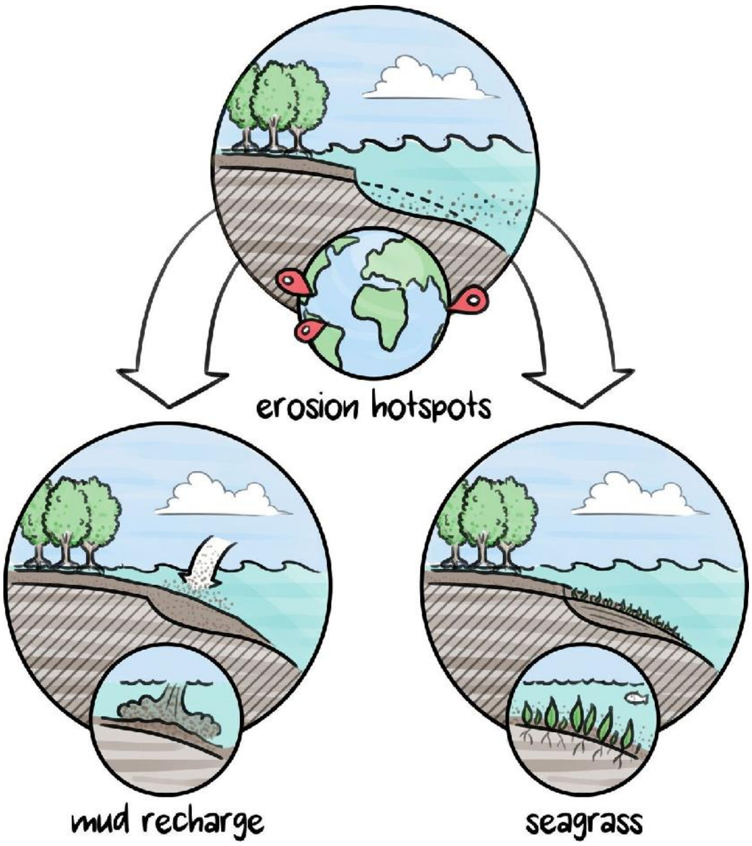
Eroding tropical mud coasts: identifying erosion hotspots and solutions



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Summary

Coastal erosion, whether natural or anthropogenic induced, is a worldwide problem. Coastal areas are however often densely populated, making erosion mitigation solutions essential. However, solution knowledge is limited for tropical mud coasts and these mainly developing countries are characterized by weak coastal management. Traditionally hard solutions were implemented. They can cause biodiversity losses, potentially increasing community vulnerability. Furthermore, the instability of mud forms an engineering problem. This indicates the need for Nature Based Solutions (NBS). Although mud recharge and seagrass are identified in design guidelines as NBS, potential ways and success and failure factors of implementation and efficiency in erosion mitigating are absent. Furthermore, an erosion hotspots overview along tropical mud coasts is lacking.

This study firstly aimed to provide an erosion hotspot overview. Secondly, to evaluate restoration techniques of mud recharge and seagrass for tropical mud coasts, complemented by identifying their erosion mitigation effectiveness and success and failure factors of implementation. Identifying erosion hotspots was done with satellite data, complemented with qualitative literature the determine coastal sediment characteristics. Evaluation of restoration techniques was based on analysis of documented projects, backed up by qualitative and quantitative data from literature.

Erosion along tropical mud coasts occurs most often in Asia and North and South America. More erosion hotspots are found along open coasts compared to bays, estuaries and barrier coasts.

Evaluation of mud recharge schemes indicated direct confined intertidal placement and trickle charge aided in mitigating erosion in estuaries and bays intertidally. Seagrass has been restored with some success in bays, estuaries and barrier coasts. Restoration was achieved with planting subtidal sods or intertidal and/or subtidal anchored rhizome fragments to staples or weights. Erosion mitigation cannot be provided by every species in every location and/or scenario. Findings should be taken with caution due to limited availability of mud recharge and seagrass restoration projects.

Long term survival of seagrass is uncertain due to its vulnerability to anthropogenic threats. However, seagrass is linked to high fishery production. When energetic conditions are not restricting and ample of donor bed is available, choosing anchored rhizome fragments above sods is advised. The first is less labor intensive, costly and impacting to donor sites. Mud recharge is ill-advised in areas with sensitive marine life or commercial activities like shellfishery due to increased levels of turbidity. When chosen, trickle charge is advised above direct placement due to its lower recharge rate.

Cover drawing: Van Ginneken (2017a)

Extended summary

Coastal erosion is a problem observed worldwide. Although it is a natural process, humans directly and indirectly add to coastal erosion. Anthropogenic drivers include construction of upstream dams that deplete coastal sediment supplies. Land use changes causes loss of wave attenuating and sediment binding services of coastal ecosystems. Another is relative sea level rise due to extraction of oil, gas and groundwater and/or anthropogenic induced climate change. Coastal erosion risks are likely to grow with the ongoing trend in climate change trend and growth in human populations. Coastal areas however, are often densely populated, highly urbanized and heavily farmed, making them vulnerable to coastal retreat, indicating the need to find erosion mitigation solutions. However, little is known concerning mitigation solutions for mud coasts, which are predominantly located in tropical countries. These countries are mostly developing countries, characterized by often weak coastal management. Traditionally hard solutions have been implemented. However, implementing hard solutions along mud coasts is difficult due to its weak soil foundation. And wave reflection on hard solutions can enhance erosion even further. Furthermore, hard solution can cause loss of habitat and biodiversity which can lead to increased community vulnerability in developing countries. This indicates the need for Nature Based Solutions (NBS) to mitigate erosion. Although mud recharge and seagrass are identified in design guidelines for NBS for coastal protection, potential ways to implement these NBS on mud coast, the related failure and success factors of implementation and efficiency in mitigating mud erosion are absent. Furthermore, up to date no global overview is available of erosion hotspots along tropical mud coasts to indicate focus areas for coastal erosion management.

This study therefore aimed to provide an overview of erosion hotspots on tropical muddy coasts and evaluate restoration techniques of mud recharge and seagrass along tropical mud coasts. This was completed by analyzing their effectiveness in erosion mitigation and their success and failure factors of implementation. Identifying tropical erosion hotspots was based on 30 years of quantitative data provided by the Aquamonitor. Based on descriptive qualitative literature the sediment characteristic of these hotspots was determined. The severity of erosion was compared based on coastal type and continent. Evaluation of restoration techniques was based on analysis of implemented projects along mud coast, backed up by qualitative and quantitative data from grey and scientific literature.

Erosion along tropical mud coasts appears to occur most often in Asia and North and South America. More erosion hotspots are found along muddy open coasts than muddy bays, estuaries and barrier coasts.

Evaluation of mud recharge schemes indicated that direct confined intertidal placement and trickle charge (both intertidal placement and water column recharge) can aid in mitigating erosion. This potential is seen in estuaries and bays in the intertidal area. To implement seagrass as a successful erosion mitigation solution, implies an intermediate step requiring successful restoration. Seagrass has been restored with some level of success in bays, estuaries and barrier coasts. It can be achieved by planting sods in the subtidal area or anchored rhizome fragments to staples or weights in the inter- or subtidal area. Levels of restoration successes vary widely in time and space, with survival rates ranging between 0-100%. Erosion mitigation services provided are not applicable for every species in every location and/or scenario. Consequently, the potential of introducing seagrass as an erosion mitigation solution remains uncertain. Findings should be taken with caution due to limited availability of mud recharge and seagrass restoration projects found along mud coasts and lack in long term monitoring.

Several success and failure factors of implementation have been identified for different restoration techniques. Reasons of poor seagrass establishment during restoration are numerous and seagrass is vulnerable to external impacts related to anthropogenic threats and climate change. Consequently, long term survival of seagrass is uncertain. When energetic conditions are not restricting and ample of donor bed is available, it is advised to choose anchored rhizome fragments as a restoration technique.

This is less labor intensive, costly and impacting to the donor site compared to sods. Implementation of mud recharge might be related to loss of sediment outside the targeted area which increases turbidity. Consequently, implementation is not advised in areas with sensitive marine life or in proximity of commercial activities like shellfishery. When mud recharge is chosen, trickle charge is thus advised above direct placement due to its lower recharge rate. Removing sediments from a borrow pit for mud recharge schemes might also cause physical and/or ecological impacts. Seagrass species however, do not cause negative effects if the seagrass species are none-invasive. Even more, they are often the dominant primary producers in coastal areas and closely linked with high fisheries production due to the critical nursery habitat they provide.

The main insight in this study is that although strong erosion is occurring most along open mud coasts, mud recharge and seagrass have not been implemented along this coastal type. Due to a lack of experimental data it is unknown if NBS could also be applied along these more energetic coasts.

Content

List of definitions

Agitation dredging	A mud recharge technique in which seabed sediments are brought into suspension over the whole water column to form a recharge source.
Direct placement	A mud recharge technique in which sediments are placed directly on the seabed, either with or without retaining structures.
Erosion	The conversion of land into water because the input of sediment is smaller than the output. Drivers of the sediment deficit can both be natural or anthropogenic.
Coastline	The line between the coast and the shore.
Foreshore	Intertidal zone. Area between mean low and high tide.
Intertidal zone	Foreshore. Area between mean low and high tide.
Inundation	Flooding.
Nature Based Solutions (NBS)	Actively introducing habitats with or without structural engineering, to utilize their dynamic natural processes as part of realizing engineering objectives.
Normal wave base	Water depth beneath which there is no wave movement.
Restoration	Actively introducing habitats. This can apply to returning natural system in areas where it was lost (aka 'rehabilitation') or establishing an ecosystem where it did not occur before (aka 'creation').
Sediment stirring	Implies agitation dredging and water injection as mud recharge techniques.
Shoreline	The mean high water-line.
Subtidal zone	Zone extending seaward from the mean low water line, well beyond the breaker zone. It includes the littoral zone. This is the zone where longshore and across-shore transport occurs.
Turbidity	The degree to which water contains particles that cause backscattering and absorption of light
Trickle charge	A mud recharge technique in which sediments are brought into the natural system to form a slow recharge source for the foreshore.
Water injection	A mud recharge technique in which water is injected into the seabed to fluidize mud. The fluid mud flows on the lower part of the water column, driven by density to form a recharge source elsewhere.

1 Introduction

1.1 Drivers of coastal erosion

Coastlines are dynamic systems, undergoing adjustments of form and process at different time and space scales in response to geomorphological and oceanographical factors (Nicholls, et al., 2007). Daily, seasonal, annual and even longer cycles of natural coastal erosion and accretion affect shorelines worldwide. Consequently, natural coastlines tend to migrate landward and seaward over time, depending on factors like sea level, wave climate and sedimentation (Pinet, 2011).

However, humans directly and indirectly influence coastal processes, adding to coastal erosion (Figure 1.1). Direct anthropogenic influences are induced by human activities in drainage basins and coastal areas. This can alter the natural sediment delivery along coasts (Wong, et al., 2014), adding additional pressure that may dominate over natural processes (Nicholls, et al., 2007). This can increase coastal erosion and the related shoreline retreat. In Asia for example, construction of upstream dams is now seriously depleting the sediment supply to many deltas, causing widespread increased erosion along shorelines (Nicholls, et al., 2007). This reduced sediment flux to the coast due to inland reservoir building has been observed in many rivers over the world, especially in Africa and Asia (Syvitski, et al., 2005). Coastal erosion can also be induced by land use changes. The conversion of tropical and subtropical mangrove forests and temperate saltmarshes for agriculture, aquaculture and industrial and urban development, causes loss of the wave attenuating and sediment binding services these systems provide (Nicholls, et al., 2007). In Thailand for example erosion was caused by large losses of mangrove forests and their related services. This is caused by the establishment of shrimp farms in place of mangroves and over harvesting of trees for timber and charcoal production (Winterwerp, et al., 2005). This large scale coastal erosion due to establishment of aquaculture ponds is observed in several countries in South East Asia and Latin America (Van Wesenbeeck, et al., 2015). Human induced land subsidence caused by extraction of oil, gas and groundwater can also add to erosion due to the related increase in relative sea level (Chu, et al., 2006). This increase in water depth will increase the wave heights, inducing erosion of the former coastal profile until a new dynamic equilibrium is reached in a coastal profile more landward (Paul & Rashid, 2017). For example, over pumping of groundwater in the southwest of Taiwan resulted in land subsidence of 4.3 cm/y causing an 80-meter inland retreat

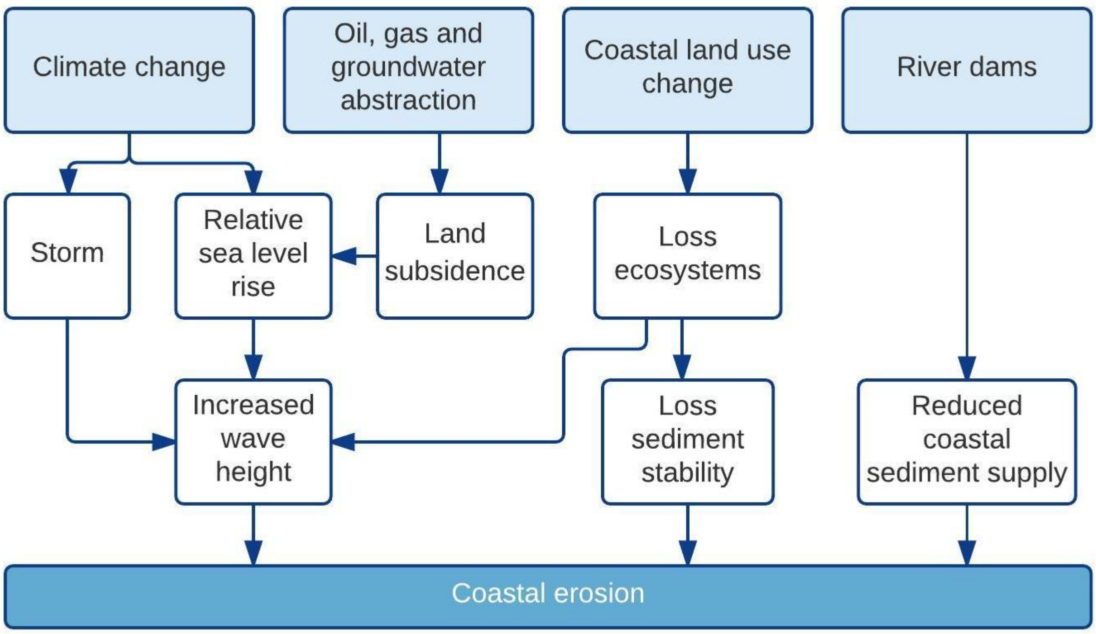


Figure 1.1: Potential anthropogenic drivers of coastal erosion (light blue boxes) with their direct and indirect effects (white boxes) on coastal zones.

of the coastline along Wefong between 1973-1983 (Hsu, et al., 2007). Similarly, in the Po River Delta in Italy coastline retreat was caused by excessive groundwater withdrawal and the related subsidence of 1-3 meter between 1950 and 1970 (Carminati & Martinelli, 2002). Indirect influences on coastal processes include anthropogenic induced climate change and the related increase in sea level and storminess which is expected to further enhance coastal erosion (Paul & Rashid, 2017; Wong, et al., 2014). Depending on the location, all these factors add to increased coastal vulnerability to erosion.

