Chapter 1

General introduction
Introduction

Many coastal communities are currently facing flood risks worldwide (Nicholls et al. 2007) due to accelerating sea level rise (IPCC 2014), land subsidence (Syvitski et al. 2009; Weston 2014) and extreme events such as storms surges (Kiesel et al. 2021), which will probably become more frequent with climate change (Menéndez and Woodworth 2010; IPCC 2014; Vousdoukas et al. 2018). Extreme sea levels have become more frequent worldwide since the 1970s (Woodworth et al. 2011; IPCC 2014), and while changes in mean sea level are long term slowly increasing, extreme sea level events have immediate impacts on the coast (Menéndez and Woodworth 2010). Particularly, extreme sea levels may become more important in areas with high flooding risk due to low land elevations and constructions near the coastline, as is the case in the Netherlands (Bouwer and Vellinga 2007).

As a result, increasing investment in coastal defence structures is needed worldwide (Temmerman et al. 2013). Hard engineering structures such as dikes, sea walls, breakwaters and storm surge barriers, also called ‘grey’ solutions, are commonly used for coastal protection (Fig. 1). Nevertheless, conventional hard engineering is often associated with negative impacts on natural functioning and biodiversity of coastal ecosystems and their ecosystem services (e.g. Lai et al., 2015). Furthermore, because conventional hard engineering is static, it can be increasingly challenged by climate change and the maintenance may become very costly (Temmerman et al. 2013; Bouma et al. 2014; Morris et al. 2020).

Coastal protection by natural foreshore ecosystems

An alternative to the ‘grey’ solutions is the ecosystem-based coastal defence, also called nature-based coastal protection or ‘green’ solutions (Fig. 1). Hybrid ecosystem-based coastal defence combines conventional hard engineered barriers with coastal ecosystems (Fig. 1 and 2) and can be a more sustainable, ecologically valuable and cost effective alternative to hard engineering alone (Shepard et al. 2011; Temmerman et al. 2013; Morris et al. 2018; Schoonees et al. 2019; Vuik et al. 2019; van Zelst et al. 2021). In contrast to engineered solutions, ecosystems can be capable to spontaneously recover from storm disturbances and
be resilient against sea-level rise (Feagin et al. 2015; Kirwan et al. 2016; Fagherazzi et al. 2020; Morris et al. 2020). In addition, they provide a broad range of ecosystem services (Orth et al. 2006; Gedan et al. 2009; Barbier et al. 2011). These include biodiversity conservation (e.g. Terrados and Borum 2004; Spencer and Harvey 2012), nutrient cycling (e.g. Human et al. 2015), support for fisheries (e.g. Lilley and Unsworth 2014; Christianen et al. 2017), water purification (e.g. Andrews et al. 2006) and carbon sequestration (Duarte et al. 2013).

Sand dunes, marshes, mangroves, seagrass, kelp forest, coral and shellfish reefs are coastal ecosystems recognised for their coastal protection value, especially along soft-bottom coasts (Temmerman et al. 2013; Schoonees et al. 2019). This thesis is focused on tidal flats, seagrass, shellfish reefs, dunes and with most emphasis on salt marshes (Box 1). These so-called foreshore ecosystems are able to attenuate waves and currents and stabilise the soil (Shepard et al. 2011; Bouma et al. 2014). For example, salt marshes have been found to reduce waves even under storm surge conditions (Möller et al. 2014; Willemsen et al. 2020), lower the wave run-up on the dikes compared to dikes with bare tidal flat in front (Vuik et al. 2016; Zhu et al. 2020), reduce the chance of dike breaching during extreme events, and reduce the size of dike breaches when dikes fail (Zhu et al. 2020). Furthermore, the protection by coastal ecosystems may be the result of long-distance interactions and facilitation effects (Bos and Van Katwijk 2007; Bouma et al. 2014; van de Koppel et al. 2015; Schoonees et al. 2019). For example, mussel or oyster beds may act as natural wave breakers behind which seagrass beds can establish, which in return may affect tidal flat morphology promoting marsh development, and finally the salt marsh may protect the seawall or dike (Fig. 2).

