

## Identifying knowledge gaps hampering application of intertidal habitats in coastal protection: Opportunities & steps to take



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### ARTICLE INFO

#### Article history:

Received 20 June 2013

Received in revised form 20 November 2013

Accepted 22 November 2013

Available online 28 December 2013

#### Keywords:

Coastal protection  
Intertidal ecosystems  
Ecology  
Salt marshes  
Biogenic reefs  
Seagrass

### ABSTRACT

Over the last decades, population densities in coastal areas have strongly increased. At the same time, many intertidal coastal ecosystems that provide valuable services in terms of coastal protection have greatly degraded. As a result, coastal defense has become increasingly dependent on man-made engineering solutions. Ongoing climate change processes such as sea-level rise and increased storminess, require a rethinking of current coastal defense practices including the development of innovative and cost-effective ways to protect coastlines. Integrating intertidal coastal ecosystems within coastal defense schemes offers a promising way forward. In this perspective, we specifically aim to (1) provide insight in the conditions under which ecosystems may be valuable for coastal protection, (2) discuss which might be the most promising intertidal ecosystems for this task and (3) identify knowledge gaps that currently hamper application and hence need attention from the scientific community. Ecosystems can contribute most to coastal protection by wave attenuation in areas with relatively small tidal amplitudes, and/or where intertidal areas are wide. The main knowledge gap hampering application of intertidal ecosystems within coastal defense schemes is lack in ability to account quantitatively for long-term ecosystem dynamics. Such knowledge is essential, as this will determine both the predictability and reliability of their coastal defense function. Solutions integrating intertidal ecosystems in coastal defense schemes offer promising opportunities in some situations, but require better mechanistic understanding of ecosystem dynamics in space and time to enable successful large-scale application.

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## 1. Introduction

### 1.1. Coastal ecosystems for coastal defense

Over the last few decades, the majority of the world population has settled in coastal areas, a trend that is expected to continue in the future (Small and Nicholls, 2003). This global trend has caused increasing anthropogenic activities in coastal areas, with both direct (e.g. coastal engineering) and indirect effects (e.g. land cover change) on coastal ecosystems (e.g., Cohen, 2003; Mora, 2008). As a result, the extent and health of many coastal ecosystems has declined (e.g., seagrasses,

Waycott et al., 2009; salt marshes, Adam, 2002; Boorman, 1999; coral reefs, Mumby et al., 2006, 2007; mangrove forests, Valiela et al., 2001). With the decline of these ecosystems, the supporting, provisioning, regulating and/or cultural ecosystem services they provide are also lost (MEA, 2005). One of them is coastal protection by wave attenuation and/or the reduction of flooding risks, which is particularly relevant for the safety in coastal areas (Borsje et al., 2011; Costanza et al., 1997; Koch et al., 2009; Temmerman et al., 2013). Given the combination of increasing storminess (Donat et al., 2011; Young et al., 2011) and accelerating sea-level rise (Donnelly et al., 2004), there is a need to improve coastal defense for the protection of coastal infrastructure and livelihoods. Integrating nature into coastal defense schemes may offer an innovative and cost effective way to achieve this (Borsje et al., 2011; Temmerman et al., 2013). Recent reviews have exemplified this by highlighting the role of coastal wetlands in protecting shore lines (Gedan et al., 2011;

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Shepard et al., 2011). To determine whether a coastal ecosystem may in practice realistically be incorporated in defense schemes, it is required to assess 1) the *coastal defense value* of an ecosystem under relevant storm conditions as may be expected within a specific time-frame and 2) the *long-term persistence* of the ecosystem over a specific time-frame. In this case, the relevant time-frame is the one that is related to the life-time for which a coastal defense structure is designed, without needing major redesigning (i.e., around 50 to 100 years).

In this perspective we do not aim at giving a comprehensive review, but rather aim at specifically pinpointing the most important knowledge gaps that need to be resolved to implement the application of ecosystems in coastal defense schemes. We start with providing an overview which ecosystem properties of intertidal ecosystems are regarded to be most important for *coastal defense values* by wave attenuation and bed stabilization. We then discuss how the *coastal defense value* of an ecosystem may be expected to depend on landscape scale (i.e., tidal and dimensional) settings. We subsequently discuss to which extent we can predict the *long-term persistence* of these ecosystems in the typical highly dynamic coastal environments. Finally, we discuss where the integration of ecology and engineering may be most promising. These considerations reflect the outcome of the authors' joined efforts within the interdisciplinary THESEUS project, which is an EU funded project aimed at developing innovative technologies to create safer European coasts in a changing climate.

## 2. Factors determining the value of coastal ecosystems for coastal defense

Most ecosystems provide a wide range of ecosystem services (MEA, 2005). Especially (intertidal) coastal ecosystems deliver valuable ecosystem services (Costanza et al., 1997), such as providing food, shelter and nursery areas for numerous species, including commercially important fish (e.g. Nagelkerken, 2000; Valentine and Heck, 1999) and representing an important carbon sink (e.g., Donat et al., 2011; Fourqurean et al., 2012). An increasingly recognized, yet understudied service provided by coastal ecosystems is their ability to contribute to coastal protection by i) attenuating waves, ii) stabilizing shore lines and iii) reducing flood surge propagation.

### 2.1. Wave attenuation by intertidal coastal ecosystems

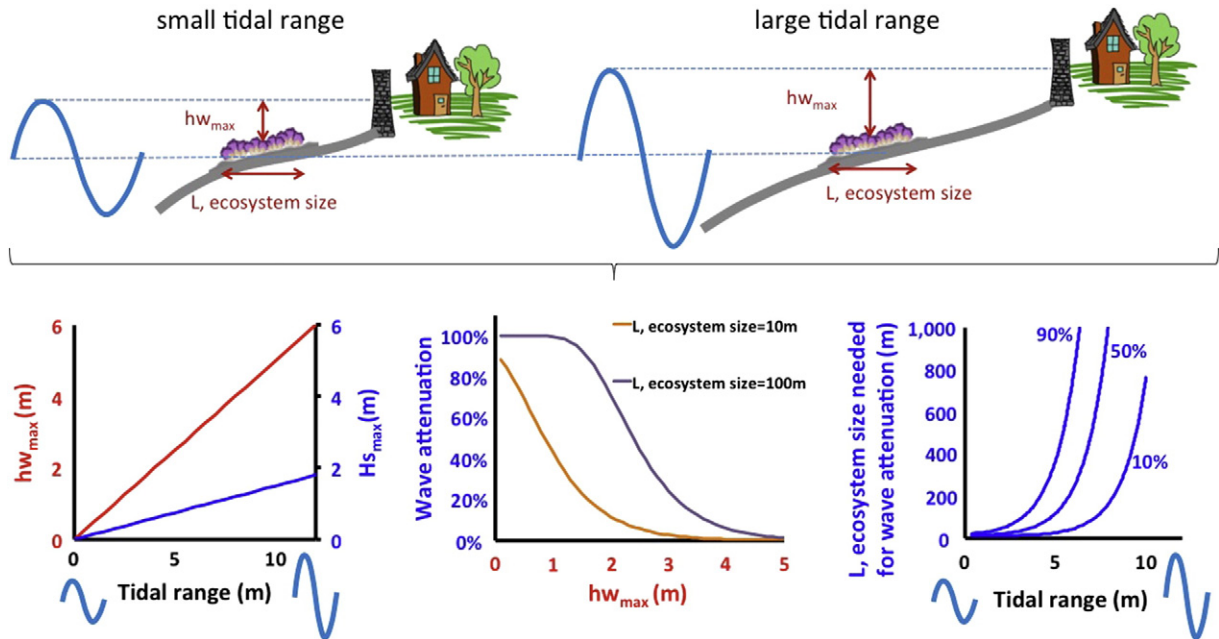
Intertidal coastal ecosystems have a defense value by reducing the wave energy reaching the coastline (Koch et al., 2009). This has perhaps been most clearly demonstrated for salt marshes, which can be flooded either from tidal water movement or from rare storm events (Möller, 2006; Möller et al., 1999, 2011). In marshes, wave-attenuating effect is related to a combination of vegetation characteristics like stiffness (Bouma et al., 2005) and standing biomass (Bouma et al., 2010) and physical factors like inundation height (Möller et al., 2011; Yang et al., 2012; Ysebaert et al., 2011) and the characteristics of the incident waves (Shepard et al., 2011). A recent meta-analysis showed that vegetation density, biomass production and marsh size were most relevant in being positively correlated to both wave attenuation and shoreline stabilization (Shepard et al., 2011).