1.2 Problem

1.2.1 Coastal vulnerability

Coastal erosion is a problem observed worldwide (Cai, et al., 2009). It may weaken the shoreline, causing its retreat and potentially increase risks of hinterland flooding (European Commission, 2004). At the same time, coastal areas are often densely populated, highly urbanized and heavily farmed (McGranahan, et al., 2007). Therefore, coastal retreat and/or flooding affect human activities and the associated infrastructure in these zones (Clark, 1995; Li, et al., 2015). For example, it is estimated that the displacement of communities in 2013 due to coastal erosion in Togo (Africa) already caused economic losses of about 2,3% of its countries GDP (FAO, et al., 2016). It is likely that these erosion risks will increase in the future due to climate change and the related increase in sea level rise (Zanuttigh, 2011) and wind waves (Beck, et al., 2014), and due to increasing concentrations of human population and economic activities in coastal areas (McGranahan, et al., 2007). This indicates the necessity for coastal management to identify where strong erosion is occurring and what measures can be implemented to reduce erosion and stabilize coastlines.

1.2.2 Mud versus sand

Despite the importance of finding measures to stabilize coastlines, most knowledge concerning erosion control measures focuses on sandy shores (Saengsupavanich, 2013). Less is known concerning mitigation options for coasts with predominantly muddy sediments. Yet, wave forcing and the related morphological processes are different and more complex on mud shores than on sandy shores (Mehta, 2002). Furthermore, the soil foundation of mud coasts is weak compared to sandy coasts (Silverster & John, 1997; Saengsupavanich, 2013), often forming an important engineering problem (Mehta, 2002). This indicates the importance of identifying potential erosion mitigation measures on mud coasts, since measures implemented on sandy shores are not necessarily successful along mud shores.

1.2.3 Tropical areas

Although mud coasts are characteristics of all continents, they are predominantly found in tropical areas (Wang, et al., 2002). Most of these tropical coasts are developing countries, which are extra vulnerable to coastal risks due to its often weak coastal management framework and lack in financial resources. Although case studies can be found in literature concerning the extent of mud erosion of particular tropical sites (Van Wesenbeeck, et al., 2015), up to date no global overview is available of erosion hotspots along tropical mud coasts.

1.2.4 Traditional hard solutions

Most efforts to protect shorelines have resulted in the construction of 'hard engineering' solutions (Byron, et al., 2011). These 'hard' or 'grey' solutions exclusively include structural features (Pontee, et al., 2016) which can provide direct coastal protection (seawalls and revetments) or indirect protection (groins and breakwaters of various designs) against erosion (Albers, et al., 2013; Pilarczyk, 2005). The construction material is usually rock and concrete (Ibid.). Breakwaters for example are barriers built offshore and parallel to the coastline. They are designed to absorb the pounding of breakers or reflect waves back to the sea (Temmerman, et al., 2013). However, they immobilize across-shore sediment transport towards the coast (Winterwerp, et al., 2005), thus reducing natural accretion. Bulkheads and seawalls are built on the shore with the purpose to prevent coastal erosion caused by storm waves. However, they armor the shore, removing a potential source of sediment further down drift. Also, these walls reflect and redirect wave energy and consequently increase the water turbulence and the related erosion at the foot of the wall. This deepens the nearshore zone and causes storm waves which normally break offshore, to reach the seawall and thereby increase erosion even further (Figure 1.2) (Ibid.). Furthermore, construction of hard solutions on muddy coasts is difficult since the unconsolidated layer of soft mud is generally weak and highly compressible. This forms a stability problem for heavy structures (Kamali & Hashim, 2010). Also, in certain geographical regions natural rock is not present and construction and maintenance costs of these traditional measures is often high, forming a bottleneck for developing countries (Pilarczyk, 2005). Hard solutions turn naturally dynamic coastlines into static ones. Natural coastlines tend to migrate landward and seaward over time, depending on sea level, wave climate, sedimentation and seasons. Hard defenses can restrict coastlines to adapt to rising sea levels (French, 2011). Furthermore, hard structures can increase habitat fragmentation and loss of habitats and biodiversity (Dugan, et al., 2011). This loss of habitat and biodiversity can cause increased community vulnerability, especially for the rural poor, due to the related loss of ecosystem services like food supply, medicinal products, fuel, construction material and protection from natural hazards such as storms and floods (Díaz, et al., 2006). Negative impacts of hard solutions are thus numerous. Consequently, since the 1980's a shift has been observed from traditional 'hard' solutions towards 'softer', eco-friendlier solutions, which can be captured in the term 'Nature Based Solutions' (Pontee, et al., 2016).

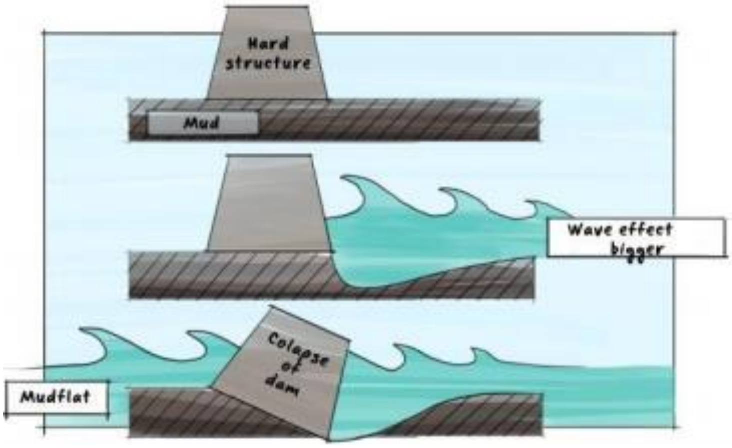


Figure 1.2: Schematic representation of increased erosion caused by hard solution (from Winterwerp, et al.,2014).

1.3 Towards eco-friendlier solutions

1.3.1 Nature Based Solutions

Nature Based Solutions (NBS) consist either wholly or partially of natural features that are designed to offer or improve coastal protection. In the last 5 to 10 years a variety of terms have started to describe these types of solutions including 'Building with Nature', 'Living Shorelines', 'Engineering with Nature', 'Ecological Engineering', 'Green infrastructure', etc. (Pontee, et al., 2016). NBS is a catch all term of these approaches, aiming to use dynamic natural processes and to provide opportunities for nature as part of realizing engineering objectives (De Vriend & Van Koningsveld, 2012). In this paper NBS are defined as either 'managed natural solution' or 'hybrid solutions'. Managed natural solutions imply that a coastal habitat is actively introduced to provide coastal protection services. Potential examples for tropical areas include mangrove forests, sea grasses, oyster reefs and sediment recharge. Hybrid solutions combine a coastal habitat with structural engineering on its landward or seaward side. An example could be a mangrove-levee system (Pontee, et al., 2016). Further distinction can be made between living solutions (mangroves, seagrass, oysters) and non-living solutions (sediment recharge). For more specifics on living solutions and the related concept of ecosystem engineering see Box 1. The advantage of implementing NBS as coastal protections are the co-benefits: implementing NBS can compensate for

> **Definition Nature Based Solutions:**
> *Actively introducing habitats with or without structural engineering, to utilize their dynamic natural processes as part of realizing engineering objectives.*

loss of coastal habitat and provide several ecosystem services like nutrient recycling, carbon sequestration, recreational benefits etc. (Pontee, et al., 2016).

BOX 1 Defining Ecosystem Engineering

Actively introducing habitats with or without structural engineering, to utilize their dynamic natural processes as part of realizing engineering objectives. Living NSB are based on the concept of 'ecosystem engineering'. Ecosystem engineering is based on the ability of ecosystem engineering species to modify their local physical environment because of their structure or activities, beyond their own spatial and temporal scale (Borsje et al., 2011). They physically modify, maintain or create habitats. This change in physical state directly or indirectly changes resource availability to other organisms. Resources can be energy, materials, space, food organisms or a combination of these (Jones et al., 1994). Examples of eco-engineering species include shellfish reefs and submerged vegetation. They have been observed to trap and stabilize sediment in the intertidal zone. This can stimulate soil elevation, resulting in attenuation of waves, reducing erosion. By elevating the soil, vegetation is able to build-up land levels with sea level rise (Borsje et al., 2011), reducing the relative sea level rise along the shore. This can stabilize shorelines and offer coastal protection against erosion.

An overview of potential ways to implement different NBS is missing. Two design guidelines for NBS coastal protection have been developed: the 'Building with Nature' guidelines in the Netherlands and the 'Engineering with Nature' framework of the US Army Corps of Engineering (Pontee, et al., 2016). The USACE 'Engineering with Nature' framework is a tool made for assessing and ranking NBS alternatives alongside other coastal protection measures for the Atlantic coast of the USA (Bridges, et al., 2015). However, it is focused on sandy shores and does not include potential erosion mitigation solutions itself. The 'Building with Nature' framework developed by Deltares includes general design guidelines (Figure 1.3) and an overview of Building with Nature experiences and opportunities for several environments, including tropical coasts (Deltares, n.d.1). It is built on lessons learnt from several pilot experiments, like sediment engines, oyster reefs and wave-attenuating forests (De Vriend, et al., 2014). Although potential NBS for tropical mud coasts (mangroves, sea grass, oysters, sediment recharge) and the related habitat requirements can be found in this framework, it is not structured in a clear overview specifically for this environment and it does not inform the reader of its effectiveness in erosion control. Furthermore, information concerning ways of introduction or implementation in coastal environments is lacking. This is not always straightforward and past attempts have not always been successful (Van Wesenbeeck, et al., 2015; Ondiviela, et al., 2014; National Research Council, 2007; Lipcius, et al., 2015). Potential ways to implement different NBS, the related failure and success factors of implementation of NBS in general and restoration techniques in specific, and their efficiency in mitigating erosion are thus absent in literature. From now on techniques to implement NBS will be referred to as restoration techniques of NBS.

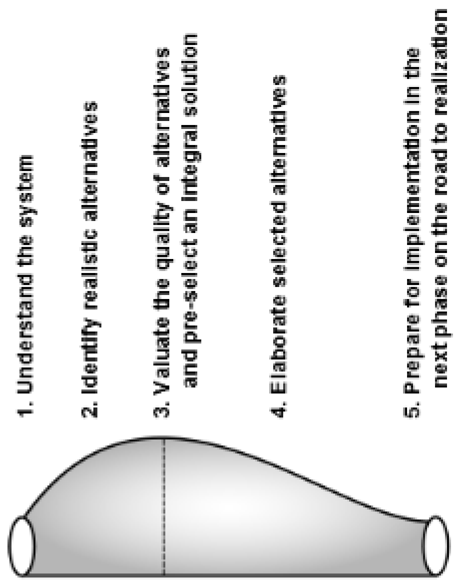


Figure 1.3: Five basic steps for generating Building With Nature design ideas (from Deltares, n.d.2).

1.3.2 Creating a framework

Since no structured overview of potential restoration techniques for NBS for tropical mud coasts was found in the existing literature, this study envisages to create a framework to do so for two NBS. Herein the focus will lie on the two least known NBS: namely, mud recharge as a non-living solution, and seagrass as a living solution. Thus, although mangroves and oysters can be implemented as living NBS, these are excluded further from this study. The generic tool formed in this study will be a starting tool for coastal managers who are creating erosion mitigation plans along specific coastal sites. This will be completed by considering success and failure factors of implementation of the mentioned NBS. Since developing adequate NBS is still an innovative process in which lessons are learned from pilot projects (De Vriend, et al., 2014; Bridges, et al., 2015), this may add to further development of erosion solutions in muddy coasts. Furthermore, this will aid in comparing different restoration techniques to each other to find the optimal solution for coastal managers for their specific location.