Although there is mounting evidence for ecosystem-based coastal defence being a cost-effective and ecological valuable alternative to ‘grey’ infrastructure alone, uncertainties about the actual effectiveness still hampers the practical implementation of these ecosystem-based measures. There is not a one-size-fits-all ‘green’ solution and each situation requires a tailor-made approach but based on common principles. For instance, natural ecosystems can adapt to sea level rise by landward migration. However, in urbanized areas, this migration may be prevented by hard structures constructed along the coastline (i.e. urbanization, roads, dikes, seawalls), a process called coastal squeeze (Doody 2013). Therefore, there are coastal settings
where nature-based solutions alone can be enough, while in other situations, with harsher environmental conditions and where landward migration of the ecosystems is not possible, hard engineering may be needed (Morris et al. 2018, 2020; Schoonees et al. 2019) (Fig. 1). On top of that, coastal ecosystems are degrading due to anthropogenic pressures and climate change (Orth et al. 2006; Syvitski et al. 2009; Murray et al. 2019). Therefore, there is an urgent need to advance the understanding of coastal ecosystem dynamics in space and time, and how management can affect these systems in order to optimize their coastal protection function (Bouma et al. 2014).

**Box 1. Systems included in this thesis**

**Salt marshes**

Found at relatively sheltered locations, salt marshes are salt-tolerant vegetated areas at the transition between land and sea, subject to periodic flooding (Allen 2000). Salt marshes can attenuate hydrodynamics (i.e. waves and currents) and create stable and elevated soils by trapping sediment and accumulating organic matter. This leads to a positive feedback by creating suitable conditions for the growth of more vegetation and in return attenuate more hydrodynamics (Allen 2000, Koppel et al. 2005).

Salt marshes can be found in the mainland (i.e. mainland marshes), or in barrier islands, growing behind the shelter of a dune or chenier ridge (i.e. back-barrier marsh) (Bakker 2014). In mainland marshes, the elevation gradient goes from the dike toe to the tidal flats, while in back-barrier marshes, the elevation gradient goes from the dune toe to the tidal flats (Bakker et al. 2015). Due to intrinsic feedbacks (Koppel et al. 2015) or factors such as changing exposure or sediment supply (e.g. Wang et al. 2017, Ladd et al. 2019), marshes can retreat landwards. This process is also called cliff or lateral erosion.
**Tidal Flats**

Tidal flats are soft-sediment environments regularly flooded by the tides that form in intertidal areas where sediment is deposited by the tides or rivers (Murray et al. 2019). They can be found in front of salt marshes, mangroves or directly in front of the dikes or seawalls and extend until the mean low-tide elevation. When composed of mud, they are also referred as mudflats. Tidal flats can host high biodiversity of benthic species (Christianen et al. 2017b) and are important for migratory birds (Boere and Piersma 2012).

**Mussel Beds**

Mussel beds are a type of shellfish reef that occurs in subtidal (always below water) and intertidal (flooded during high tide) zones along the coast (Folmer et al. 2017). They provide hard substrate in soft-sediment environments and can accrete sediment, which is related to waves and currents attenuation (Donker, Vegt, & Hoekstra 2013). Mussel beds were not directly studied in this thesis, but the description serves as a background for the chapter 7, where we utilised biodegradable artificial reefs aimed at mussel bed restoration.
**Seagrass beds**

Seagrasses are aquatic flowering plants that evolved from freshwater plants to grow on marine or brackish environments. They can grow in the intertidal to subtidal zones along the coast, always at lower tidal elevations than the salt marshes (Short et al. 2007). They can trap sediment and attenuate waves and currents, which makes them valuable for coastal protection (Ondiviela et al. 2014). *Zostera marina*, also known as eelgrass, is the species studied in this thesis and the most wide-ranging marine flowering plant in the Northern Hemisphere (Short et al. 2007).
**Soft-sediment barrier islands**