Next to marshes, wave energy can be attenuated by any intertidal ecosystem that creates aboveground structures of significant size, such as biogenic reefs, seagrass, kelp and mangroves. Several studies have described the effect of mangroves for wave attenuation (e.g., see Aziz et al., 2013; Bao, 2011; Barbier et al., 2008). For wave attenuation by seagrass meadows there is also a substantial body of work done (e.g., see Manca et al., 2012; Maza et al., 2013; Paul et al., 2012; Infantes et al., 2011 and references therein), whereas for wave attenuation by reef-building bivalves such as oysters and mussels (e.g., see Borsje et al., 2011; Donker et al., 2013) there seems to be relative few publications available to date.

Studies on seagrasses point out that their value for coastal protection can strongly depend on the environmental boundary conditions such as water depth and seasonal influences on shoot density (Fonseca and Cahalan, 1992; Paul and Amos, 2011; Stratigaki et al., 2011). However, the relatively high flexibility of seagrass makes them less effective in attenuating waves than marsh vegetation (Bouma et al., 2005) unless they have a very high biomass (Bouma et al., 2010; Paul and Amos, 2011). Moreover, seagrass shoots easily bend under currents, thereby losing wave-attenuating capacity (Paul et al., 2012), making them less effective in macro-tidal areas with strong tidal currents. The effect of tidal currents on wave attenuation by flexible coastal vegetation, including seagrass and marsh vegetation, remains understudied, and forms an important knowledge gap. In nutrient rich environments, seagrass plants may become more brittle and easily break when exposed to waves (La Nafie et al., 2012). This decreases their wave attenuation capacity and emphasizes the importance of good water quality management when using ecosystems for coastal defense purposes. The relative lack of knowledge on the indirect effect of water quality on wave attenuation and stability of intertidal ecosystems, by affecting vegetation development, is another important knowledge gap.

Biogenic reefs in temperate climate zones, as created by e.g. oysters, mussels or honeycomb worms, are usually found below mean sea level (Barbier et al., 2008) and may therefore be less effective for the protection of coastal structures from waves. The exact value will depend on the local tidal amplitude and size of the ecosystem, as explained schematically in Fig. 1. However, due to their rigidity, reefs are efficient breakwaters when compared to flexible vegetation (reviewed in Bosje et al. 2011). Furthermore, similar to the effect of vegetation, their active role in stabilizing the substrate might also be important. This is a currently an under-appreciated service, that affects long-term wave forcing of the coastline (Storlazzi et al., 2011). Compared to work done on coastal vegetation, data is scarce on the wave attenuation by biogenic reefs in temperate areas, deserving further study.

The aforementioned aspects are generalized in a conceptual diagram (Fig. 1) and overview table (Table 1) that both were based on simple calculations to extrapolate experimental data as explained in Box 1. The conceptual diagram and overview table demonstrate that ecosystems occurring high in the intertidal zone (i.e. marshes) will be more effective for wave attenuation than ecosystems that occur lower in the intertidal zone (i.e., biogenic reefs and seagrass meadows), because of the lower maximum flooding depth ( $hw_{max}$ ; Fig. 1). On top of that, tidal range in relation to the elevation where the ecosystem occurs, will affect both the wave-attenuating effect and the point in the tidal cycle where wave attenuation is optimal (by affecting  $hw_{max}$ ; demonstrated for oyster reefs in Fig. 1). Hence, wave attenuation especially of those ecosystems occurring relatively low in the intertidal (Table 1) will be most beneficial in micro and meso-tidal ecosystems, as inundation height ( $hw_{max}$ ) will be relatively small, and the time during which the waves are affected by intertidal ecosystems is the longest (Table 1). Estimating the maximum tidal range at which intertidal habitats can still attenuate 50% of the incident wave height over a length of 50 ( $MT_{50/50}$ ) and 100 ( $MT_{50/100}$ ) m shows that especially those ecosystems located high in the intertidal (i.e. salt marshes) can effectively attenuate waves over a much wider spectrum of tidal ranges than ecosystems located at lower elevations (Table 1). For ecosystems lower in the intertidal, it is mainly the wave decay coefficient ( $k_{habitat}$ ;  $m^{-1}$ ) that determines the tidal range over which waves can be effectively attenuated. The biogenic structures that are situated in the lower intertidal zone should, however, not be fully discarded, as they may at a local scale stabilize the sediment bed and protect ecosystems in the higher intertidal zone from hydrodynamic energy. As these ecosystems higher in the intertidal have a strong attenuating effect on waves, the ecosystems lower in the intertidal may provide an important indirect value for coastal protection (Fig. 2 and see section on ecosystem stability). Such positive interactions via physical processes have since the original paper of Bruno (2000) and Bruno et al. (2003) been intensively studied at the small-scale of the community level (e.g., see Guo



**Fig. 1.** Schematization of the relation between tidal amplitude and the effectiveness of wave attenuation by intertidal habitats, using simplified calculations to extrapolate experimental data as explained in Box 1. Given that all intertidal ecosystems require a typical well-defined flooding period, the maximum flooding depth ( $hw_{\max}$ ) and thereby the maximum wave height ( $Hs_{\max}$ ) that such intertidal ecosystem can encounter will increase with tidal range (lower left panel). The effectiveness of wave attenuation by intertidal habitats decreases with (maximum) flooding depth ( $hw_{\max}$ ) (lower middle panel). As a consequence, the length/size of an intertidal habitat needed to provide a specific defense value in terms of wave attenuation (e.g. 10%, 50% or 90% wave reduction), increases with tidal range (lower right panel). This principle was illustrated, using simplified basic calculations as explained in Box 1.

and Pennings, 2012; Petterson and Bell, 2012) and regarded important for restoration (Crain and Bertness, 2006), but to our knowledge, their importance for coastal defense at the landscape level is still understudied (also see Koch et al., 2009).

Here, it is shown that ecosystems located high in the intertidal effectively attenuate waves over a much wider ambit of tidal ranges than ecosystem located at lower elevations.