1.4 Aim

This study is initiated as part of an internship at Wetlands International, to gain more knowledge in the field of erosion protection along muddy tropical coasts. Wetlands International is an NGO which aims at conserving and restoring wetlands among others to create resilient coastal landscapes in which people are kept safe from coastal threats like erosion and sea level rise. Due to the likely growing erosion risks (Zanuttigh, 2011; McGranahan, et al., 2007), there is a need to identify locations of strong erosion and find eco-friendly erosion mitigation solutions for mud coasts. Especially for tropical areas which are often developing countries with weak coastal management. This study will first provide an overview of coastal erosion hotspots along tropical mud coasts, including both natural and anthropogenic induced erosion. Secondly, a state of the art assessment of potential restoration techniques for the nature based solutions mud recharge and seagrass restoration for tropical muddy coasts will be created. Assuming an executed technique can be implemented in a different location with similar environmental conditions, the question remains under what conditions these solutions can thrive? Furthermore, how do they influence morphological coastal processes to mitigate erosion? And to what extent can they be considered successful in abating erosion? To answer these questions a literature review will be performed in which projects will be analysed which attempted to reduce coastal erosion by implementing different restoration techniques of mud recharge and seagrass. Based on literature the success and failure factors of the two NBS and their related restoration techniques will be identified to aid coastal managers in weighing pros and cons of the two solutions. The findings will aid coastal managers in identifying potential solutions for their muddy coasts. Overall, this study will aid coastal managers in identifying on a global scale the most vulnerable coastal areas where mitigating coastal erosion is of highest importance and give insight in two potential solutions which could be implemented in these eroding tropical mud coasts.

1.5 Research questions

This study will answer the following research questions:

1. Where is erosion on mud coasts occurring strongest in tropical regions?
2. What restoration techniques of mud recharge and seagrass can be implemented to mitigate erosion along tropical mud coasts and how effective are they?
3. What success and failure factors can be identified concerning the implementation and lifespan of the erosion mitigation solutions, and their related restoration techniques, along tropical muddy coasts?

2 Theoretical background

2.1 Research area

2.1.1 Tropical zone

The research area of this study implies muddy coasts located in the tropical zone, as defined by Kelletat et al. (2014) (Figure 2.1). This is a warm to hot and frost-free environment in a belt along the equator. This zone is divided into two subzones: 1) the subtropical zone, which is a permanent humid inner-tropical zone with high precipitation and a very small temperature range throughout the year and between day and night, and 2) an alternating tropical zone, characterised by an alternating dry and humid zone with well-defined seasons caused by a shift in the Intertropical Convergence Zone. Discrimination between the two zones is difficult. However, since tropical cyclones generally impact coastlines poleward of 5° N and S, the inner-tropical zone is much less affected by cyclone related strong swell and wave dynamics. The tropical coastal zone is characterized by the presence of mangroves, sea grasses and coral reefs, although the latter cannot grow on muddy sediments. Grassy marshlands are almost absent (Ibid.).

2.1.2 Muddy coasts

Sedimentary coasts can be classified as consisting of gravel, sand or mud sediments. However, most sedimentary coasts contain sediments of all sizes (Healy, et al., 2002; Kjerfve, et al., 2002). Mud, or cohesive sediments, may be defined as a fluid-sediment mixture consisting of water, sands, silt, clay and organic material in which sediment particle size is smaller than 0,063 mm, with silt and clay particles at a minimum boundary of 0,002 mm. Muddy coasts are defined as land sea transitions which are entirely or in substantial part composed of muddy sediments (Flemming, 2002).

Muddy coasts are formed on low gradient coasts, typically under low wave energy conditions and a high tidal range (Healy, et al., 2002). However, they can also occur in relatively high energy environments (Mehta, 2002) and in micro (< 2 m) and meso (2 – 4 m) tidal range environments (Wang, et al., 2002). Mud can be present in such quantities than entire stretches of coast can be classified as being muddy. An example is the Amazon River dispersal system, northwest of the Amazon River mouth. However, most coasts cannot be classified as entirely muddy. Rather they include sections of coasts, large bays, estuaries, drowned river valleys, lagoons and tidal flats with extensive areas of dominantly mud sediment (Kjerfve, et al., 2002). The typical form of expansive muddy coasts are shallow tidal flats (Healy, et al., 2002).

Muddy coasts can form with rising and falling sea level rise. Muddy coasts can be classified into (1) open muddy coasts, (2) muddy coasts along estuaries and bays, and (3) muddy coasts protected by barriers (Kjerfve, et al., 2002). Examples of each type of coast are depicted in Figure 2.2. In this, back-barrier areas are least exposed to waves, followed by estuaries/bays and consequently open coasts (Daidu, et al., 2013).

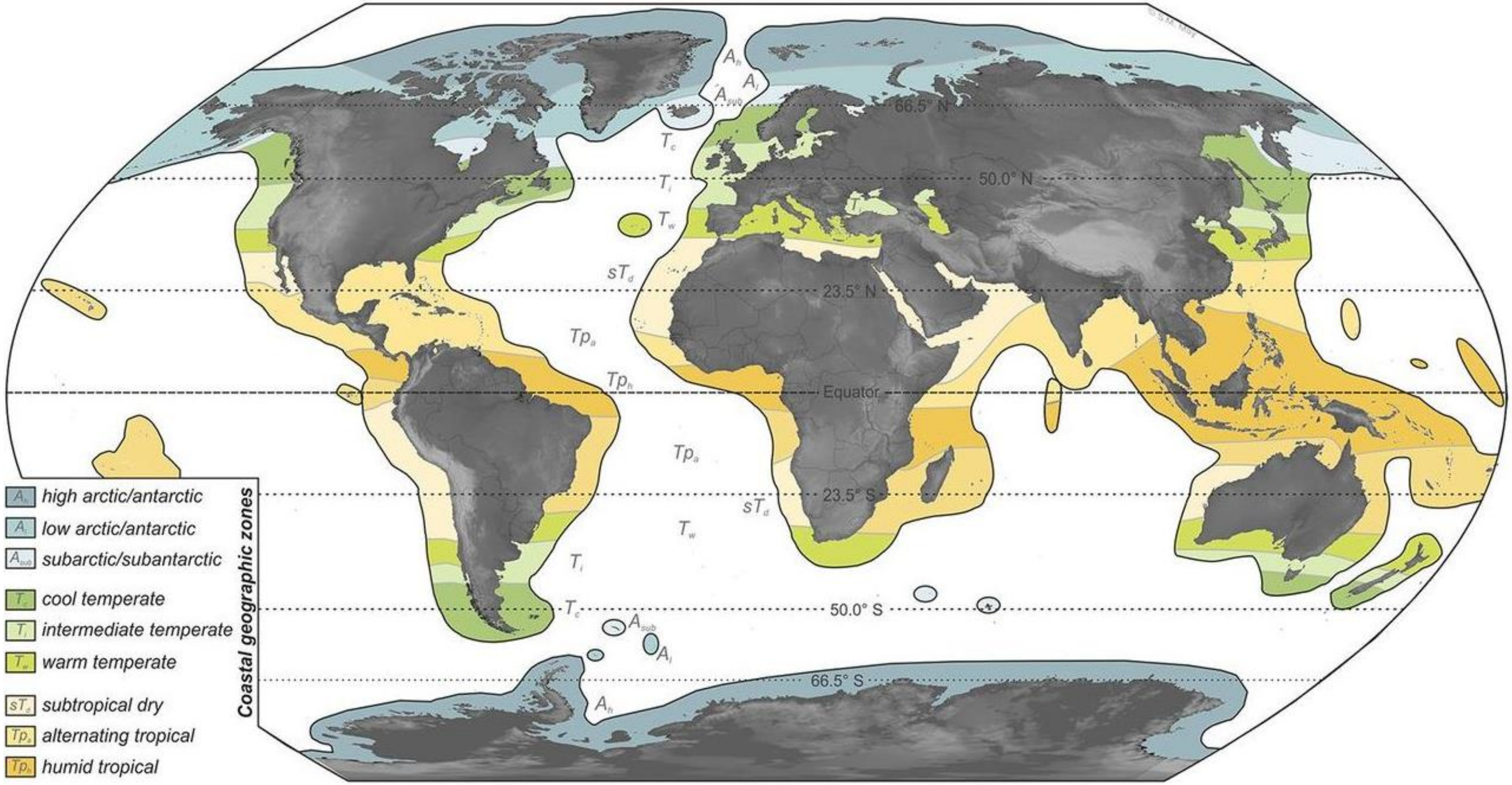


Figure 2.1: Map of geographical coastal zones of the world. Only main islands/ archipelagos are shown in the open oceans. The two dark yellow colors indicate the tropical coastal areas (from Kelletat, et al.,2014).

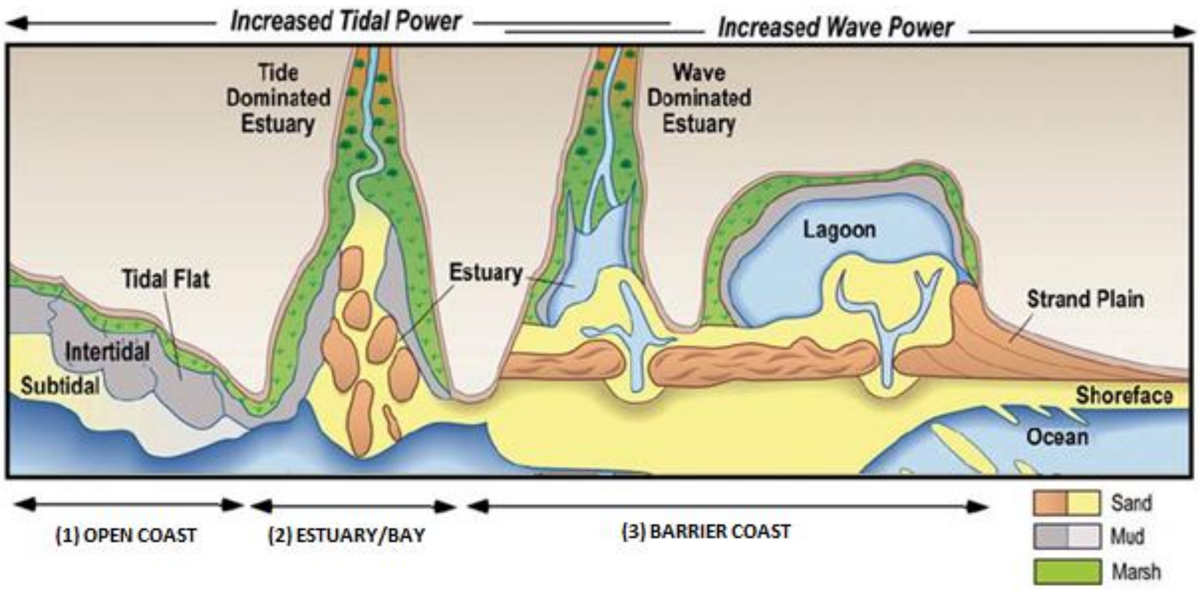


Figure 2.2: Range of common shallow-water depositional systems in a coastal classification on ratio of wave power to tidal power. From left to right examples are shown of a (1) open coast tidal flats, (2) an estuarine coast, and (3) two types of barrier coast: lagoon and estuary (adapted from Steel & Milliken, 2013).

2.2 Coastal erosion processes

Whether a shoreline will retreat or advance depends on the net balance of sediment losses and gains over time, the so-called sediment budget (Figure 2.3). This is governed by sediment transport in the alongshore and across-shore direction. A beach will erode when input of sediment is smaller than output. Input is determined by longshore and onshore transport, output due to longshore and offshore transport. The final sediment budget will indicate erosion, accretion/sedimentation or a steady state of a shore (Pinet, 2011).

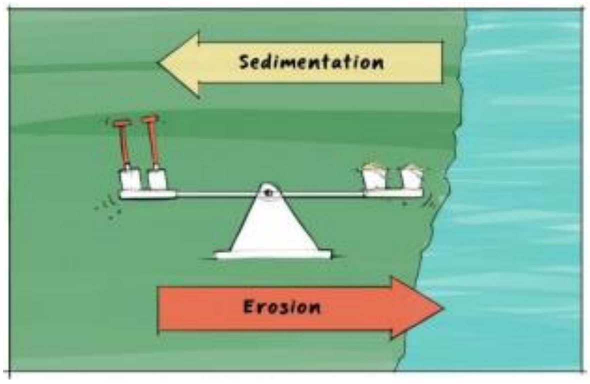


Figure 2.3: Sediment budget (from Winterwerp, et al.,2014).

Definition erosion:
The conversion of land into water because the input of sediment is smaller than the output. Drivers of sediment deficit can both be natural or anthropogenic.

The dominant formation factor of muddy coasts is the presence of a sediment source (Wang, et al., 2002), often river sediments brought in by alongshore currents, and large tidal currents. The strong currents associated with a high tidal range provide a mechanism to transport silt and clay particles towards the shore. At high tide, the tidal currents approach zero, which provides opportunity for the mud particles to settle (Kjerfve, et al., 2002). Thus, as mentioned in the introduction, building a dam in a river will reduce the sediment input of a coast, inducing erosion. This also suggests that inhibiting tidal currents can add to erosion. Sediment erosion is governed by wind waves which introduce energy to coasts (Chinnarasri & Kittirart, 2012) and as mentioned in the introduction, relative sea level rise (Paul & Rashid, 2017; Wong, et al., 2014; Chu, et al., 2006). Storms and storm surges can also add to coastal erosion. However, these events can also accrete coasts and thus overall effects differ per location (Mehta, 2002; Ke & Collens, 2002). Concerning wave erosion, Winterwerp et al. (2005) argues that mainly small waves and not large waves add to erosion on mud coasts; occasionally occurring large waves do not only erode sediments, they also mobilize sediments on the foreshore which can be transported to the coast during rising tide, forming a sediment source. However, continuously occurring smaller waves along the water line only erode mud particles and cannot mobilize sediment on the foreshore (Winterwerp, et al., 2005). Removal of coastal ecosystems also adds to erosion.