Barrier islands are migrating and dynamic soft-sediment coastal landscapes that accommodate both sandy dune barriers and silty back-barrier marshes in their wake (Oertel 1985). Dune formation occurs above the mean high-tide level and it is mostly driven by aeolian sand deposition facilitated by vegetation (Otvos, 2020). Situated parallel to the mainland coast, barrier islands are the first barrier attenuating waves coming towards the coast from the open seas (Otvos, 2020). Back-barrier islands, like Griend in the Netherlands, are a type of smaller barrier islands that can be found behind oceanic barrier islands (Cooper et al. 2007). In contrast to the ocean barrier islands, back-barrier islands do not often form a dune through aeolian sand transport (Pilkey et al. 2009). Instead, they present a beach, small dunes or a chenier ridge on the exposed side formed during strong storm events (Pilkey et al. 2009).
Introduction

**Foreshore ecosystem impacts on wave attenuation and run-up**

After storms, the maximum height that waves reach onto the dikes can be measured as the vertical elevation at which beach wrack is left on the dike (Spencer et al. 2015; Vuik et al. 2016; Zhu et al. 2020) (Fig. 3). The actual wave run-up onto a dike is the distance from the storm-surge water level to the beach wrack level (Fig. 3). Therefore, beach wrack levels depend on the wave properties and storm-surge levels, with large waves combined with high water levels resulting in the highest beach wrack levels (Vuik et al. 2016; Zhu et al. 2020). The probability of water overtopping the dike (i.e. water flowing over the dike) increases if storm-surge levels and thus wave run-up increase due to global change. Dike overtopping risk may be reduced by the use of ecosystem-based coastal defence, where natural ecosystems in front of dikes would reduce wave heights leading to lower wave run-up onto the dike (Vuik et al. 2016). Furthermore, the reduction of wave energy arriving to the dikes may also prevent dike failures (Vuik et al. 2019; Zhu et al. 2020).

Fig. 3. Diagram describing the terms related to the run-up on the dikes. Below, a photo of a beach wrack in a dike fronted by a salt marsh. NAP is the Dutch Ordinance Level, similar to mean sea water level.
Bare tidal flats can attenuate waves and currents (Le Hir et al. 2000; Hu et al. 2015). Particularly, wave forcing is more efficiently dissipated in higher, wider, convex and gentler sloping tidal flat profiles than concave and steep profiles (Le Hir et al. 2000; Hu et al. 2015). In addition, structures on tidal flats such as reefs, macroalgae, seagrasses or annual saltmarsh plants will promote further waves and flow attenuation by the creation of drag forces (Pinsky et al. 2013; Walles et al. 2015; Leonardi et al. 2018). Such communities at low elevations along the foreshore can attenuate hydrodynamics (e.g. waves and currents) and trap sediment, thus changing the tidal flat morphology (e.g. Bos et al. 2007; Christianen et al. 2013; Walles et al. 2015), which may ultimately facilitate the establishment and stability of communities at higher elevations such as salt marshes and mangroves (Gillis et al., 2014; van de Koppel et al., 2015). Communities higher in the intertidal zone such as perennial salt marsh vegetation, mangroves or dune vegetation will attenuate more hydrodynamics than communities at lower elevations (Bouma et al. 2014) as a result of their persistent and higher foreshore which enhances wave attenuation combined with vegetation friction ( Möller et al. 1999; Vuik et al. 2016; Gracia et al. 2018; Brooks et al. 2020).