## 2.2. Wave attenuation and seasonal biomass loss: the importance of biogeomorphological effects

In the case of mangroves and tropical seagrasses, aboveground vegetation structures are present year round. In temperate climates the majority of the aboveground structure of perennial plants can be partly or completely lost during winter (e.g., Paul and Amos, 2011) and annual plants may even disappear entirely (e.g. *Salicornia* spp.), which will minimize wave attenuation by their structures. As a consequence, in the most extreme case only the body of sediment that was trapped and accumulated by the plant structures during the growing season remains, either mineral or organic. In minerogenic marshes (e.g. most marshes in NW Europe), this trapped sediment can cause a significant increase in elevation (Cahoon et al., 1996, 2006; Callaghan et al., 2010; van de Koppel et al., 2005 and references therein). Even in the absence of a full vegetation cover during winter, the increased elevation of the marsh platform, combined with the high surface roughness from remaining plant parts, can significantly increase wave attenuation (Callaghan et al., 2010). To contribute to coastal defense, it is essential that the accumulated sediment is strongly stabilized, as otherwise the elevated sediment body could easily erode (e.g., see erosion of sediment trapped by *Zostera marina* patches; Bos et al., 2007). Both seagrasses (e.g., Hendriks et al., 2008, 2010) and salt marshes (Van de Koppel et al., 2005) are known to be able to trap sediment and have a sediment stabilizing effect. For tropical systems it has been shown that even heavily grazed seagrass meadows can have a substantial sediment stabilizing effect (Christianen et al., 2013). Recent studies have reinforced the importance of belowground biomass for decreasing sediment erosion rates (Deegan et al., 2012; Silliman et al., 2012) opposing the

conclusion that plants do not directly reduce wetland edge erosion (Feagan et al., 2009). However, relatively little is known about root growth strategies and root turnover in response to environment settings for both seagrass and salt marsh species (Bouma et al., 2001a,b; Kiswara et al., 2009), making this an important knowledge gap to address. The stability gained by persistent root mats deserves to receive much more attention in temperate vegetation, as too many studies only focus on the effect of the aboveground vegetation that disappear during winter.

Overall the current state of understanding makes it clear that in order to contribute to coastal defense by wave attenuation, ecosystems should provide either i) year-round high standing biomass that directly causes wave attenuation, or ii) have a biogeomorphic effect that creates a stable and persistent alteration of the fore-shore bathymetry that enhances wave attenuation.






## 2.3. Ecosystem effects on flood propagation & space

The effects of intertidal ecosystems, like salt marshes and mangroves, on flood propagation are less studied than wave attenuation, but can also be important, provided that the ecosystems cover a large area (Temmerman et al., 2012; Zhang et al., 2012). This is also true for wave attenuation, for which extensive ecosystem dimensions are important (Barbier et al., 2008; Koch et al., 2009). Areal extent may be one of the most important constraints related to the use of coastal ecosystems for coastal defense. As illustrated in Fig. 1, the space needed to attenuate waves will strongly increase with greater tidal ranges (for details on calculations and underlying assumptions, see Box 1). In systems with a large tidal range, waves will be less attenuated due to the larger maximum flooding depth ( $hw_{\max}$ ; Fig. 1) than in systems with a small tidal range. Hence, larger cross-shore habitat dimensions are needed to compensate for this loss in wave reduction efficiency per unit length (Fig. 1).

To obtain validated quantitative insight in the relation between tidal range and the space needed for effective coastal defense services, we are in need of smart monitoring programs at the European scale that cover areas of different tidal amplitudes. Given the relatively few remaining coastal wetlands of significant size, plus the (fortunately) rare occurrence of extreme storms causing flooding threats, it is extremely

**Table 1**

Conceptual illustration on how the coastal defense value of a number of intertidal ecosystems depends on the tidal range of the area where they occur. To elucidate the importance of tidal range for the effectiveness of various ecosystems in reducing wave height, we estimated for each ecosystem the maximum tidal range at which they can still attenuate 50% of the incident wave height over a length of 50 ( $MT_{50/50}$ ) and 100 ( $MT_{50/100}$ ) m, using simplified calculations with wave decay coefficients ( $k_{habitat}$ ;  $m^{-1}$ ). The decay coefficient for wave attenuation ( $k_{habitat}$ ;  $m^{-1}$ ) represents maximum values based on measurements where the water height equals the height of the vegetation/reef, which can then be used in calculations for different water heights (details explained in Box 1). For an in-depth review of the wave attenuation by salt marshes the reader is referred to the recent review papers by Gedan et al. (2011) and Shepard et al. (2011); for wave attenuation by seagrasses to the paper of Ondiviola et al. (2014). It is noted that most papers express wave attenuation as a relative decrease in wave height per meter, rather than by an integrative parameter like the decay constant as used in our calculations (Box 1).

Intertidal ecosystems	Habitat characteristics		Coastal protection service			
	Wave exposure	Height in intertidal frame as submersion period (typical % [range %])	Sediment stabilization	Wave attenuation		Maximum tidal range (m) reducing 50% wave height over 50 ( $MT_{50/50}$ ) and 100 ( $MT_{50/100}$ ) m ecosystem
				Wave decay coefficient ( $k_{habitat}$ ; $m^{-1}$ )	Does seasonality affects wave attenuation?	
 Salt marshes	Sheltered	5% [ $<30\%$ ]	Binding by roots & rhizomes <sup>a</sup> ; reduction of currents	0.01–0.05 <sup>b-d</sup>	Yes, due to loss of aboveground biomass in winter <sup>e</sup>	Marshes are always effective for any realistic tidal range (i.e. $MT_{50/50} = 22.6$ ; $MT_{50/100} = 33.2$ )
 Seagrass meadows	Moderate exposed	45% [ $>30\%$ ]	Binding by roots & rhizomes <sup>g</sup> ; reduction of currents <sup>h</sup>	0.001–0.01 <sup>i,j</sup>	Yes, due to loss of aboveground biomass in winter <sup>e</sup>	$MT_{50/50} - OMT_{50/100} = 0.7$
 Mussel beds	Moderate exposed	63% [ $>45\%$ ]	Sediment covering; reduction of currents <sup>m</sup>	0.05–0.15 <sup>m,k</sup>	No	$MT_{50/50} = 1.8-3.2$ $MT_{50/100} = 2.7-4.2$
 Oyster reefs	Exposed	75% [ $>55\%$ ]	Reduction of currents <sup>e</sup>	0.15–0.30 <sup>m</sup>	No	$MT_{50/50} = 2.8-3.5$ $MT_{50/100} = 3.5-4.3$
 Sabellaria reefs	Exposed	95% [ $>75\%$ ]	Sediment binding <sup>l</sup> ; reduction of currents <sup>l</sup>	no data	No	Unknown

<sup>a</sup> Cahoon et al., 1996.

<sup>b</sup> Möller et al., 1999.

<sup>c</sup> Yang et al., 2012.

<sup>d</sup> Ysebaert et al., 2011.

<sup>e</sup> Koch et al., 2009.

<sup>f</sup> Bos et al., 2007.

<sup>g</sup> Christianen et al., 2013.

<sup>h</sup> Hendriks et al., 2008.

<sup>i</sup> Fonseca and Cahalan, 1992.

<sup>j</sup> Paul et al., 2012.

<sup>k</sup> Donker et al., 2013.

<sup>l</sup> Gruet, 1986.

<sup>m</sup> Borsje et al., 2011.

difficult to obtain field measurements under relevant conditions. Hence, modeling provides an important tool to improve our insights into these aspects, provided that we have ways to validate them. Hence, we recommend investing in fully self-contained and automated monitoring systems that will obtain long-term field measurements (including rare extreme events) as needed for model validation.