3 Research strategy and method

3.1 Structure

The approach to identify erosion hotspots and potential erosion mitigation solutions for tropical mud coasts was divided into two steps. Step one is related to research question 1 (identifying erosion hotspots). Step 2 to research question 2 and 3 (finding solutions). These steps were executed through a literature research on quantitative data and qualitative information. The structure is shown in Figure 3.1. Dutch and English scientific and consultancy reports and Internet sources were found through the search engines Scopus, GoogleScholar and Google.

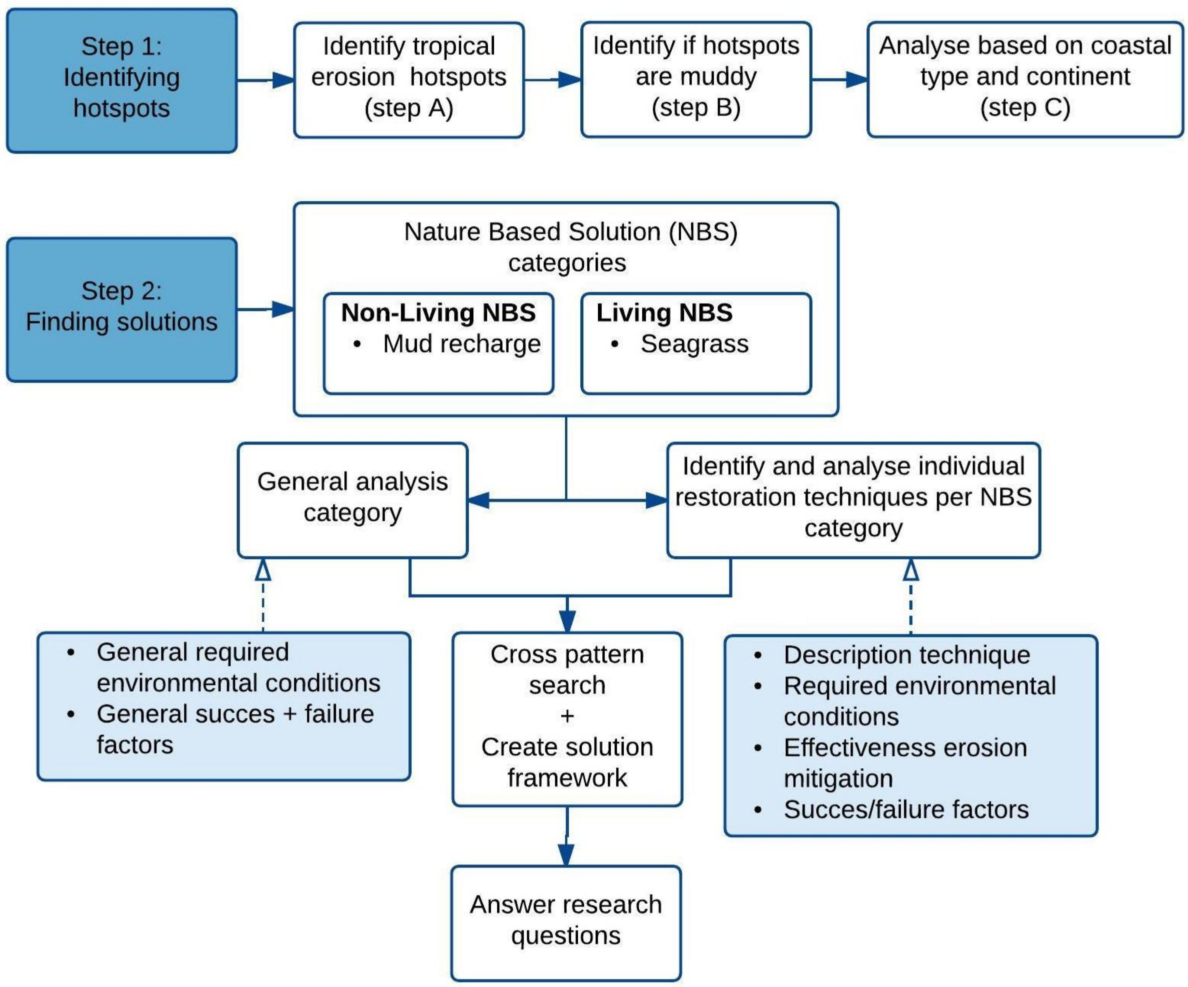


Figure 3.1: Methodology framework. The light blew boxes with the striped arrows indicate the factors analysed of the boxes they feed into. Step 1 is related to research question 1. Step 2 to research question 2 and 3.

3.2 Step 1: Identifying hotspots

3.2.1 Research strategy & data collection

Aim

Step 1 aimed to identify where erosion is occurring strongest along tropical mud coasts, also identified as muddy erosion hotspots. This was done by identifying where strong erosion is occurring in tropical regions (step A) and subsequently determine if the found coastal locations were muddy or not (step B). Due to time constraints, it was chosen not to map the presence of all mud coasts in this region.

Step 1A

First, tropical erosion hotspots were identified. This was done based on quantitative data provided by the Aquamonitor. The Aquamonitor is a global scale tool developed by Deltares, that shows where water is converted to land and vice versa (Donchyts, et al., 2016). It uses Google Earth Engine and Global Land Survey Landsat data to create a full planetary scale view with a spatial resolution of 30 meter from 1985-2015. It is a freely available tool which makes it possible to look at any area of interest on global or local scale. Since drivers of coastal erosion like sea level rise, lack in sediment delivery and subsidence can cause highly nonlinear erosion and accretion, it is assumed that a period of 30 years is long enough to cover a climatological relevant period which allows distinction between a consistent trend and noise of (multi) annual variation (Ibid.).

Erosion hotspots in this tool were identified as coastal land areas which have been converted into water. Indicated in the Aquamonitor by blue areas. The Aquamonitor makes no distinction whether erosion was natural or anthropogenic induced. When the conversion driver is relative sea level rise, then conversion can be caused both by erosion processes and long term inundation. However, it is still a research issue to quantify coastal erosion and the related muddy coast land loss solely upon sea level rise (Wang, et al., 2002). Making a distinction between inundation and erosion is therefore not possible yet. Identification of coastal erosion hotspot was shown when erosion occured more or less continuous for an alongshore distance of at least 10 km or when the coastline had retreated with a minimum distance of 2 km between the year 1985 and 2015. Further specifics are given in Table 3.1. Hotspot identification was done on a scale of 1:1.000.000.

Coastal shape	Erosion hotspot definition

Along a continuous coast (open coast, bay, lagoon, estuary)	• Alongshore
	    ○ A continuous alongshore coastal stretch of minimum distance of 10 km
	    ○ A discontinuous alongshore coastal stretch of more than 10 km, with maximum interruption length of 1 km per gap of no erosion and a minimum total erosional length of 10 km. Accretion does not occur in the gaps.
	• Across shore
	    ○ The coastline has retreated with a minimum distance of 2 km.

Along a discontinuous coast (delta, estuary, bay river mouth)	Hotspots are identified as for a continuous coast, including further specifics as identified below.
	• Alongshore
	    ○ A continuous alongshore coastal stretch of minimum distance of 10 km
	    ○ A discontinuous alongshore coastal stretch of more than 10 km, with maximum interruption length of 1 km per gap of no erosion and a minimum total erosional length of 10 km. Accretion does not occur in the gaps.
	• Across shore
	    ○ The coastline has retreated with a minimum distance of 2 km.
	• When a river/gully or tidal channel enters the sea and on either side of the river erosion occurs alongshore the coastline (with a total minimum length of 10 km) then this is considered as a single hotspot.
	• In case of a non-continuous coastline due to for example the presence of gullies, tidal channels and river mouths, inclusion of erosion locations is only considered which occurs at the boundary between the ocean and the land, with a maximum inland depth of 5 km land inwards.

Table 3.1: Definition of erosion hotspots along different coastal types.

Step 1B

Secondly, identifying whether an erosion hotspot was muddy or not was based on qualitative information from an extensive literature survey of Flemming (2002) concerning the geographic distribution of muddy coasts. This survey includes documented coasts or shorelines along which mud is visibly exposed for a substantial cross-shore width and longshore distance at low tide at the very least. In this a substantial width and distance implies that the mud deposit should form a mapable geological unit at a scale of at least 1:100.000. This excludes numerous small estuarine deposits (Ibid.). The presence of mud is given in a descriptive way.

When necessary (and available), other sources were consulted to complement the review of Flemming (2002) (Appendix 11.2). It should be mentioned that although Flemming's (2002) survey is extensive, it was often not complete. Even with the addition of more recent studies, knowledge was not always available. Beside the fact that it is inevitable that some relevant studies would have been missed and that the sediment character of some coasts may have been misrepresented (Ibid.), the information concerning the presence of mud coasts was not always available or specific enough to clearly determine its exact location. For some countries, only a percentage of muddy coastline was available or the description was vague. For example, when it was stated that 'numerous small estuaries' along a specific coastline are muddy it is unclear whether all estuaries can be considered muddy or not. Other vague terms related to the presence of mud coasts include 'some', 'common', 'north of' etc. Even though this can lead to misinterpretation, it was considered an adequate starting point to point out muddy erosion hotspots on a global level. However, when identifying muddy coasts, an ordinal scale was given to the presence of mud coasts, to indicate these uncertainties (Table 3.2).

Ordinal scale levels	Terms used in literature
High	• Mud is present • Coast is 'extensively' or 'almost continuously' lined by mud
Intermediate	• Numerous/number of/ most coasts in a region are muddy • Vague description of exact mud location or boundaries, but mud is present • More than 66% of a coastal region is considered muddy
Low	• Parts of this coast are muddy • Less than 67% of a coastal region is considered muddy

Table 3.2: Ordinal scale indicating the likeliness of an erosion hotspots of being muddy. No hotspots were identified along coast which indicated more than 66% of the coastal regions was muddy.

This study only indicates the midpoint of the locations of hotspots. The erosion surface is not measured. Of each muddy hotspot, the GPS coordinates of the centre of the erosion area, continent, coastal type and the level of likeliness of a hotspot being muddy is determined. Data is transferred to excel.

3.2.2 Data analysis

Step 1C

Since the erosion surface of the hotspots was not measured, comparison between the extent of erosion of mutual hotspots was not possible. To identify where in tropical regions erosion is occurring strongest the erosion hotspots were analysed based on their coastal type and the continent of occurrence. To do so, the excel data with GPS coordinates, continent, coastal type and mud likeliness per hotspot were transferred to qGIS to make global maps to visualize hotspot distribution. This would give coastal managers a clear overview of the locations of hotspots. Furthermore, to indicate where erosion is occurring strongest, graphs were made in excel to show where the largest number of hotspots were present per continent and per coastal type. This aids in identifying on a global scale which continent is of prime importance when identifying areas to make erosion mitigation plans.

Furthermore, identification of the coastal type where most erosion occurs gives a focus point for finding mitigation solutions which can be implemented along this coastal type. Further research on these solutions should be key focus.

3.3 Step 2: Finding solutions

3.3.1 Research strategy & data collection

Step 2 aimed at finding solutions for muddy coastal erosion and identifying their success and failure factors. To do so, a literature review was performed based on grey and scientific literature. The research strategy of this step was based on Eisenhardt's framework on how to build theory from case studies (1989). To build theory from case study research, the following steps were completed. The steps are loosely based on Eisenhardt's framework (1989);

A. Case study analysis
B. Cross case pattern search + create solution framework
C. Add to literature

In this study, case studies were identified as different restoration techniques, since often the information available for these solutions is derived from implemented projects or 'cases'. They were selected based on theoretical or stratified sampling. This implies that solutions were chosen that are likely to replicate or add to emergent theory, based on different categories (Eisenhardt, 1989). Two solution categories are chosen of each type of NBS:

- *seagrass beds* as a living NBS
- *mud recharge* as a non-living NBS

Several restoration techniques were analysed of each solution category. To identify NBS restoration techniques for each category, documents were consulted which described the implementation and effects of measures in tropical muddy environments. However, when the techniques were not available (yet) for tropical areas, lessons could still be learned from projects implemented along mud coasts in other climate zones. Documentation of implemented projects was used to indicate whether different techniques could aid in mitigating erosion and under which environmental conditions they could be implemented. Only projects were selected which mention the soil type as being muddy or the percentage of mud of the soil being higher than 70%. Of the selected projects, implementation of the techniques had to be finished to obtain information concerning the endurance of implementation. Data concerning the erosion mitigation potential was based on qualitative and quantitative information, depending on availability. Grey and scientific literature was used to find success and failure factors of implementation of different NBS and their related techniques. Appendix 11.1 indicates the search terms entered in different online search engines. In addition, the reference list of studies was scanned and checked to find relevant other studies. Because this study only aimed to give a general overview of potential solutions, specific species names of each NBS category are not used as search terms. Furthermore, machinery needed to execute different restoration techniques was not discussed in this study.

It should be noted that in this study the term 'restoration' is given a broader definition than in literature, covering both ecosystem 'rehabilitation' and 'creation'. According to Seddon (2004) 'restoration' refers to restoring degraded (existing) ecosystems to it pre-existing state. 'Rehabilitation' refers to returning seagrass beds in areas where it was lost and 'creation' to establishing an ecosystem where it did not occur before (Table 3.3). The latter two, 'rehabilitation' and 'creation', overlap the definition of NBS of actively introducing a habitat. However, in documentation these terms of are all used intertwined and/or not always clearly defined. Making it difficult to distinguish them. Since restoration is the most used term in literature, this term has been chosen to cover all possible implementation techniques of NBS.