The ratio of water depth to structure height is an important variable for wave attenuation. The lower the water levels are relative to the structure height, the more attenuation will occur (e.g. Möller et al. 1999; Ysebaert et al. 2011; Yang et al. 2012; Chowdhury et al. 2019). This is one of the reasons why marsh vegetation is more effective in the overall wave attenuation than seagrass beds or biogenic reefs which are generally associated with deeper water levels (Bouma et al., 2014). In the case of salt marshes, the promotion of sediment accretion by the vegetation and resulting increase in soil elevation, will in return enhance wave attenuation and promote better conditions for vegetation, since plants become less exposed to inundation (Redfield et al. 1972; Morris et al. 2002; Van de Koppel et al. 2005; Bouma et al. 2009b). Vegetation may also increase the soil elevation by belowground biomass production (Morris et al. 2002; Cahoon et al. 2006; Kirwan and Guntenspergen 2012). Marshes where organic accumulation dominates the vertical accretion and have low sediment input can be classified as organogenic marshes (e.g. found in the United States and microtidal marshes) (Nolte et al. 2013; Kearney and Turner 2016). In contrast, vertical accretion in minerogenic marshes depends mainly on sediment supply from tidal flooding (common in Europe and south-east USA) (Bakker et al. 2015). In general, organogenic marshes will have higher rates of subsidence as their sediment is compacted or decomposed faster than in minerogenic marshes (Nolte et al. 2013).

Wave attenuation by vegetation friction will also depend on the vegetation characteristics as the number of structures (leaves or branches), stem density and stiffness (Möller 2006; Bouma et al. 2005, 2010; Ysebaert et al. 2011). Stiffer plants such as Spartina sp. (marsh species) will pose more friction to the water, therefore dissipating more waves and currents compared to more flexible vegetation such as Zostera sp. (seagrass species) (Bouma et al. 2005b; Schwarz et al. 2015). Wave attenuation will also vary over the seasons, with greater attenuation in summer due to the higher density of vegetation (Vuik et al. 2016, 2017).
Nevertheless even for the considerable inundation depths that occur during storms and with winter state vegetation, saltmarshes can be efficient in reducing wave impacts, especially due to their elevated and stable foreshores (Möller et al. 2014; Vuik et al. 2016; Willemsen et al. 2020). Finally, wider marshes are expected to attenuate more waves and therefore provide more protection compared to narrow marshes or bare tidal flats (Shepard et al. 2011). For this reason, marsh width is one of the most important factors related to the coastal defence function of marshes (Shepard et al. 2011; Bouma et al. 2014).

The effect of (vegetated) foreshores in wave run-up can be modelled based on wave properties and storm surge levels (e.g. Vuik et al. 2016). However, experimental evidence on how (vegetated) foreshores affect wave run-up including contrasting foreshore types (i.e. different wind exposure, vegetation, tidal range and bathymetry) remains scarce (Spencer et al. 2015; Zhu et al. 2020). Understanding the effect of foreshores on run-up requires in depth understanding of i) at which locations you can have salt marshes and ii) how effective such (vegetated) foreshores are in reducing wave run-up onto a seawall. This will be addressed in chapter 2.

Foreshore sediment stability

Sediment erosion can occur on the soil surface or on the lateral of a cliff (Box 1, Fig. 4). Soil surface erosion, hereafter called top erosion, can be caused by the initiation of horizontal sediment transport (i.e., bed load) or by sediment resuspension (i.e., suspended load) (Einstein et al. 1940; Brown et al. 1995) due to increased bed shear stress by currents or waves (Van Rijn 1984a; b; Brown et al. 1995; Ganthy et al. 2015). Bed load occurs when sediment particles move along the bottom horizontally by rolling whereas sediment resuspension occurs when the sediment particles are lifted vertically into the water column creating turbidity and reducing light (Einstein et al. 1940; Brown et al. 1995). Lateral erosion, also called cliff erosion, can be in form of gradual detachment of soil particles from the sediment due to hydraulic pressure (e.g. Feagin et al. 2009; Twomey et al. 2021) (Fig. 4). Lateral erosion can also occur in form of big blocks or “mass failure”, which can occur due to undercutting, normally by wave swash, and the creation of tension cracks (Schwimmer et al. 2001; Fracalanci et al. 2013; Priestas et al. 2015). Undercutting can occur in marsh cliffs when the sediment below the cohesive top layer of the marsh starts to erode, and the top layer with roots overhangs until it breaks off and collapses (Fig. 4).
Fig. 4. Types of sediment erosion present in foreshores. Same erosion as represented for a salt marsh can be applied to bare tidal flats. Seagrass can be subtidal (always below water) or intertidal (flooded daily).