### 3. Factors determining the long-term persistence of coastal ecosystems with defense value

As coastal defense schemes are designed with a decadal life span, incorporating coastal ecosystems in such coastal defense schemes requires these ecosystems to remain stable for a similar decadal period. This requires in-depth understanding and predictive ability of the long-term persistence of those ecosystems. As coastal environments are typically highly dynamic, the ecosystems inhabiting these areas can also be dynamic with periods of range extension and contraction. These phases may result from inherent ecosystem dynamics (e.g. van de Koppel et al., 2005), or from changes in environmental setting. The

latter can be driven by inherently dynamic nature of some tidal channels (Pringle, 1995) or local anthropogenic change, for example deepening of estuarine channels for navigation, or from global change such as sea-level rise (IPCC, 2007) or enhanced storminess (Donat et al., 2011; Young et al., 2011). All these factors create uncertainty on the long-term persistence of ecosystems, which may be further enhanced by geomorphic processes (Balke et al., 2012; French, 2006). We speculate that time-scale and uncertainty increase from ecological processes to anthropogenic processes to global change processes (see conceptual diagram in Fig. 3 (left panel)). We propose that the most robust compensation measure for long-term uncertainties is to provide sufficient space for coastal ecosystems to enable them to adjust to changing conditions. In addition, the uncertainty also creates the need for long-term monitoring (Fig. 3 (right panel)).

#### 3.1. Effects of sea-level rise on long-term ecosystem persistence

Unless ecosystems have the space to adjust their location or elevation in the intertidal zone to the sea-level rise, they will be stressed by

**Box 1**

A minimal method to estimate wave attenuation by intertidal habitats in environments with different tidal amplitudes (calculation background for Fig. 1).

Intertidal habitats can serve in coastal protection by attenuating waves before they reach the shore or seawall. Here we illustrate a strongly simplified calculation based on measurements to roughly estimate the dimensions of intertidal habitats which are needed to have in order to significantly contribute to wave attenuation in environments with different tidal amplitudes, as used to generate Fig. 1.

In this calculation we assume that (i) when winds blow onshore, wave conditions will reach maximum waves over the wetland given the depth limitation of wave height, which need not be true if waves are fetch limited e.g. in estuarine settings, and (ii) waves approach in an onshore direction during extreme conditions, which also need not be true for particular locations. We ignored complicating factors such as shoaling as may occur over cliffed marsh margins (Möller and Spencer, 2002) and bed-interaction such as viscous damping of muddy foreshores. Decrease in wave height, when attenuated by an intertidal ecosystem, is assumed to roughly follow an exponential decay, with the main modifiers the cross-shore length of the habitat over which the waves attenuate and the water depth which determines how much friction the waves experience from the intertidal habitat (cf. Yang et al., 2012; Ysebaert et al., 2011):

$$H_L = H_0 * \exp(-K * L)$$

where  $H_0$  (m) is the incoming wave height,  $H_L$  (m) is the wave height after attenuation by the intertidal ecosystem,  $L$  (m) is the cross-shore length of the habitat and  $K$  is the decay coefficient. In shallow waters, waves ( $H$ ) are depth ( $h$ ) limited, so that the maximal significant incoming wave height ( $H_s$ ) can be roughly derived from the inundation depth (Callaghan et al., 2010; Ysebaert et al., 2011):

$$H_0 = 0.3 * h.$$

The habitat specific decay coefficient depends on the friction that the waves experience from the intertidal habitat, as determined by the density, stiffness and complexity of the structures (Bouma et al., 2005, 2010; Mariotti and Fagherazzi, 2010; Ysebaert et al., 2011). The decay constant ( $K$ ) for a specific habitat type with a specific coverage of above seabed structure for a given water depth can be approximated by:

$$K = k_{\text{habitat}} * B / B_{\text{max}} * \exp(-d * h)$$

where,  $k_{\text{habitat}}$  is the habitat dependent decay constant as obtained from experimental measurements (see Table 1 for typical value ranges of intertidal habitats),  $B$  is the % coverage and  $B_{\text{max}}$  is % maximum coverage of the biota along the cross-shore length ( $L$ ) of the habitat, and  $d$  is a decay coefficient for the loss of friction with water depth.

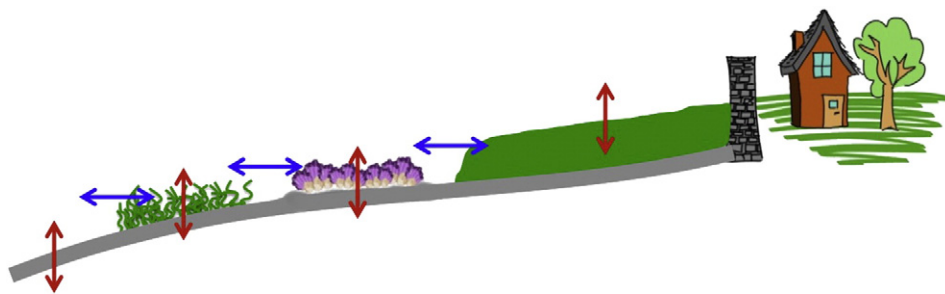
In our calculations for Fig. 1, we used the oysters to illustrate how intertidal habitat can play an important role for wave attenuation and how the tidal settings determine the functioning. For this calculation, we estimated  $d$  based on measurements in salt-marshes ( $d = 1.5$ , Ysebaert et al. 2012) and assumed maximum coverage with shellfish ( $B = B_{\text{max}}$ ) and substituted these parameters for oysters gives:

$$K = k_{\text{shellfish}} * \exp(-1.5 * h).$$

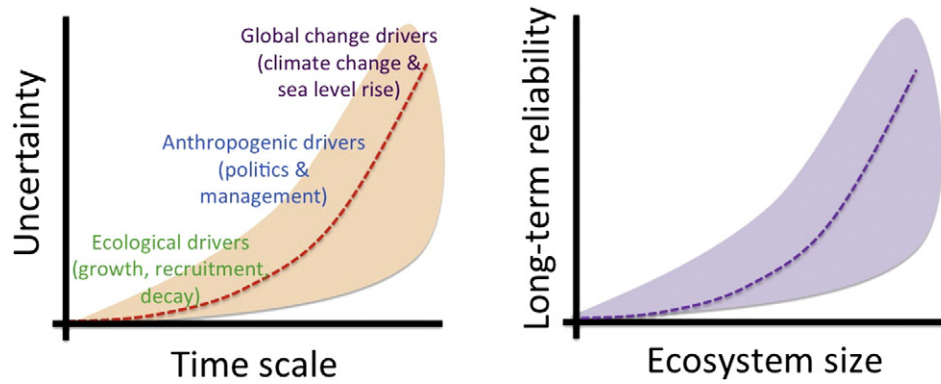
Using this calculation method, the maximum tidal range at which intertidal habitats can still attenuate 50% of the incident wave height over a length of 50 ( $MT_{50/100}$ ) and 100 ( $MT_{100/100}$ ) m is derived (Table 1). This analysis shows that ecosystems located high in the intertidal (i.e., salt marshes) can effectively attenuate waves over a much wider spectrum of tidal ranges than ecosystems located at lower elevations (Table 1). It is, however, noted that the above estimation method can only be used for shallow to intermediate sloped foreshores (i.e., non-reflective or dissipative foreshores), where wave attenuation due to friction by the intertidal ecosystem is dominant. At steeper foreshores wave breaking makes this basic approach unsuitable.

