes and may only be of benefit in some areas.

### 6.3. Limiting erosion

Erosion of marsh shorelines is common and whether the erosion results in long-term reduction in marsh width (Type C in Fig. 3) or is part of cyclic dynamics at the marsh edge (Type B) depends on whether sediment eroded is retained within the transition zone or the system as a whole. There is extensive literature and practice in preventing erosion of shorelines (e.g., National Research Council, 2007; Gittman et al., 2014; Nordstrom, 2014). For marsh shorelines there has been an increasing emphasis on ‘living shorelines’, a type of estuarine shoreline erosion control that incorporates native vegetation and preserves native habitats (e.g., Davis et al., 2015; O'Donnell, 2016). Palinkas et al. (2017) evaluated the effects of different shoreline interventions on sedimentation in Chesapeake Bay and found breakwaters were effective sediment traps while riprap isolated marshes from tidal flats, thus decreasing sediment deposition in marshes. This supports studies of breakwater effects on tidal flat deposition and marsh erosion in Essex (Pethick and Reed, 1987; Cooper et al., 2001). Thus, hardening of the marsh shoreline with sills can limit erosion but there may be a tradeoff between marsh area and marsh elevation due to effects on sedimentation. Further, the long-term relative elevation of the tidal flat has consequences for wave erosion at the margin (see discussion above regarding cliffs) implying that retaining sediment within the system may be as important as limiting its release from the marsh edge.

In several areas of the world, permeable fences have been used to stop marsh, mangrove and tidal flat erosion. Originally, permeable wooden structures were used for land reclamation in the Dutch and German Waddensea (Bakker et al., 2002). The structures, made of poles with a brushwood filling, reduce wave heights, increase sediment trapping and reduce erosion, without potential adverse effects, such as increasing reflection of waves (Winterwerp et al., 2013). Winterwerp et al. suggest that groins account for morphodynamics and rehabilitation of accreting convex intertidal profiles. These structures can be used in muddy intertidal profiles with either marshes or mangroves. Their use for restoration of eroding mangroves is increasing and has been documented for Indonesia, Vietnam and Surinam (van Wesenbeeck et al., 2015; Schmitt et al., 2013).

### 6.4. Managed realignment

Migration space for coastal wetlands in the face of sea-level rise is an issue of concern where landward margins are hardened – often termed ‘coastal squeeze’ (Pontee, 2013; Torio and Chmura, 2013). Interventions to expand the space available involve realigning coastal defenses and tidal reintroduction into previously drained marshes (French, 2006), both increasing coastal habitat in the near term and enabling landward migration of wetlands (Esteves, 2013). Studies of managed realignment schemes in the UK show variations in the rate of change within the newly opened areas. Rapid sedimentation is often observed (Rotman et al., 2008; Burgess et al., 2016), especially in sheltered areas (French et al., 2000). The breach morphology also develops fast, as does the channelization within the new area (Friess et al., 2014). Colonization by vegetation can be rapid (Mazik et al., 2010) or slow (Brooks et al., 2015), depending on site specific factors. The evolution rate of delivery of ecosystem services, including flood defense, is therefore also variable (Boerema et al., 2016).

Rarely are the estuary-wide effects of the new tidal prism and sediment sink considered. Townend and Pethick (2002) argued that the practice of leaving most of the existing embankment intact, by allowing only a limited breach, expands the tidal prism without allowing the estuarine cross section to adjust, potentially contributing to erosion of adjacent marshes. Implications for the sediment budget have been a major concern in San Francisco Bay where the planned restoration of about 6000 ha of former commercial salt-evaporation ponds to tidal marsh and managed wetlands is underway. Brew and Williams (2010) modeled whether marsh restoration within the ponds would be at the expense of tidal flat habitat and showed a loss of tidal flats in the long-term even without the restoration. Shellenbarger et al. (2013) used a sediment budget approach to show it would take centuries for existing sediment delivery to fill the newly opened area. Thus, In the face of sea-level rise, reintroduction of tides into former salt ponds can support the landward transition of habitats. However, due to the long-term decline in sediment delivery to San Francisco estuary (Jaffe et al., 2007), it is unclear whether overall flood defense functionality can be maintained as this transition occurs.

### 6.5. Intervention vs. natural evolution

The discussion in this paper regarding marsh development and the role of specific features and characteristics in supporting flood defense shows how physical and biological process act in concert to influence morphodynamics and enhance functionality. Not all marshes are the same, or at the same developmental stage, and different types of interventions can be made to increase the near-term and long-term sustainability of the marsh-tidal flat systems. Sediment availability and fate is an overarching concern, with vegetation cover and structure being a response rather than a driver of the effectiveness of the intervention. Most interventions modify natural processes and readjust aspects of marsh dynamics, either altered by human actions or deemed inadequate under future sea-level rise.