Fine grained soils with silt, clay and organic matter are more cohesive and less erodible (Brown et al. 1995; Gailani et al. 2001; Grabowski et al. 2011). In contrast, sandy soil is much less cohesive and therefore less resistant to erosion (Brown et al. 1995). On the other hand, even small percentages of clay (5-10%), will make the sediment more cohesive, having a significant effect on erosion reduction (Brown et al. 1995; Gailani et al. 2001; Grabowski et al. 2011). Silty sediment without clay, even being very fine, can be non-cohesive and be easily eroded (Brown et al. 1995). Clay fraction can be more time consuming to determine, and commonly used instruments to determine the grain size, such as Malvern Laser Particle Sizer®, can underestimate the clay content (Vroom et al. 2014). In some regions, like in the Wadden Sea, which is one of our study sites, the clay-silt fraction in marine soils, also called mud, correlates to the clay fraction (Van Ledden et al. 2004). For example, sandy sediment with more than 20% of mud was found to be more cohesive and had lower erosion rates than sandy sediment with low percentage of mud (Houwing 1999; Lo et al. 2017). For these reasons, in many studies on erosion, the clay-silt fraction or ‘mud’ has been used as a variable to relate to erosion instead of clay alone.

Coastal vegetation can reduce erosion by waves and currents (Ward et al. 1984; Christianen et al. 2013; Brooks et al. 2020), which is normally attributed to a reduction of the hydrodynamics within the canopy and hence lower shear stress on the soil (Bos et al. 2007; Infantes et al. 2012; Möller et al. 2014; Feagin et al. 2019; Türker et al. 2019). However,
vegetation with little or no aboveground biomass, such as in winter-state, due to grazing (Christianen et al. 2013; Paul and Kerpen 2021) or with the leaves broken or flattened due to wave energy (Møller et al. 2014; Spencer et al. 2016), has been observed to still limit erosion. This suggests that changes induced in the sediment as a result of vegetation presence and the belowground biomass (i.e. roots and rhizomes, Fig. 4) can increase soil resistance to erosion.

Vegetation is able to accumulate fine sediment (silt and clay) and organic matter (Allen 2000; Feagin et al. 2009; Marani et al. 2013), thus enhancing properties that can make the sediment more cohesive and less erodible (Brown et al. 1995; Gallani et al. 2001; Grabowski et al. 2011). In terrestrial ecosystems, top erosion by runoff is reduced with increasing fine root density (below 1 or 0.5 mm Ø, depending on the author) compared to bare soils (Li et al. 1991; Baets et al. 2006, 2007; Burylo et al. 2012). Studies on salt marshes show that higher belowground biomass is related to less top erosion (Coops et al. 1996; Spencer et al. 2016), even with the vegetation in winter state (Paul and Kerpen 2021). Belowground biomass has been also related to a reduction in lateral erosion in marshes (Ford et al. 2016; Wang et al. 2017), with the strongest effects found in sandy soils (Lo et al. 2017; De Battisti et al. 2019). However, the soil stability of the marshes under fast flow conditions like would occur during a dike breach, has not yet been studied (Fig. 4). This is becoming increasingly important when dikes protect people living in low-lying areas that are being faced with sea level rise (Zhu et al. 2020). Hence, there is urgent need to gain in depth understanding of the erosion resistance of foreshores fronting dikes against fast flow running over the soil surface (chapter 3).