Seagrass restoration terminology	Definition
'restoration'	Returning a seagrass meadow to its pre-existing condition (i.e. same species composition, distribution, abundance and ecosystem function).
'rehabilitation'	A more general term and implies returning seagrass to an area where seagrass meadows previously existed (but not necessarily the same species, abundance or equivalent ecosystems function).
'Meadow creation'	Implies the establishment of seagrass meadow in an area that has not previously been known to support seagrass.

Table 3.3: Seagrass restoration terminology according to Seddon (2004).

It is also important to note that this study only included on site erosion control solutions. However, the origin of erosion can be located in the drainage basin, and not necessarily along coasts. For example, when oil, gas or groundwater abstraction is the erosion driver, this should be stopped and solutions should be found to reduce subsidence. In case of groundwater over pumping this could for example be achieved through artificial recharge of aquifers (Xue, et al., 2005). Or when upstream dams are the cause, this could imply finding solutions to remove sediments from these reservoirs to bring them back in the downstream river like described by Jokiel & Detering (n.d). This indicates the need to address root causes of erosion beyond on site erosion control solution to create integrated management plans. However, these solutions on drainage basin level are beyond this studies scope.

3.3.2 Data analysis

This study analysed both the restoration techniques (or 'cases') and the NBS categories. This embeds the techniques in a larger context and made overall comparison possible. Both were done in a descriptive way, backed up by quantitative data concerning erosion mitigation when available.

Before analysing the individual techniques, a general description will be given concerning each category. Including the general required environmental conditions in which the restoration techniques are set in. Determining what restoration techniques can be implemented along tropical mud coasts, does not only imply identifying possibilities, but also gaining insight in the potential environmental factors restraining or enabling the implementation of these solutions. A certain technique might not be suitable everywhere. The factors analysed to answer research question 2 and 3 are depicted by the light blue boxes in Figure 3.1. Table 3.4 describes further operationalization for three factors.

Consequently, similarity and differences of the techniques were compared to see if cross case patterns are present within categories and between categories (Eisenhardt, 1989). Cross case patterns within the two NBS categories were described in Section 5.5 for mud recharge techniques and Section 6.3.4 for seagrass techniques. Cross patterns between NBS were described in Section 7. Section 7 also includes solution frameworks which give overview of potential restoration techniques for different coastal types. Based on these emerging patterns and the solution framework the research questions can be answered.

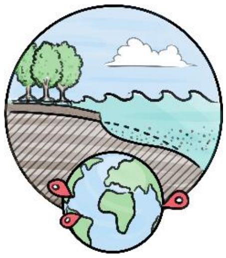
Variables	Operationalization
Environmental conditions	• The necessary environmental conditions in which different solutions can reside, are in this study related to three factors: 1) type of muddy coast (Section 2.1.2), 2) where in the across-shore direction of the coastal profile the solution can be located: in the intertidal or subtidal zone, and 3) the necessary habitat requirements.
Effectiveness of erosion mitigation	• The effectiveness of erosion mitigation of a restoration technique is operationalized in five factors: 1) reduced shoreline or coastline retreat 2) onshore wave attenuation translated into reduced wave height or wave energy 3) reduced current velocity (only applicable to living NBS) 4) bed stabilization (only applicable to living NBS) and 5) deposition of sediments causing increased soil elevation. The latter two add to reducing relative sea level rise and wave attenuation (Shepard, et al., 2011; Ondiviela, et al., 2014). The influence of the categories and their related restoration techniques on the morphodynamical system will be visualized schematically.
Success and failure factors	• Success and failure factors can be technical (for example life span of the technique, maintenance, local availability of material etc.), social (for example acceptance by the community), economical (average costs) or environmental (for example ecological effects or pollution).

Table 3.4: Operationalization of variables related to the different restoration techniques of mud recharge and seagrass. These variables will be analysed to answer research question 2 and 3.

4 Erosion hotspots

4.1 Results

4.1.1 Erosion hotspots



Globally, 129 tropical erosion hotspots have been identified which might be located along mud coasts as depicted in Figure 4.1 and Figure 4.2. Coastal regions which show more than 4 erosion hotspots in proximity of each other have been zoomed into and shown as separate maps. In the figures an ordinal scale has been assigned to indicate the uncertainty in the coastal area being muddy based on the terms used in literature (as explained in the method in Table 3.2).

However, only 112 of the 129 erosion hotspots are considered in this study. This is because it appeared that satellite data in the Aquamonitor was not available for every coast for the period 1985-2015. For 40 of the 129 hotspots data was only available from 1990 or 1992 onwards. For 7 hotspots along the Brazilian open coast, this was only from 2000. These 7 hotspots, together with 10 other hotspots which are identified based on data from 1990 and 1992 onwards, are located along the South American coast between the Amazon river and Orinoco river in Venezuala (Figure 4.1, red hotspots in map 3). This coast is characterized by alongshore migrating mud banks of up to 5 m-thick, 10 to 60 km-long and 20 to 30 km-wide. Source is the huge suspended sediment discharge from the Amazon river. As the banks migrate alongshore, their interaction with waves results in complex and fluctuating shorelines that are associated with space- and time-varying coastal accretion phases and erosional 'inter-bank' phases (Anthony, et al., 2010). The mudbanks migrate at a rate of about 0,5 to 5 km/year, depending on the angle of the coastline to the direction of trade winds. Along the Guyana coast a roughly 30 year period of mud bank accretion was followed by a 30 year coastal erosion process (Eisma, 1998). For the Surinam's coast an approximately 15-20 year period of accretion is followed by a 15-year period coastal erosion (LievenseCSO, 2017; Winterwerp, et al., 2013). Mudbank migration rates along the French Guyana coast have been shown to vary (Gardel, 2005). However, along this coast some areas experienced erosion of more than 2 km due to mudbank migration and others prograded over more than 3 km in 20 years (Gratiot, et al., 2008). Consequently, the 17 hotspots identified between the Amazon and Orinico river cannot be considered reliable based on 15 to 25 years of satellite data. They are thus excluded further from this result section. Consequently, only 112 of the 129 hotspots are considered in this study. According to Anthony et al. (2010) this mud-bank system between the Amazon and Orinico river is unique in terms of magnitude due to the extreme large mud supply from the Amazon river. Mud migrating bank systems however do exist on other mud-rich coasts, such as the West African coast between Guinea–Bissau and Sierra Leone and the Gulf of Papua (map 3 in Figure 4.2). They are however much smaller and it is unknown if these banks influence the location of the shoreline. Furthermore, the 6 hotspots found on these two coasts are identified based on 30 years of satellite data. The reliability of the presence of these hotspots is therefore considered adequate. It is assumed that the other 30 hotspots which are identified based on 23 to 25 years of satellite data still represent a long enough period to indicate long term erosional processes.

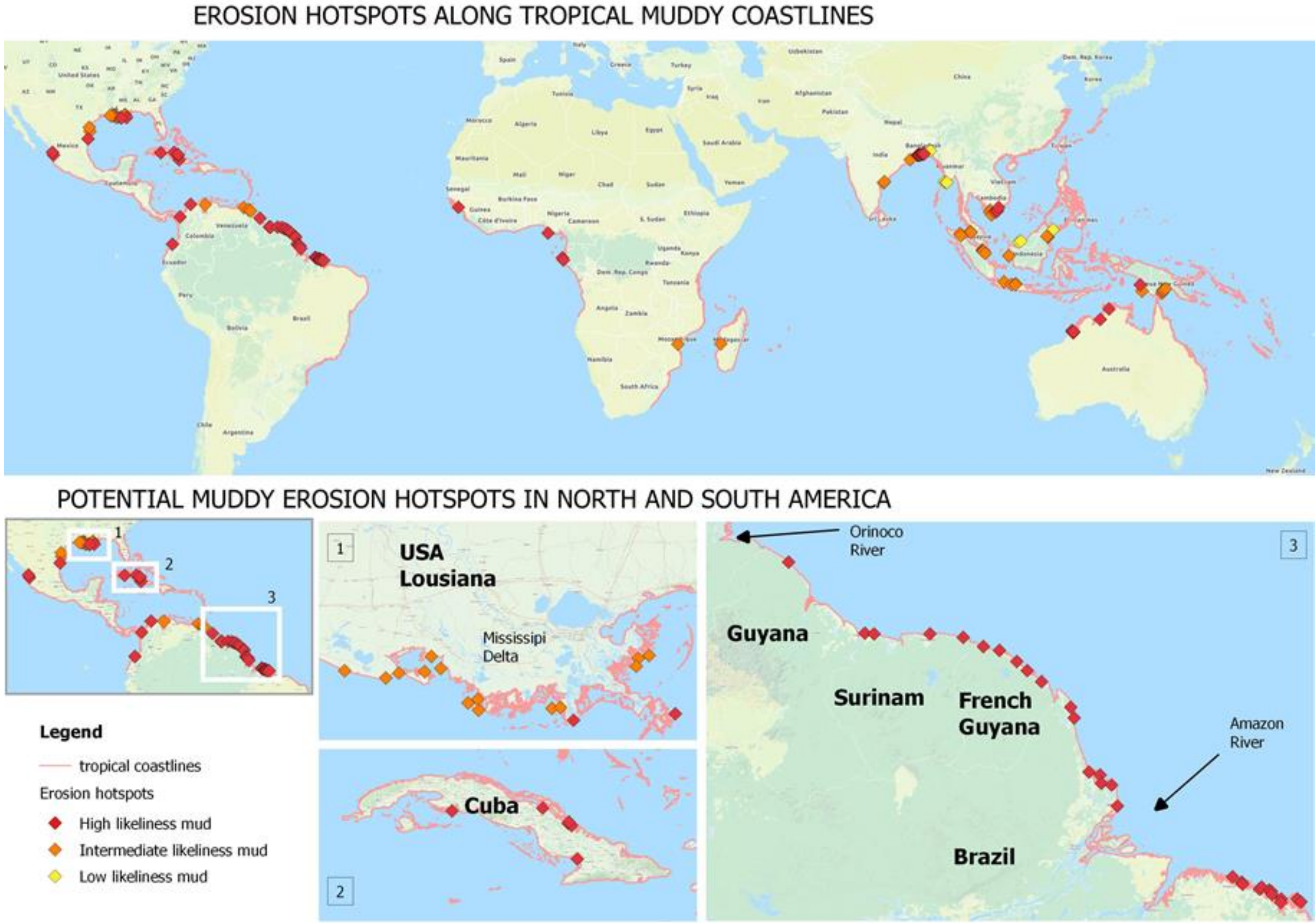


Figure 4.1: Global overview of muddy erosion hotspots, with inclusion of several zoomed in maps of the north and south American coasts. Hotspots are identified based on 23 to 30 years of satellite data. Exception are the 17 red hotspots along the northern South-American coast between the Orinoco and the Amazon river, shown in map 3. They are considered unreliable due to the limited period of satellite data available of this coast (Section 4.1.1).

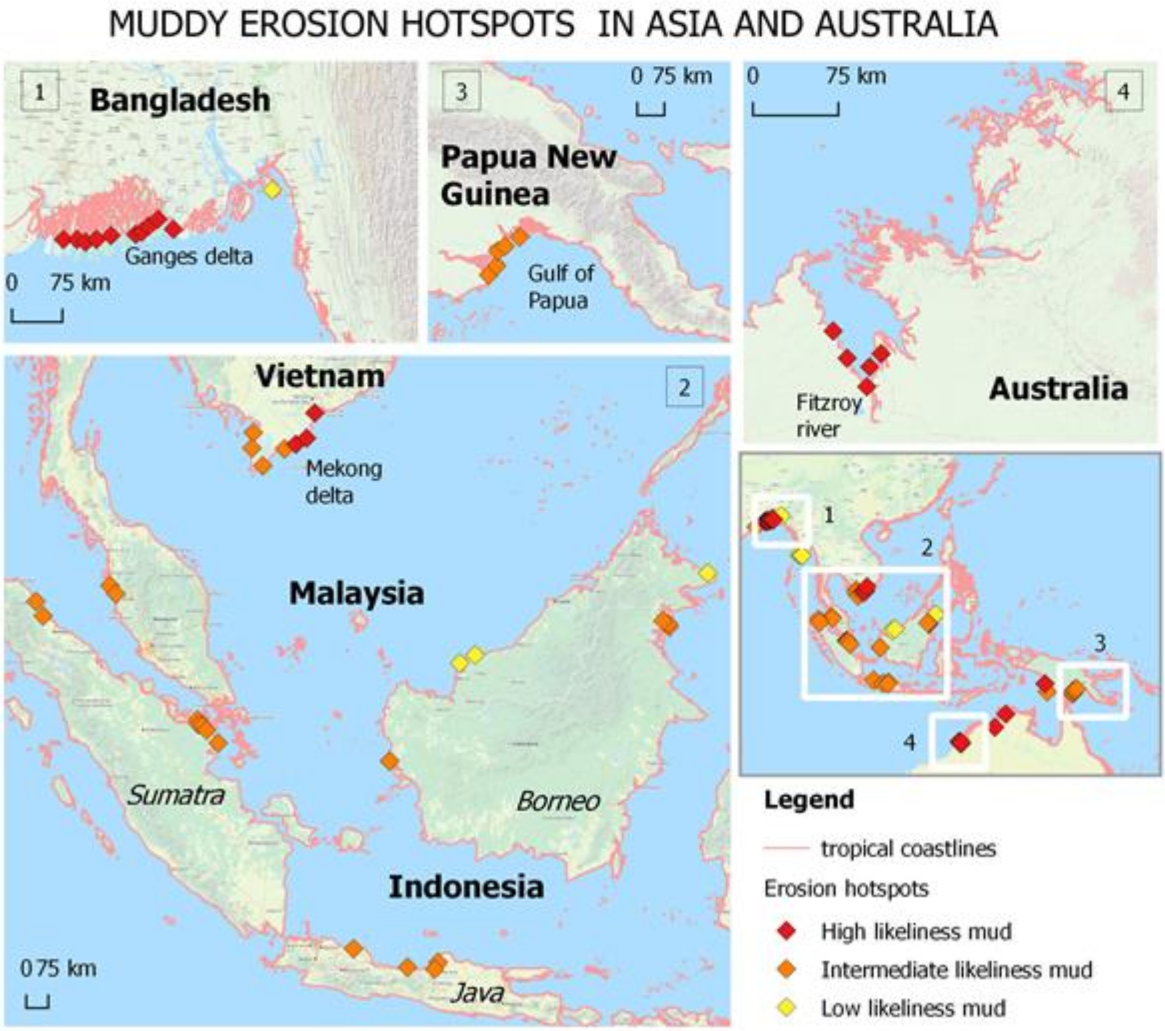


Figure 4.2: Some locations in Asia and Australia with several muddy erosion hotspots. Hotspots are identified based on 23 to 30 years of satellite data.