Can well-designed interventions succeed in increasing sustainability? and what are the likely implications for flood defenses? The response is obviously site specific; and successful design of interventions requires detailed understanding of biogeomorphic outcomes at the system scale to avoid unintended consequences. However, for success, interventions need to be targeted toward specific outcomes. Designing for flood defense functions like wave attenuation, requiring higher elevation marsh platforms and robust vegetative cover, may reduce other functions, e.g., fisheries habitat. Moreover, dynamic interactions between marshes and tidal flats mean that measures to elevate the marsh relative to the tidal flat, will result in morphological instability, and the system will tend to re-adjust. Heightening of marshes requires widening of the coastal profile in order to avoid over steepening of the profile.

Natural marsh evolution produces complex systems with topographic variation, tidal channels and variations in vegetative cover (Figs. 1 and 4) that support a variety of functions and influence continued marsh development and long-term sustainability. Tidal channels, for example, maintain morphology and dimensions in equilibrium with the tidal prism (Pethick, 1992) and transport sediment to interior marsh areas (French and Stoddart, 1992; Leonard et al., 1995 among others). Interventions, particularly marsh construction in areas of high tidal range, need to ensure appropriate tidal channel development, found to be best accomplished in San Francisco Bay by allowing natural process to develop the network (Callaway et al., 2011). In the Netherlands, artificial drainage networks to stimulate marsh formation increased marsh aging into a homogeneous cover of Sea Couch (*Elytrigia atherica*) (Esselink et al., 2000; Bakker et al., 2002).

While vegetated wetlands are often seen as the ‘nature-based defense’, this paper has shown that, for long-term development as well as short term dynamics, the vegetated marsh should be considered as part of a system with the adjacent tidal flat. Interventions that place sediment on the tidal flat anticipate that this will enhance marsh development, but also need to consider the equilibrium profile of the tidal flat and its interactions with the channels. A more holistic approach to interventions can help keep sediment, even if eroded from the marsh platform, within the system. Larger system consideration, however, introduces additional complexity and likely less certainty regarding the outcome of the intervention and this could be of concern to decision makers.

## 7. Summary and conclusions: planning for natural flood defenses

The decades of studies from across the world demonstrate extensive understanding of the process dynamics of tidal flat-wetland systems. These processes manifest in different coastal settings to produce different morphologies. For marsh environments, the variation can be characterized by three types of cross profile (Fig. 3). Planform complexity is less readily summarized but the development and dynamics of key spatial features are sufficiently understood to enable site specific assessment of their current and future role in flood defense. Coastal managers and planners must recognize that tidal flat-wetland systems are neither homogeneous nor static in character. This is even more important given the common simplifying assumptions of homogeneity in many modeling studies of nature-based flood defenses.

The important role of sediment supply in determining the current typology of tidal flat-wetland systems and their future character under accelerated sea-level rise (in many areas exacerbated by subsidence) requires that wetlands not be seen in isolation of their coastal setting. Broader estuarine or coastal setting influences sediment availability. Dredged channels which become sediment sinks and armored shorelines that prevent sediment release are just two of the common human influences on coastal sediment supply. In the context of flood defense, the coastal setting also determines the hazard, and thus the potential effectiveness of the tidal flat-wetland system. Planners and managers need to be cognizant that local tidal flat-wetland systems may be effective defense against only some types of hazard – a concept which also applies to traditional flood defense systems when designed to protect against a specific ‘standard’ event or return interval. This systems context is vital and this review demonstrates that understanding of the physical environment is an important first step in consideration of the flood defense role of tidal flat-wetland systems.

It is challenging to predict the specific character of tidal flat-wetland system decades into the future. However, using understanding of their dynamics and plausible change in key external factors such as sea-level rise, a range of potential future conditions can be estimated. Numerical modeling can be used to identify the range of flood defense outcomes and their sensitivity to uncertain factors, such as storm damage. Sensitivity analysis to future scenarios of climate, sea level and sediment availability is equally important. The importance of local effects requires tailor-made and site-specific intervention plans.

Planning for natural flood defense should not be held to a higher standard than traditional approaches. There are reported examples of underperformance or even failure (e.g., 1953 North Sea surge, Hurricane Katrina in 2005, Storm Xynthia in 2010) from traditional risk reduction structures, with high maintenance costs that will only increase as sea-level rises. Yet traditional approaches are seen by many as more reliable and effective than natural systems. The practice of their design is also well established. While there are few coastal hazards where tidal flat-wetland systems can eliminate all risk, there are likely many where they can make a meaningful contribution. Where these possibilities exist, application of existing knowledge of their morphodynamics, combined with detailed characterization of the hazard, makes it possible to bound their incremental contribution to risk reduction. Just as understanding their potential role requires a more holistic consideration of tidal flats and wetlands as systems, tailoring interventions to enhance or sustain their flood defense role takes a holistic approach to integrate them with other flood defense features.

## Declarations of interest

None.

## Contributions

DR, BvW, PH and EM all participated in the conceptualization; DR led the development of the manuscript; BvW, PH and EM all provided detailed comments on drafts and contributed to revisions.

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