Sandy systems such as sandy barrier islands and back-barrier islands, are also an important part of coastal protection in shallow soft-bottom coasts like the Wadden Sea as they are the first barrier attenuating waves coming towards the coast from the open seas (Cooper et al. 2007; Otvos 2020). Sandy islands can include dunes, marshes and tidal flats (Box 1). These sandy systems can be vulnerable to erosion due to their low cohesive soils, especially if there is not enough recovery time in between storms (Timmons et al. 2010; Cooper 2013; Durán Vinent and Moore 2015; Vinent et al. 2021). For this reason, these systems are increasingly being managed to prevent their complete erosion (Govers and Reijers 2021). However studies testing their erodibility are scarce, especially in back-barrier islands. Hence, it is important to understand their stability to better understand how to manage them (chapter 4). Furthermore, this knowledge is important for being able to integrate sandy coastal systems into ecosystem-based, sustainable flood defence systems. As mentioned above, marsh vegetation can reduce cliff and top erosion. Regarding dune vegetation, to the best of our knowledge, studies on the effect of belowground biomass on top erosion have not been done yet, but belowground biomass has been related to a reduction in lateral erosion (De Battisti et al. 2020).

Finally, seagrass beds are also known to stabilize soils and attenuate waves which are valuable properties for coastal protection (Bouma et al. 2014; Ondiviela et al. 2014). However the effect of belowground biomass on soil stability is still relatively poorly studied (Bouma et al. 2014). Christianen et al. (2013) found that seagrass rhizomes and roots seem to play a major
role in topsoil erosion, but to the best of our knowledge, there are no mechanistic studies that
directly quantify the effect of belowground biomass. Understanding the effect of seagrass on
sediment erosion is important to maintain the water clarity, necessary for seagrass development
(Dennison 1987; Duarte 1991), and to retain the sediment in coastal areas (Christianen et al.
2013; Ganthy et al. 2015). The role of eelgrass on top erosion under wave conditions will be
studied in chapter 5.

Grazing management on salt marshes for coastal protection

Biophysical (grain size, compaction, rooting) properties of marsh soils can change with grazing
management in combination with marsh age and elevation (Davidson et al. 2017 and
references therein). In turn, these changes may affect the lateral erodibility of marshes. For
example, marshes which develop from initially sandy intertidal flats and sand banks, will develop
a cohesive fine-grained top layer over time during vegetation succession which will be even
thicker at lower elevations due to higher flooding frequency and hence higher sediment imports
(Olff et al. 1997; Van de Koppel et al. 2005; Elschot et al. 2013). Livestock grazing, which is
often used for managing biodiversity in salt marshes, can modify these soil properties (Howison
et al. 2015; Davidson et al. 2017 and references therein), thereby potentially also reducing
erosion (Pagés et al. 2018). Small grazers such as hare and geese have also shown to structure
salt marsh vegetation communities (Kuijper and Bakker 2005; Chen et al. 2019), although
previous studies showed no significant effects of these small grazers on soil properties (Elschot
et al. 2013). However, as different plant species have different amount of belowground
biomass, we could expect differences on erosion depending on the vegetation species (Lo et
al. 2017; Wang et al. 2017). Understanding how grazing management on salt marshes can
affect soil resistance to lateral erosion is important especially in face of climate change. The
frequency and intensity of storms and storm surges are expected to increase (IPCC 2014).
Extreme storms may induce the formation of a marsh cliff (Bouma et al. 2016), where especially
frequently occurring moderate (winter) storms will determine the rate of lateral retreat (Leonardi
et al. 2016). Hence, understanding the susceptibility of salt marshes to lateral erosion is of key
importance for being able to integrate marshes as sustainable flood defence strategies. An
integrated view on how grazing management in combination with abiotic factors, such as marsh
age, marsh elevation and sediment layering, affect the susceptibility of marsh edges to lateral
erosion, will be addressed in chapter 6.
Management options for tidal flats to promote marsh expansion