changed inundation periods. This will affect not only ecosystem health and stability, but also their wave-attenuating capacity (cf. Fig. 1). The potential impact of sea-level rise on ecosystem health will differ across intertidal ecosystems, depending on if the ecosystem is stressed by emergence (i.e., 'aquatic' ecosystems) versus stressed by submergence

(i.e., 'terrestrial' ecosystems). In the first group, e.g. biogenic reefs and seagrasses, sea-level rise may decrease emergence stress as caused by e.g. desiccation or too short feeding time. However, as most intertidal species also have a lower depth limit for their physiological functioning (e.g., light availability for seagrass) or an optimization between feeding



**Fig. 2.** Schematic representation of how sub- and intertidal habitats are connected, and can facilitate each other directly and indirectly. Above ground structures, such as formed by e.g. seagrass canopies or shellfish reefs, can directly facilitate ecosystems higher up in the intertidal, by attenuating wave energy (blue arrows). As most of these sub- and intertidal habitats affect sediment stability and sediment accumulation (red arrows), they alter the bathymetry, and thereby indirectly the hydrodynamics. Whether such effect is facilitative or negative to organisms higher in the intertidal depends on whether the sediment is stabilized or destabilized, and on the habitat requirements of the neighboring organism.



**Fig. 3.** Conceptual diagram for the relation between the planned lifespan of a coastal defense scheme involving intertidal habitats, and uncertainty as affected by various factors. Uncertainty increases exponentially over time, due to the fact that ecosystem stability depends on many factors at different temporal scales (*left panel*). On short time scales, ecological drivers might be the most important cause of uncertainty, making things ‘relatively easy’ to predict. Over longer time scales, anthropogenic drivers might become increasingly important, causing larger uncertainties, as political views and economic impacts on coastal management practices may change unpredictably. Over the very long time scales uncertainties in climate change and sea-level rise may add an extra factor to the ecological and anthropogenic uncertainties. The best buffer against these increasing uncertainties over time is a large size of the ecosystem (*right panel*), to provide ample space for wave attenuation and to allow for ecosystem dynamics.

time and predation chance (e.g., enhanced feeding by mussel beds vs. enhanced risk of predation by starfish), a shift to deeper waters may also pose problems for such ecosystems.

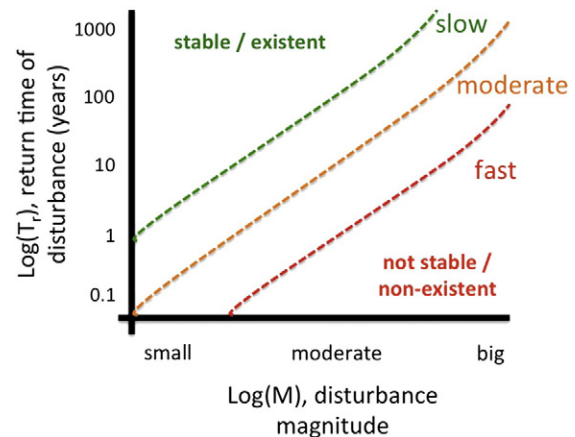
For ecosystems stressed by submergence, e.g. mangrove and salt marsh vegetation, the threats of sea-level rise are more evident (Gedan et al., 2011). Intertidal wetlands like salt marshes can drown, unless they can migrate landward or follow sea-level rise by sediment or peat accretion (Kirwan and Temmerman, 2009). The presence of a fixed landward boundary (e.g. dikes) may prevent intertidal wetlands to migrate upward, resulting in wetland loss with sea-level rise, often referred to as ‘coastal squeeze’ (Wolters et al., 2005). In areas where there is sufficient sediment available, it is expected that salt marshes will accrete sediment at the same rate as sea level (Kirwan and Guntenspergen, 2010; Kirwan and Temmerman, 2009). It is noted that, when determining if a marsh will in the future be drowning or not, it is important to account both for sea-level rise and the often ignored shallow subsidence processes (Webb et al., 2013). Shallow subsidence is a highly important problem occurring on a global scale, and that may in many parts of the world strongly exceed the rate of sea level rise (Temmerman et al., 2013).

The effect of sea-level rise on the long-term marsh stability by lateral erosion remains, however, much less understood. Models predict that these lateral dynamics are mainly affected by storminess, and that sea-level rise is an important factor in diminishing the effect of wave attenuation by the bare intertidal flats on the seaward side of the marsh (Callaghan et al., 2010; Mariotti and Fagherazzi, 2010; van de Koppel et al., 2005). Time series analyses have shown the importance of the width of the tidal flats in front of a marsh and the position of the channel for marsh stability (French, 2006; Mariotti and Fagherazzi, 2013) and that seaward growth under sea level rise conditions is possible, provided that the slope and sediment concentrations are sufficient (French, 2006). Understanding the interactive effects of sea-level rise and increased storminess on lateral marsh dynamics requires experimental studies to identify processes that actually drive the lateral marsh dynamics. Therefore, identifying these key processes has been a key focus within the THESEUS project, and has shown to be related to local short-term (i.e., within a season) sediment dynamics (Bouma et al., in prep.).

### 3.2. Effects of increased storminess on long-term ecosystem persistence

A future increase in storminess (Donat et al., 2011; Young et al., 2011) may enhance ecosystem disturbance. Disturbance consists of a magnitude and frequency component, with the magnitude determining the scale of the ecosystem setback and frequency determining the time for an ecosystem to recover before being disturbed again. Together, the magnitude and frequency of storm events determine which type of coastal ecosystems are able to persist on intertidal areas, which directly

determines the relevance of intertidal areas for coastal protection (Fig. 4). Long undisturbed periods will typically enable stable climax species to establish, which may be expected to accumulate more biomass than pioneer species, and hence contribute more to coastal protection in terms of wave attenuation. Of course the exact response of a particular ecosystem to the magnitude and frequency of storm events will also depend on other abiotic conditions. For example, sediment availability and stability can be a main factor in ecosystem establishment (Balke et al., 2011, 2012, 2013; French, 2006; Han et al., 2012; Infantes et al., 2011). Such conditional modifiers do however not change the importance of the magnitude and frequency of storm events as driver of ecosystem dynamics.



**Fig. 4.** Schematization of ecosystem dynamics needed to persist under different disturbance regimes, defined as the combination of disturbance magnitude (X-axis) and frequency (Y-axis). Ecosystems with fast dynamics (i.e., typical pioneer species) can tolerate higher disturbance regimes than slowly developing ecosystems (i.e., typically climax ecosystems). As (1) pioneer ecosystems have in most cases a much lower biomass than climax ecosystems, and (2) wave attenuation typically increases with standing biomass, this implies that the ecosystems with the highest wave-attenuating capacities are most difficult to obtain at high energy environments with strong disturbances, unless these disturbances are very rare (like tsunamis) and there are in between these disturbances enough Windows of Opportunity for the ecosystem to establish (e.g., see Balke et al., 2011, 2013). If the disturbance regime is too high to enable ecosystems with wave attenuating above-ground (epibenthic) structures to persist in the intertidal, one can only use supra-tidal ecosystems for coastal defense. The latter is the case on many sandy shores with dunes. The shape of the dashed lines is only indicative for a positive relationship; the actual shape is unknown. Ideally, we would be able to plot within this figure the types of intertidal ecosystem that are useful for coastal defense. However, we currently lack the data to do this, posing an important challenge for future research.