4.1.2 Division per continent

From Figure 4.1 and Table 4.1 it becomes clear that most erosion hotspots are in Asia, followed by North-America and South-America. Although Asia has the highest number of erosion hotspots (52), only 27% has a high likeliness of being on a mud coast and 60% an intermediate likeliness. This uncertainty is largely explained by the lack of detailed documentation concerning the sedimentology of Indonesia (Flemming, 2002). After Asia, most hotspots are in North-America. Of the 27 hotspots in North-America, it is very likely that 40% is located along mud coasts, while 52% pertain intermediate likeliness, among others due to the vague description of exact location of the presence of mud along the Mississippi delta. Although less hotspots have been observed in South-America (19), it is very likely that 78% of these hotspots are located along mud coasts. This high percentage is largely explained due to the presence of the well documented world's longest muddy coastline along the north-east coast of South-America. This coast is shaped by sedimentation processes directed by the muddy Amazon River (Flemming, 2002). Considering the number of hotspots and the uncertainty in the presence of coasts being muddy, it can thus be stated with high likeliness that erosion is occurring strongest along the South-American (15 hotspots) and Asian (14) coast. Followed closely by the North-American (11) and Australian (8) coast. Differences in the number of hotspot is thus not large. However, when hotspots are included that have an intermediate likeliness of being muddy, Asia clearly stands out (45 hotspots), followed by the tropical North-America (27) and South-America (19). To avoid bias, it's important to mention that these results only give an indication of the number of hotspots and do not indicate the total areal extent of erosion. Looking at the latter factor might give different results.

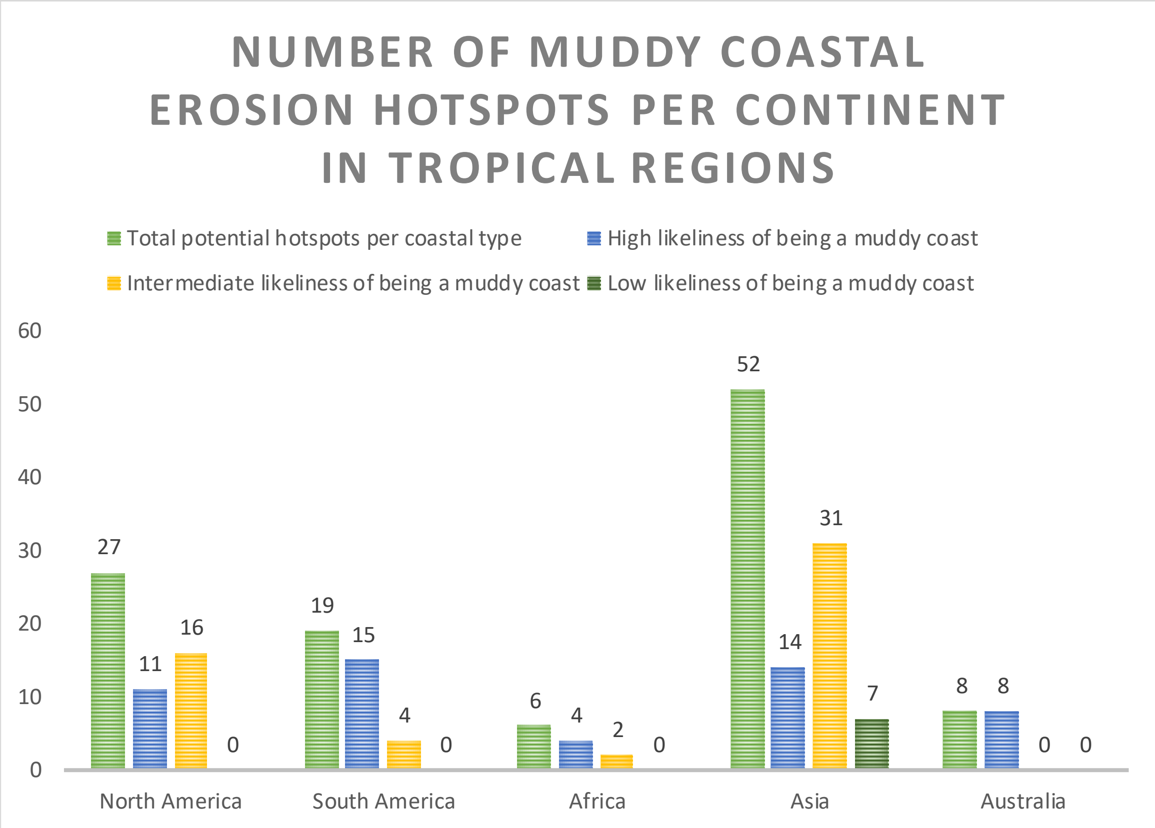


Table 4.1: Number of erosion hotspots found in tropical regions per continent, in total and per category of the likeliness of a hotspot being muddy. (hotspots are defined as a coastline which retreated with a minimum distance of 2 km or an alongshore coastal erosional stretch of minimum 10 km in total length, with maximum alongshore gaps of no erosion of 1 km. For further specifics see Table 3.1).

4.1.3 Division among coastal types

When looking along what type of muddy coasts erosion hotspots are located (Table 4.2), it becomes obvious that most hotspots are located along open coasts (66 of 112) and only 27 in bays or estuaries and 13 within barrier coasts. This trend remains when only the hotspots are considered with high likeliness of being muddy, either with or without inclusion of hotspots with intermediate likeliness of being muddy. Furthermore, from Figure 4.1 and Figure 4.2, it can be observed that coastal regions which show several hotspots (more than 4) in immediacy of each other are often located in the proximity of a river or it's influential area. Regions include the Mississippi Delta in the south of the USA (Figure 4.1, map 1), the South American coast west of the Amazon river (from Brazil to Venezuela, Figure 4.1 , map 3), the Ganges delta in Bangladesh (Figure 4.2, map 1), the Mekong delta in Vietnam (Figure 4.2, map 2), a delta along the north-west coast of the Gulf of Papua, formed by several rivers, in the south of Papua New Guinea (Figure 4.2, map 3) and the estuary of the Fitzroy river in the north-west of Australia (Figure 4.2, map 4). Several other more isolated hotspots also seem to be located in the proximity of a river. To name a few: the four hotspots in Malaysia, the four most southern hotspots of Borneo (Indonesia), 2 of the three hotspots in Colombia, the hotspots in Gabon and Guinea etc. This seems to suggest that strong erosion in tropical regions is often located in the proximity of a river. However, this is biased since mud coasts are often located in the proximity of a river. This can be explained since the largest extent of muddy coastal deposits is formed by a high riverine sediment delivery (Kjerfve, et al., 2002). This presence of a continuous abundant sediment supply is the most important formation factor of muddy coasts (Wang, et al., 2002).

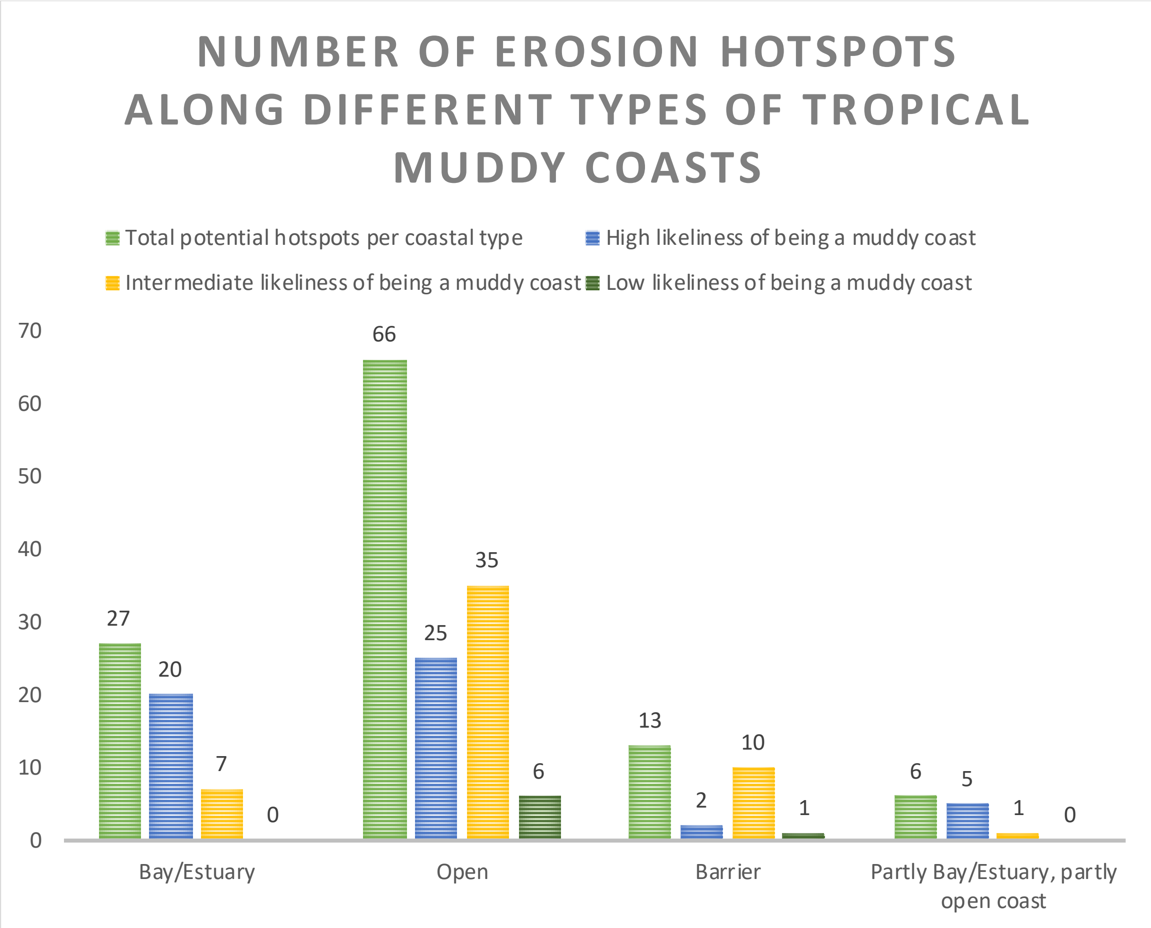


Table 4.2: Number of erosion hotspots found in tropical regions per coastal type, in total and per category of the likeliness of a hotspot being muddy. (hotspots are defined as a coastline which retreated with a minimum distance of 2 km or an alongshore coastal erosional stretch of minimum 10 km in total length, with maximum alongshore gaps of no erosion of 1 km. For further specifics see Table 3.1).

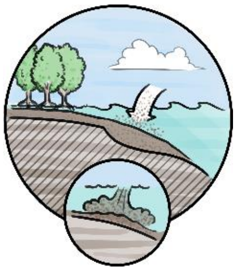
4.2 Conclusion

Strong erosion along tropical mud coasts appears to occur more often along muddy open coasts then muddy bays, estuaries and barrier coasts. When looking on a global scale, it can be stated that the number of erosion hotspots which pertain a high likeliness of being muddy, are similar between continents. Most erosion hotspots occur along the South-American and Asian coasts. Closely followed by North-America and Australia. However, when erosion hotspots are included with intermediate likeliness of being located along a mud coast, then Asia clearly stands out. Followed by South-America and subsequent North-America. Only a few erosion hotspots have been observed in Africa and Australia.

Mudflat along mangrove coast (Wetlands International, 2017a)

5 Mud recharge

5.1 Introduction



Mud recharge can be used to create/restore intertidal mudflats, wetlands and saltmarshes (Burt, 1996) to combat their erosion (Park & Lee, 2007). Recharging mud flats can be done with different sediment types. In terms of physical processes, placing sands and gravel on mudflats can decrease resuspension of muddy material and hence the mobility of the foreshore profile. Such schemes have been successful in reducing erosion along saltmarshes (Kirby, 1995). However, this might have negative environmental effects. From an ecological point of view, it is best to use material with similar grain size to the existing mudflat. Using coarse sediment can cause the replacement of the high biomass benthic communities typical of muddy habitats with lower biomass, higher diversity communities associated with coarser sediments. This change can reduce the food supply for birds and fish of which mudflats form an important feeding ground (Kirby, 1995). Therefore, recharge of mudflats with primarily sands and gravel material will not be considered in this study. However, it is more complex to replenish with mud then with sand because fine sediments take longer to reach equilibrium due to complex de-watering and consolidation processes (Atkinson, et al., 2001).