Salt marsh presence and expansion depends on different factors such as
i) soil elevation relative to the mean sea level (Balke et al. 2016), which is directly related to
the flooding frequency and duration (Cox, Wadsworth and Thomson, 2003; Mateos-Naranjo
et al., 2008; Wang and Temmerman, 2013; Silinski et al., 2016),
ii) the presence of undisturbed periods of time (windows of opportunity) for the seedlings to
establish and develop before a disturbance occurs (Balke et al. 2011; Hu et al. 2015),
iii) the wind exposure of the area, which is related to increased waves and currents and in turn
to less seedling establishment even having enough soil elevation (Silinski et al. 2015; Wang
et al. 2017)
iv) bed level changes at the transition of tidal flats and salt marshes (Bouma et al. 2016;
Willemsen et al. 2017) and
v) having enough sediment supply to sustain or create new elevated tidal flats for marsh
expansion and keep up with sea level rise (Ladd et al. 2019; Fagherazzi et al. 2020; Liu et
al. 2021).

Creating higher and convex foreshores may lead to higher wave dissipation and longer
distance from wave breaking points toward the potential pioneer marsh zone (Mariotti and
Fagherazzi 2013; Hu et al. 2015; Bouma et al. 2016). This would reduce erosion in the pioneer
vegetation zone of a salt marsh and promote marsh establishment (Mariotti and Fagherazzi
of marsh expansion is often managed with manmade interventions in the tidal flats in front of
the marshes such as building wave-breaking brushwood groynes and digging drainage channels
(Dijkema et al. 2011; Siemes et al. 2020), hard-engineered coastal structures such as groynes
and breakwaters (Schoonees et al. 2019; Siemes et al. 2020), stone dams (Van Loon-
steensma and Slim 2013) or concrete reefs (Chowdhury et al. 2019). These structures can
attenuate waves and increase soil elevation as a result of sediment trapping in front of the
marshes, thus creating favourable conditions for marsh expansion.

An alternative nature-based approach to change the morphology, width and elevation
of the tidal flats in front of the salt marshes and simultaneously increase ecological value may be
the restoration of adjacent foundation species such as seagrass meadows or shellfish reefs (e.g.
oyster or mussels) (Angelini et al. 2011; van de Koppel et al. 2015; Schoonees et al. 2019)
(Fig. 1). These communities found in the tidal flats can also attenuate waves and stabilise
and/or accrete sediment in the tidal flat (e.g. Meyer et al. 1997; Borsje et al. 2011; Donker et
al. 2013; Walles et al. 2015; Chowdhury et al. 2019). Trials of more environmentally friendly
options, such as the restoration of biogenic reefs (e.g. dominated by oysters or mussels) at a
large scale and at exposed sites remain scarce (Bouma et al. 2014; Morris et al. 2018).
Chapter 7 will investigate the effects of an alternative management measure using
ecoengineering solutions (i.e. biodegradable artificial reefs) on unvegetated intertidal flats to
shape the foreshores by restoring mussel beds, which ultimately may reduce marsh retreat
and/or facilitate marsh expansion.
Main objectives of this thesis

As described above, the main objectives of this thesis are to assess i) the distribution and development of salt marshes in relation to the foreshore bathymetry (chapter 2), ii) the effectiveness of such (vegetated) foreshores in reducing wave run-up onto a seawall (chapter 2), iii) topsoil and lateral erosion resistance across foreshore ecosystems and management types (chapters 3, 4, 5 and 6) and iv) the use of ‘green’ management measures to stabilise tidal flats and thereby facilitate marsh expansion (chapter 7) (Fig. 5). The questions addressed in this thesis are summarized in box 2 and the embedding in the AllRisk project is summarized in box 3.

Fig. 5. Illustration of the parts included in this thesis.
Introduction

Box 2. Thesis outline

Chapter 2
Questions: Are marshes growing where we need them most? Which factors drive differences of run-up and beach wrack levels across different foreshore types?

Through the analysis of vegetation and bathymetry maps of the last two decades, in chapter 2 I study i) at which locations are the salt marshes found in the Dutch Wadden Sea in relation to the foreshore bathymetry and ii) the salt marsh expansion in relation to the sediment dynamics in the adjacent tidal flats. Simultaneously I study how effective are salt marshes with different settings in reducing wave run-up onto the dikes compared to bare tidal flats by conducting field measurements for 3 years.