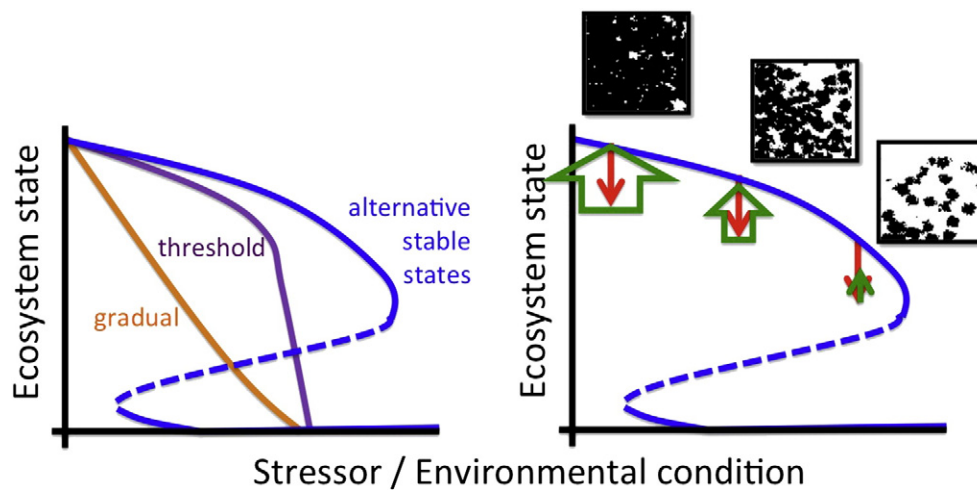
Assessing the sensitivity of ecosystems to disturbance regimes is complicated, especially as many coastal ecosystems follow alternative stable-state behavior (Fig. 5; Scheffer et al., 2009), meaning that disturbances may induce ecosystem collapse due to non-linear threshold dynamics (Scheffer et al., 2001, 2009; van der Heide et al., 2007; van Wesenbeeck et al., 2008). Depending on the ecosystem, this collapse may be fast (e.g., seasonal wiping out of a mussel bed during a storm) or be a gradual process that takes several decades of eroding the slowly built-up ecosystem (e.g. decadal lateral erosion of a saltmarsh; van de Koppel et al., 2005; van der Wal et al., 2008). Once the ecosystem has fully collapsed, restoration of lost coastal vegetation (i.e., seagrasses, salt marshes and mangroves) or biogenic reefs (e.g., mussel beds) is notoriously difficult, as re-establishment often requires size or biomass thresholds to be surpassed (Balke et al., 2011, 2013; Bouma et al., 2009a; van Wesenbeeck et al., 2008) in line with alternative stable state theory (Corenblit et al., 2011; Petraitis and Latham, 1999). To enable the use of ecosystems for coastal protection with predicted safety and persistence values requires knowledge on the long-term ecosystem stability and dimensions. The minimal knowledge demand is the capacity to predict the minimum area of effective ecosystem that will remain available for a few decades, and/or to have good indicators that warn for impending decline, so that timely measures can be taken to prevent degradation.

Based on theoretical models it has been suggested that indicators of impending decline may be derived from spatial vegetation patterns (Fig. 5b; after Rietkerk et al., 2004), but experimental evidence remains extremely rare with a few exceptions (van der Heide et al., 2010). Within THESEUS it was shown for salt marshes, that spatial pattern of tussocks in a salt marsh pioneer zone (van Belzen et al., in prep.) and the morphology of individual tussocks (Balke et al., 2012) may be used to indicate the direction of marsh development in response to intrinsic biogeomorphic feedbacks and external physical forcing by hydrodynamics and sediment dynamics. Both studies were, however, not able to provide indicators for the rate of decline.

Another promising way to assess the long-term stability of ecosystems and their proximity to imminent (catastrophic) decline is based on measuring the time needed for an ecosystem to recover after a disturbance. This concept, known as ‘Critical Slowing Down’, indicates that the time an ecosystem needs to recover is inversely related with ecosystem health, so that ecosystems that are closer to collapse will

recover more slowly (Fig. 5b; after Scheffer et al., 2001, 2009). Albeit an appealing theoretical idea, the translation from these theoretical concepts into practice, to assess resilience based on simple measurable ecosystem properties, remains an important challenge for ecologists to resolve. Within THESEUS, it has been shown for salt marshes (van Belzen et al., in prep.) that the concept of critical slowing down can be used to detect changes in resilience as a result of changes in the environmental conditions. This approach does, however, need further development before it can be used to quantify the exact time scales of ecosystem persistence as needed for deployment in coastal defense.

Our limited ability to make quantitative predictions on the long-term persistence may be the main knowledge gap that limits the use of coastal ecosystems in coastal defense. As long as reliable indicators to quantitatively predict ecosystem collapse are lacking for many coastal ecosystems, the best practice would be to use current knowledge on the time scale of inherent ecosystem dynamics. For example, for salt marshes in the Westerschelde estuary it is known that they have inherent cyclic behavior at a decadal scale, as can be derived from analyzing aerial photography (van der Wal et al., 2008). For seagrasses, the time scale of ecosystem dynamics can be much shorter, with a high variability on the temporal extent over the years (Valle et al., 2013). But this depends on the seagrass species studied and local conditions, as under favorable conditions seagrass beds can persist for prolonged periods. Some reef-building organisms create stable structures that can even persist after the organisms die (e.g. *Sabellaria* spp. reefs & oyster reefs; Gruet, 1986). In contrast, other reef builders such as mussels, create beds that are much more volatile, and may not persist for long after the animals die. At this stage, there is a lack of fundamental approaches to predict long-term ecosystem stability, making local expert judgment an important input in the design of coastal defense structures. This knowledge gap is the most urgent challenge that needs being addressed by ecologists, to enable a broader implementation of coastal ecosystems in coastal defense structures in the future. In our opinion, addressing this knowledge gap requires process-based studies that describe mechanisms involved in passing the thresholds (e.g., Balke et al., 2011, 2013), so that these mechanisms can be incorporated in long-term biophysical modeling of coastal ecosystems (e.g. Kirwan and Murray, 2007; Temmerman et al., 2007).



**Fig. 5.** Schematized explanation of the concept of alternative stable states (ASS) and measuring critical slowing down to measure nearness to collapse (modified after Scheffer et al., 2001 and Rietkerk et al., 2004). Ecosystems can respond in various ways to a gradual increase of stressors or environmental conditions (left panel): gradual or linearly versus non-linear. In case of a non-linear response, the response typically becomes stronger when surpassing a threshold. In case of discontinuous thresholds, multiple stable states exist, meaning that at the same environmental setting an ecosystem can occur in two contrasting states (e.g., vegetated versus bare). In case of two alternative stable states, reversal of an ecosystem state change is very difficult once the threshold is surpassed. Theoretical investigations suggest that it is possible to foresee impending state changes (right panel) by studying the spatial structure of the ecosystem (i.e., patterning) or by measuring the return time (green arrows; a wider arrow indicates a faster return) to equilibrium after a disturbance (red arrows; arrows of equal width to indicate same disturbance level) following the critical slowing down concept (Scheffer et al., 2009).