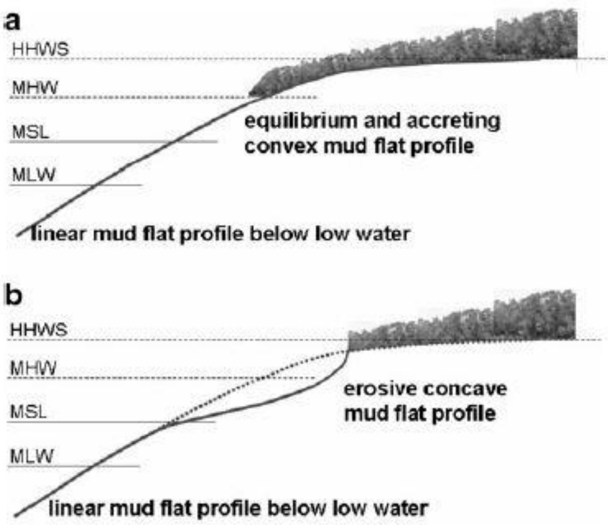


Figure 5.1: Convex and concave cross shore slope of the foreshore (from Winterwerp, et al., 2013).

Although sediment recharge with muddy material can be done in different ways, the overall goal is the same. Eroding muddy tidal flats tend to have low and concave cross section such that large waves reach close to the high waterline, inducing rapid coastal erosion. Creating a high level tidal flat to restore the convex profile of the flat will attenuate waves (Figure 5.1) (Kirby, 2013; Winterwerp, et al., 2013), thereby reducing wave energy (Bray, 2008) and subsequent erosion. Depending on where the added sediments are placed or settle, they can thus mitigate erosion by elevating the bed and/or reduce shoreline retreat. Recharge of sediment impoverished coasts can be done through direct placement, trickle charge and sediment stirring (Figure 5.2) (Fletcher, 2008). These three groups of techniques all pertain to different sub techniques.

All techniques are described in the following sections. Since no general information was found related to the required environmental conditions and the success and failure factors of mud recharge, these factors will only be discussed per category or technique. Techniques of direct placement are discussed in Section 5.2, trickle charge in Section 5.3 and sediment stirring in 5.4. All three sections include description of environmental conditions in which the techniques can be placed, their potential in mitigating erosion and their success and failure factors of implementation. Section 5.5 will compare the different techniques followed by a conclusion in Section 5.6.

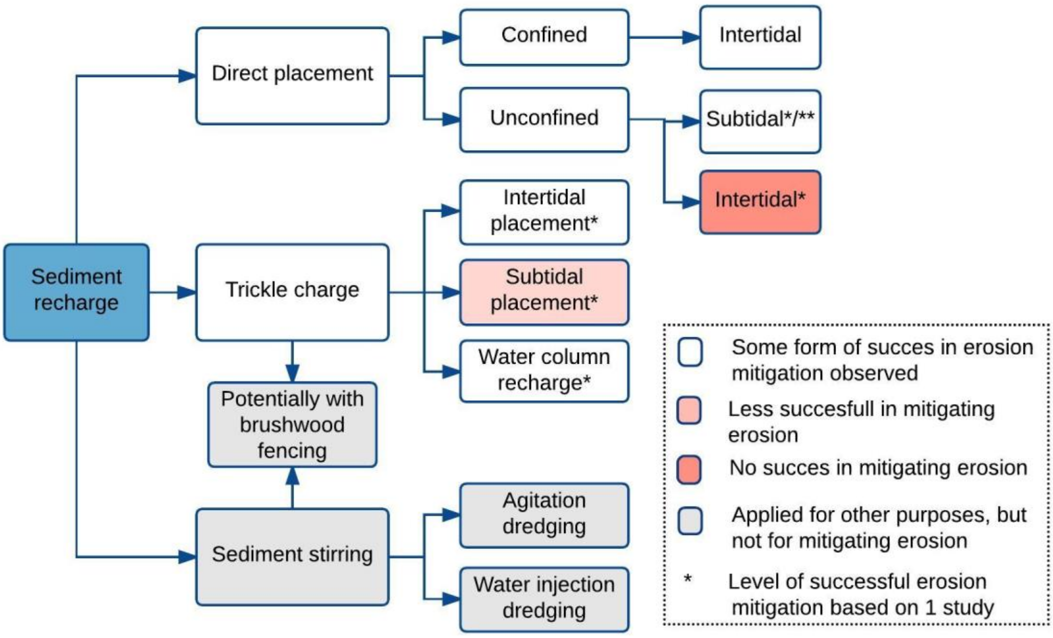


*Figure 5.2: Potential muddy sediment recharge techniques to mitigate coastal erosion. Light red= subtidal trickle charge has been observed to be less successful in elevating the bed than water column recharge. **=wave attenuation observed. However, it is not specified if this also referred to onshore small waves which mainly induce erosion.*

5.2 Direct placement

5.2.1 Technique description

Direct placement implies placing muddy sediments directly onto the seabed to raise the elevation relative to the tidal frame and/or to increase the lateral extent of mudflats (Foster, et al., 2013), either with construction of retaining structures (confined placement) or without (unconfined placement). According to Fletcher et al. (2000) direct placement can both be intertidal or subtidal. Direct placement is also referred to as recharge, replenishment or habitat creation/restoration. Retaining structures can either be soft structures that are likely to become mobile in the longer term, like for example gravel, sand or clay berms, or be more permanent structure that are in principle immobile (Fletcher, et al., 2000; Schratzberger, et al., 2006). Examples of implemented immobile structures include silt curtains, brushwood fencing, straw or coconut matting, steel sheet piling together with rubble mound

Definition direct placement: *Placing sediments directly on the seabed, either with or without retaining structures.*

breakwaters (ABP Research, 1998) or geotextile tubes placed on sand foundations covered with geotextile anchored scour blankets (Colenutt, 1999c). However, no documentation or comparative studies related to factors like the effectiveness, durability and costs of these retaining structures were available.

Figure 5.3 schematically shows how the different techniques could influence hydrodynamic and morphological processes. The retaining structures used with confined intertidal placement dampen waves. The area landward of these structures thus becomes less energetic, and consequently placed sediments can settle. This bed elevation can alter the erosional profile of the mudflat into an accretional profile (Burt, 1996). The heightened mudflat bed that forms with confined and unconfined intertidal placement might dampen waves and thereby reduce coastal erosion. Unconfined subtidal placement has been observed to dampen waves. Sediments might be able to settle landward of the subtidal placement due to the reduced energetic conditions (Ibid.)

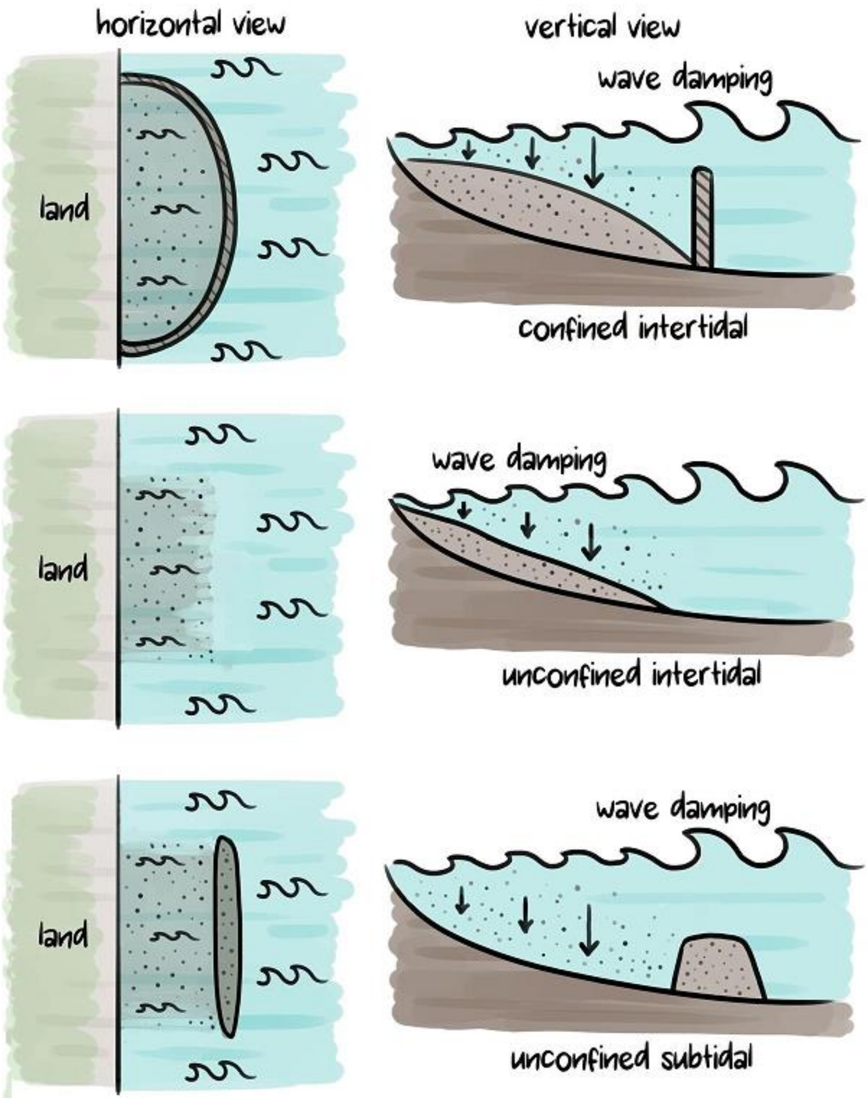


Figure 5.3: Conceptualized influence of 3 direct placement techniques on hydrological and morphological processes along coasts: confined intertidal, unconfined intertidal and unconfined subtidal mud recharge (Illustrated by Van Ginneken, 2017b).

Confined intertidal recharge can also aid in the development of wetlands by elevating the bed level. A small number of experimental recharge schemes for salt marsh restoration schemes with fine material have been undertaken in the UK (ABP Research, 1998). Over time saltmarshes have been shown to establish naturally (Hamer, 2007; Fletcher, et al., 2000).

5.2.2 Environmental conditions

Across-shore

Confined direct placement can both be intertidal and unconfined placement both subtidal and intertidal (Fletcher, et al., 2000). Confined intertidal placement seems to be performed the most (Appendix 11.2).

Habitat requirements / coastal type

Based on the limited amount of documentation available, direct placement seems to be more suitable in less energetic environments. For example, Burt (1996) indicates that unconfined subtidal mud berms can only be placed in areas with moderate wave action and weak tidal currents to avoid erosion. Sediment losses may be high when placed in areas with high currents. Also, a study performed by French & Burningham (2009) showed that confined intertidal recharge of cohesive material can restore degrading mud flats and saltmarshes on low wave energy estuarine foreshores. Furthermore, all implemented recharge schemes found in literature (8 in total) are in estuaries and bays (Appendix 11.3). Examples of direct recharge schemes in more energetic environments were not found.

5.2.3 Erosion mitigation

Direct placement alters the morphology, for example through extending the area of intertidal habitat (Fletcher, et al., 2000). Based on Appendix 11.3, Figure 5.4 shows whether different types of direct placement schemes have been observed to mitigate erosion by attenuating waves, elevating the bed and/or reducing coastline retreat. It is based on both quantitative and qualitative data from fieldwork and descriptive documentation. As can been seen, limited number of implemented projects were found. The implemented projects on which Figure 5.4 is based are described in Box 2.