Chapter 3
Questions: How do different foreshore types resist to fast flow erosion, which could occur during a dike breach?

In chapter 3 I conduct a fast-flow flume experiment to study the resistance of salt marshes and tidal flats with different soil and vegetation properties to fast flow erosion, which could occur during a dike breach.

Chapter 4
Questions: How resistant are soft-sediment ecosystems from a fetch-limited back-barrier island to top and lateral erosion in relation to its management?

In chapter 4 I investigate how the management of a sandy island can affect its stability. I do that by experimentally testing the top and lateral erosion resistance of dunes, marshes and tidal flats soils found in the island, using wave mesocosms and a fast-flow flume. Then, the soil properties and erosion results are related to the past management and development of the island.
Chapter 5

Questions: What is the role of eelgrass on bed-load transport and sediment resuspension under wave conditions?

Using a wave flume, in chapter 5 I experimentally test the role of eelgrass, with and without canopy, on sediment resuspension and top erosion under controlled wave conditions.

Chapter 6

Questions: How does grazing management in combination with abiotic factors, such as marsh age, marsh elevation and sediment layering, affect the susceptibility of marsh edges to lateral erosion?

In chapter 6 I conduct a wave flume experiment to test the lateral erodibility of marsh soils collected in areas with different grazing management, marsh age and marsh elevation.

Chapter 7

Questions: Can we develop nature-based management on the tidal flats to stabilize the marshes?

In chapter 7 I assess the effects of artificial reefs on tidal flat morphology, which may ultimately be beneficial for marsh stability. A large scale experiment is performed were artificial reefs are deployed in a bare tidal flat. The effect of the reefs on the tidal flat is assessed by measuring their wave attenuation capacity, soil elevation changes and soil properties changes on the tidal flat.
Box 3. Embedding in the AllRisk project

This thesis is part of the large integrated multidisciplinary AllRisk project, funded by STW and led by Prof.dr.ir. M. Kok (Delft University of Technology), which employs several PhD’s, post-docs and technicians. The AllRisk project aimed to support the Dutch Flood Protection Program (HWBP) by investigating key knowledge gaps in flood risk assessment and the use of flood defences to reduce this risk. More specifically, the AllRisk project aimed to make improvements in the assessment of flooding probability and in the design measures by i) reducing statistical and model uncertainty by adding new knowledge on hydraulic loads, subsoil characteristics and the strength of flood defences, ii) evaluating measures to reduce hydraulic loads on the dikes or strengthen flood defences (e.g. eco-engineered foreshore measures to reduce waves) and iii) exploring opportunities for other utilization of flood defences in the ‘new flood risk approach’. Another aim of the AllRisk project was to create connections between science (i.e. researchers) and practice (i.e. end users). This imposed questions ranging from legal, governance, engineering to ecology. The research reported in this thesis is the subproject B1 “Uncertain foreshore ecosystem dynamics”, which is part of the project B: Dynamics in hydraulic loads.

The joined NIOZ-RUG subproject B1 was focussed on the ecological part and addressed uncertainties about foreshore ecosystem dynamics of the Wadden Sea and Scheldt Estuary coasts and how to manage the foreshore ecosystems to gain both coastal protection and ecological value. Adequate management of foreshores along the Wadden Sea dikes may safe huge costs of ‘hard’ coastal protection through higher and stronger dikes (Temmerman et al. 2013; van Wesenbeeck et al. 2016). However, uncertainties about the actual effectiveness still hampers the practical implementation of these ecosystem-based measures. The aims of this project were to reduce these uncertainties by providing thorough understanding of coastal ecosystem dynamics in relation to their coastal protection role, as well as investigating nature management strategies in order to optimize both the coastal protection services and the ecological value of these ecosystems.