#### 4. Potential use of ecosystems in coastal defense designs: integrating ecology & engineering

Implementing the use of ecosystems in practical coastal defense schemes poses a number of practical questions: (1) In which environments can we benefit from including intertidal ecosystems in coastal defense? (2) To what extent can we create those ecosystems as needed for defense also at the location where they are needed? (3) If ecosystems are created for coastal defense, are they also valuable in terms of e.g., diversity and ecosystem functioning? (4) How do we account for the benefits of an ecological design in terms of direct cost (savings) vs. indirect services?

##### 4.1. In which environments can we benefit from using intertidal ecosystems in coastal defense?

There are many coastal defense designs where engineering could benefit from integration of ecology. Naturally, the first requirement is that conditions should allow an ecosystem to establish and persist. As a rule of thumb, it may be logical to expect that the higher the *average* energy level occurring in the ecosystem, the smaller the opportunity for using intertidal ecosystems for defense (Fig. 4). In general, intertidal ecosystems that are valuable for coastal defense perform better in low energy environments, with relatively mild *average* hydrodynamic stress. This may wrongly suggest that these ecosystems are not valuable for coastal defense, as they occur in the ‘wrong’ (i.e., too sheltered) places. The interesting aspect is that intertidal ecosystems adjusted to mild *average* hydrodynamic stress, may however be very valuable for protection under rare extreme conditions. The latter may perhaps be most clearly demonstrated for mangroves. For their establishment, they require long enough disturbance free periods with extremely mild hydrodynamic conditions, referred to as “Windows of Opportunities” (Balke et al., 2011, 2013). Despite these narrow establishment chances, once mangrove forests are well established, they may protect people against the effect of events as big as a tsunami (Alongi, 2008; Feagin et al., 2010; Gedan et al., 2011; Spencer and Möller, 2012). Overall, it is thus highly important to account for disturbance free periods and the expected event-size distribution (Fig. 4). That is, as long as there is opportunity for an ecosystem to establish, and the average growth conditions allow the system to persist, they may offer important protection against severe events that occur only rarely. In that perspective, temperate system may have an advantage in that the storms often fall outside the growth conditions. In the most exposed locations where conditions are too harsh for an ecosystem to establish, temperate engineering measures might be used to enable initial establishment.

##### 4.2. Can we create ecosystems at the locations where we want them?

Ecosystems may exhibit alternative stable state behavior (Fig. 5), meaning that with the same physical conditions a habitat may be occupied by two contrasting ecosystem states (Scheffer et al., 2001). For example, a tidal flat can be unvegetated and fully occupied by sediment destabilizing benthos versus fully vegetated with seagrass meadows, salt marsh vegetation or mangrove forests (e.g. Marani et al., 2010; van der Heide et al., 2010). These alternative stable-state dynamics are the result of stabilizing feedbacks, causing a strong resilience against state changes (Petraitis and Latham, 1999). Because these stabilizing feedbacks are typically density or size dependent, they do not ‘help’ single establishing individuals (Bouma et al., 2009a, 2009b). Hence, ecosystems exhibiting alternative stable states typically have strong establishment thresholds (Balke et al., 2013; Suykerbuyk et al., 2012). This makes the creation of ecosystems at desired locations rather cumbersome, as may be seen from the large number of failed restoration projects in coastal ecosystems.

As indicated in the previous section, establishment of ecosystems may require a window of opportunity with relatively mild conditions, either of physical (Balke et al., 2011, 2013) or ecological nature

(Suykerbuyk et al., 2012), to allow it to surpass an establishment threshold. Engineering measures, either aimed at temporarily reducing disturbances (e.g. hydrodynamic forces, sediment dynamics or negative biotic interactions) or aimed at providing more settlement substrate (e.g., providing stable substrate for reef builders to settle), might offer a useful way to enhance the establishment of species at locations where they have been lost, or where they are desirable as part of the coastal defense scheme. This does, however, mean that ecologists need to provide engineers with detailed mechanistic insights in the establishment thresholds: what are the main processes that need (temporary) alteration for establishment to take place? Whereas for some ecosystems this knowledge is developing, for most it is still lacking in part due to limited interdisciplinary research of engineers and ecologists.

##### 4.3. Are ecosystems created for coastal defense valuable: importance of natural dynamics?

Within the context of mitigation and compensation, the legislation puts relatively little attention on the question to which extent restored and/or created ecosystems are equally valuable as the original that has been lost. This is surprising, as from a cultural perspective we would never allow an original statue or painting to be willingly destroyed and replaced by a new copy. In that sense, it is an important question to try to define which criteria must be met, for ecosystems that have been created for coastal defense purposes to be as valuable as naturally evolved ecosystems. Parameters as species diversity and ecosystem functions might be the obvious ones to compare ecosystems. Several studies have shown that at first sight well restored or created ecosystems may score lower on these quantitative measures than a naturally developed ecosystem (Moreno-Mateos et al., 2012; Mossman et al., 2012).

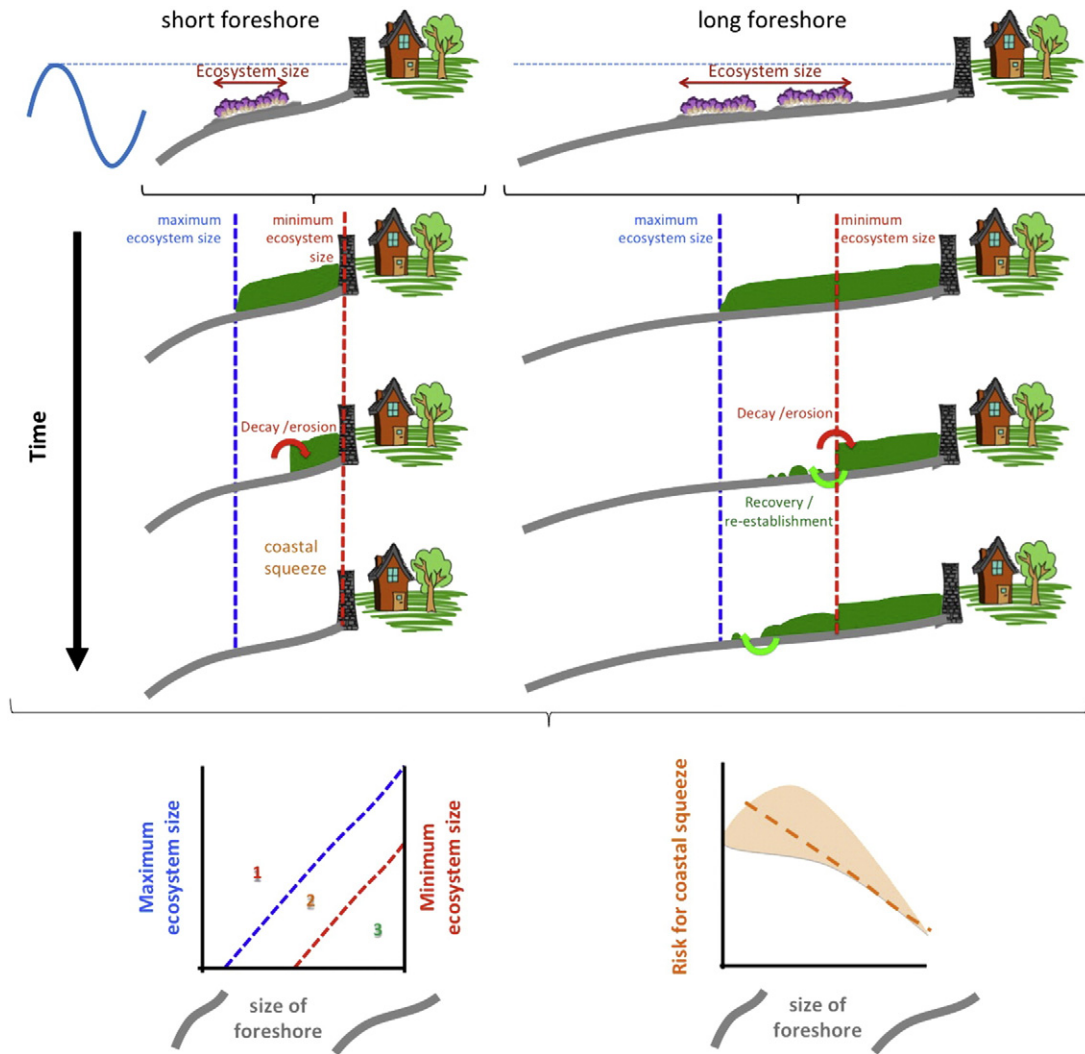
Coastal ecosystems may have inherent properties that follow from how natural processes organize (e.g., for marshes see Temmerman et al., 2007) and continuously rejuvenate (e.g., for marshes see van de Koppel et al., 2005) the landscape. Such self-organization and natural dynamics may be more difficult to assess and evaluate, given the long time needed for these processes to evolve within a landscape. Current policies often have static targets of protecting and maintaining what is there, and often do not account for the natural dynamics that is characteristic for coastal ecosystems. With respect to safety aspects, the margin of allowing natural dynamics to occur will even become narrower. The only way to avoid this is to make the area for an ecosystem sufficiently large to account for natural dynamics while always maintaining the minimal area needed for coastal defense (Fig. 6). As the latter may imply giving up terrestrial areas for coastal defense, this ideal solution may only be possible at some locations where human land use allows it. In other areas, maintaining the ecosystem may require regular human intervention, reducing the inherent value of the ecosystem.

##### 4.4. Account for benefits of an ecological design: direct cost (savings) vs. indirect services?

The two great promises of integrating ecosystems in coastal defense schemes is that (1) they may be adaptive to climate change processes (Borsje et al., 2011) and (2) that they may provide additional ecosystem services. Adaptation to climate-change processes means that by natural processes (e.g., trapping and accretion of sediment), the ecosystem may adapt to long-term trends (e.g., follow sea-level rise). The latter is a great opportunity, but may also impose a great threat if the long-term persistence cannot be quantitatively assessed (see sections on long-term persistence).

Research on ecosystem services (supporting, provisioning, regulating and cultural services; MEA, 2005) is currently booming, and our insight in the additional values of ecosystems for coastal protection is rapidly growing (e.g., see Fisher et al., 2009, 2011; Fourqurean et al., 2012). Overall, coastal wetlands rank as highly (economical and ecological) valuable ecosystems (Costanza et al., 1997), and





**Fig. 6.** Schematization of the relation between foreshore dimensions and the maximum and minimum widths of intertidal ecosystem with wave attenuating aboveground (epibenthic) structures. The maximum and minimum widths relate to the borders reached by intertidal ecosystems with cyclic dynamics: the minimum width is the size of the ecosystem that will persistently present, whereas everything between the maximum and minimum will be variable over time. A wider foreshore will inherently offer more space for intertidal ecosystems. Moreover, a wider foreshore will generally have less strong gradients in wave energy than a narrower foreshore, thereby making it easier for epibenthic ecosystems to establish. As a result of this, both the maximum and minimum widths of an intertidal habitat will have a positive relationship with the size of the foreshore. As such, a sufficiently wide foreshore is important to enable epibenthic intertidal ecosystems to go through natural cycles of decay and re-establishment (cf. van de Koppel et al., 2005), without suffering from coastal squeeze. On a narrow foreshore, re-establishment of degrading epibenthic ecosystems might be hampered by too strong gradients in wave energy. Summarizing, this implies that (1) at too short foreshores the chances that wave-attenuating epibenthic intertidal ecosystems can establish are relative low; (2) wave attenuating epibenthic intertidal ecosystems occurring on foreshores of intermediate size may face risk of coastal squeeze; (3) wide foreshores provide the space that enables intertidal habitats to go through (natural) cycles of decay and recovery. Consequently, the length of the foreshore affects the long-term stability, and thereby sustainability, of the intertidal ecosystem as part of the coastal defense scheme. The shape of the dashed lines is only indicative for a positive relationship; the actual shape is unknown.

creation and restoration of them are thus highly desirable. An important aspect that remains understudied, in our opinion, is the service that creating new ecosystems provides enhancing connectivity between ecosystems (e.g., Nagelkerken, 2000). This service is however extremely hard to quantify, and needs further attention in future research (also see Koch et al., 2009). There should also be attention for potential negative aspects, such as the chance of facilitating the spread of invasive species or the transformation of valuable habitat for wading birds such as tidal flats (soft substrate) into vegetated areas or constructed reefs (hard substrate).

**5. Conclusions**

Ecosystems can provide a valuable contribution to coastal defense, which deserves to be more actively considered when planning coastal defense schemes in the future. This may be especially valuable in those areas where coastal ecosystems have been lost in the past. Coastal

defense value of ecosystems is, however, strongly site specific, in that it depends on the tidal range of the area and on the space available in the intertidal zone. Ecosystems can contribute most to coastal protection by wave attenuation in areas with relative small tidal amplitude, but may in such areas also be most sensitive to sea level rise. Applying ecosystems in coastal defense schemes is most effective and sustainable when intertidal areas are wide. This aspect is currently understudied. The main bottlenecks to large-scale application to date are: (A) data on the true protective value of coastal ecosystems under extreme storm conditions, (B) knowledge on long-term resilience and (C) insight in thresholds for establishment. These bottlenecks may be for an important part be ascribed to the lack of interaction between coastal engineers and ecologists.

We have increasing insight in which ecosystems and organism traits are important for the protective value of ecosystems, but we lack data on the true protective value of such coastal ecosystems under extreme storm conditions. To address this knowledge gap, we need models

that predict their effectiveness under relevant storm conditions as well as fully self-contained automated monitoring systems that can gather data on the rare events needed to validate model predictions.

Coastal ecosystems are dynamic systems. This offers advantages, in that they can adapt to environmental conditions (e.g., keeping up with sea-level rise). However, these dynamics also create uncertainty. There is potential for collapse or shift to an alternative ecosystem that may not have the same contribution to coastal defense. This makes the long-term defense value hard to predict. It is clear that coastal protection by ecosystems imposes space requirements, with the exact dimensions depending on local settings, including tidal range.

Most ecosystems that are relevant for coastal protection typically only establish in areas with average moderate physical forcing, where windows of opportunity enable passing establishment thresholds. This does not mean that these ecosystems are not important for coastal defense, as their defense value is only really demanded for during rare extreme events. However, it does mean that ecosystem creation and restoration may require (temporary) engineering measures. This requires ecologists to provide engineers with detailed mechanistic insights, which is currently often lacking.

## Acknowledgments

The authors would like to appreciate feedbacks provided by two anonymous reviewers, who greatly improved the manuscript. The EU is greatly acknowledged for funding our research through FP7.2009-1, contract 244104—THESEUS (“Innovative technologies for safer European coasts in a changing climate”).

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