Results suggests confined intertidal recharge can aid in mitigation erosion by elevating the bed, attenuate waves and potentially reduce coastline retreat. However, the latter service has only been observed once. Fieldwork data indicated a bed elevation of 1,1 m of an intertidal recharge scheme in the Orwell estuary (UK). After 2,5 years the mudflat was still stable (French & Burningham, 2009). Another confined intertidal recharge project at Lymington Estuary (UK) measured bed elevations between 3 to 29 cm depending on the location on the mudflat. The mudflat also seemed to be stable after 2 years (Wightlink Ltd, 2015). The differences in bed elevation between the two studies can be explained due to differences in the volume of placed sediments. Implementation of unconfined intertidal and subtidal placement is both limited to one time. In contrast to subtidal placement, intertidal unconfined placement was unsuccessful. Subtidal unconfined direct placement has been observed to contribute to wave damping. Under two different wave conditions a reduction in wave energy was measured of 29% and 46% in Mobile Bay (USA) (Bray, 2008; Mehta & Jiang, 1993). Although only implemented once, the fact that natural unconfined subtidal mud berms have been observed to attenuate waves (Box 2) shows the potential of this type of direct placement when feasible in practice to implement. However, it is unclear if this wave damping refers to attenuation of small waves along the waterline. This is important since it has been suggested that mainly small waves along the waterline contribute to erosion (Winterwerp, et al., 2005). However, the observed wave attenuation does indicate the potential of the erosion mitigation solution. Due to a lack of monitoring it is unclear over what length of time these services are provided

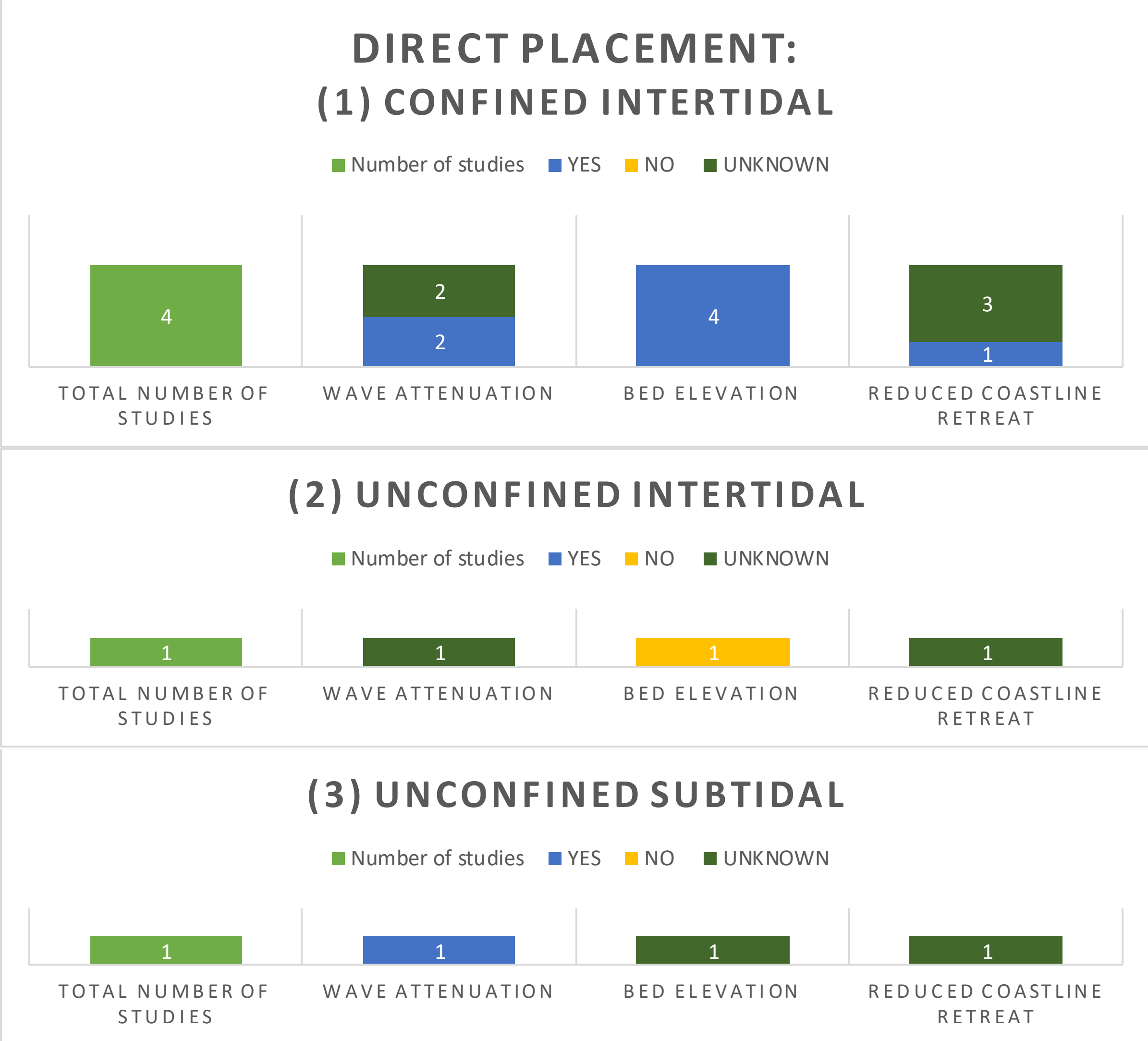


Figure 5.4: Observed erosion mitigation services provided by 6 mud recharge projects for (1) Confined intertidal, (2) Unconfined intertidal and (3) Unconfined subtidal recharge schemes. Information based on Appendix 11.3. For unconfined subtidal recharge (3) it is not mentioned whether the observed wave attenuation also refers to small onshore waves (the main contributor to erosion).

BOX 2 **Direct placement: implemented projects**

Confined intertidal: Four example projects where found which mentioned the provision of erosion mitigations services. Examples all include restoration of wetlands. French and Burningham (2009) for example state that their intertidal recharge scheme in the Orwell Estuary (UK) was successful in elevating the bed with 1,1m, restoring the salt marshes and reducing wave-erosion along a 450-m length of seawall. Also, foreshore recharge with silt material behind brushwood fencing and sand barriers along an eroding salt marsh at Horsey Island in England increased the mudflat area. This elevated the bed level, so that over time saltmarsh could establish naturally and subsequent schemes topped up the saltmarsh with more mud (Hamer, 2007; Fletcher et al., 2000). An intertidal recharge scheme at Lymington Estuary (UK) showed that after two years the mudflat was stable and stalled the ongoing process of physical erosion of the marsh. Elevation levels varied between 3 to 29 cm (Wightlink Ltd, 2015). More projects have been implemented of which no mention is being made whether the scheme contributed in mitigating erosion (Appendix 12.3). Overall, it becomes clear from Figure 5.4 that all confined intertidal schemes contributed to mitigating erosion by elevating the bed. One study also states coastline retreat was reduced and two waves where dampened.

Unconfined intertidal: Only one scheme was found for unconfined intertidal placement: recharge on an eroding saltmarsh along Horsey Island (UK) was deemed unsuccessful since most silt washed off by the spring tide (EA, n.d).

Unconfined subtidal: Naturally occurring underwater (subtidal) mudbanks can absorb wave energy, consequently attenuating waves on their leeward side. Thus that sediments can settle. The degree of alteration in wave energy depends on their dimension, the composition of the sediment and the incoming wave characteristics (Mehta, 2002). Natural examples include mudbanks in Kerala (India) and former, prominent mudbanks on the eastern margin of the Louisiana chenier plain (USA) where episodic waves were measurably damped over the banks (Mehta & Jiang, 1994). Only one example of a constructed unconfined subtidal mud berm was found: it was constructed in 1988 in the Gulf of Mexico off Dauphin island in Alabama (Mobile Bay) from dredged material which aimed to absorb wave energy. Under two different wave conditions the measured reduction in wave energy amounted to 29% and 46% (Bray, 2008; Mehta, 2002). It is not confirmed if this wave damping refers also to small waves along the shoreline. In general, these offshore berms are aligned roughly parallel to the coastline (Burt, 1996). No other implemented projects where found who executed similar schemes.

5.2.4 Success factors of implementation

Technical

1. Hundreds of cubic kilometres of sediments are dredged each year for commercial and recreational purposes, which is often discharged in the ocean. This <u>dredged material could be used as a resource</u> for stabilizing and restoring eroding coasts in case the dredged material is uncontaminated (Costa-Pierce & Weinstein, 2002). However, much of the dredged material removed from harbour and channel maintenance contains a mixture of contaminants arising from industry, agriculture and domestic activities (Colenutt, 1999b). When uncontaminated, mud replenishment can be a useful employment though of dredged material (UK CHM, 1999). However, to minimize transportation costs, the dredging area should be as close as possible to the placement area, but not too close as to encourage immediate return of the material in case of river dredging (Burt, 1996). For remote areas, the beneficial use of dredged material might therefore not be economical. In this case a nearby borrow pit must be chosen (Julianus, 2016).

Environmental

1. Although infaunal species are smothered during placement of dredged material, <u>rich and diverse infaunal communities have been shown to establish on intertidal mudflats</u> constructed of dredged material within one to three years (Bolam & Whomersley, 2005; Ray, 2000).

However, studies have shown differences to reference sites in community structures (Bolam & Whomersley, 2005) and in total biomass. The latter indicates the importance to further investigate future mudflat constructions, since mudflats have an important function to support higher trophic levels (e.g. fish and birds) (Ray, 2000). Furthermore, Harvey et al. (1998) states that the impact of sediment placement on benthic communities vary depending on many factors like the amount and nature of disposed sediments, the water depth, frequency of disposal, hydrography, time of year, types of organisms inhabiting the disposal area and similarities of de deposited sediments to that of the area of placement. This could imply for example that frequent placement of sediments inhibits resettlement of benthic communities.

5.2.5 Failure factors of implementation

Technical

1. While consolidation takes place, high losses of material can be expected due to the low resistance to erosive forces (Burt, 1996), except in the lowest energy environments (Fletcher, et al., 2000). This is especially the case of unconfined placement. However, an intertidal confined recharge scheme in the Lymington Estuary experienced also loss of more than half of the silty sediments placed in the recharge area (Wightlink Ltd, 2014). Kirby (1995) proposes two available methods to increase the strength of dredged material prior to placement to reduce erosion after placement. This implies using mechanical/chemical de-watering process plants or use temporary holding areas ashore for the dredged material. However, this may change the particle size distribution of sediments, which can potentially impact successful colonization of particular flora and fauna who require particular sediment characteristics (Kirby, 1995). Furthermore, this will increase costs. Burt (1996) also indicates that erosion of unconfined subtidal mud berms (after placement) is likely due to wave forcing, potentially requiring maintenance. However, when the eroded sediments move onshore this could also aid in mitigating coastal erosion. In general, how long sediment particles will remain after settlement/placement will determine their effectiveness in erosion mitigation. This is still a knowledge gap and is likely site specific due to differences in wave dynamics.
2. Soft retaining structures like gravel bunds placed on low wave energy foreshores, have shown to roll over landward in the Orwell estuary (UK), reducing the width of the created mudflat by 60% in 10 year time (leaving 15 to 25 m of mudflat width behind) (French & Burningham, 2009).

Environmental

1. Sediments lost may cause a temporary unacceptable level of turbidity from an ecological point of view in the nearby area (Burt, 1996). Turbidity refers to "the degree to which water contains particles that cause backscattering and absorption of light" (Dankers, 2002, p. 45). The higher the amount of suspended mud particles, the higher the turbidity and consequently the amount of light penetrating into the water column and onto the seabed is reduced. This may affect primary producers (aka phytoplankton and bed vegetation) at the basis of the marine food chain and predators like fish and birds that feed on sight. Furthermore, the higher concentration of suspended material can enhance sediment depositions. The latter may influence the growth and survival of bed flora and fauna (Ibid.). Direct placement also smothers the existing habitat on site (Figure 5.5) (Fletcher, et al., 2000).
2. Removing sediments from a borrow pit to use as a sediment source for direct placement can influence alongshore or across-shore sediment transport at the pit (depending on its location) (Julianus, 2016). Furthermore, potential ecological impacts of removing sediments from a bed should also considered (Figure 5.5). The possible physical and ecological effects should be researched.

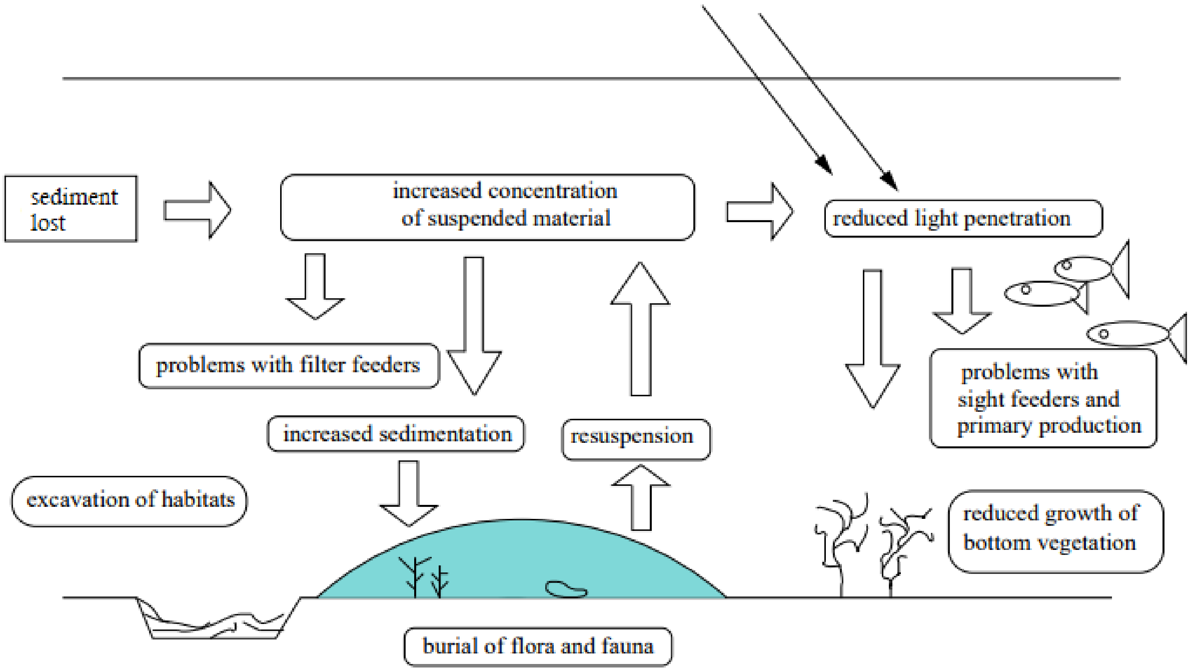


Figure 5.5: Potential impacts of sediment recharge schemes on marine ecology (adapted from Dankers, 2002).

Economical

1. To minimize transport costs, the dredging area should be as close as possible, but not too close as to encourage immediate return of the material in case of river dredging (Burt, 1996).
2. Besides the potential ecological impact of sediments lost, this loss of material can also impact commercial interest along- and off-shore, such as siltation of shellfisheries and sedimentation of dredged channels (Colenutt, 1999c). This can cause opposition against recharge schemes.

Social

1. When applying dredged material, potentially public opposition can arise. This was the case in the New Yorks New Jersey Harbor (USA) where opposition arose to the near-shore placement of dredged material determined to be unsuitable for open ocean disposal, even though this scheme was intended to restore degraded habitat (Yozzo, et al., 2004). Regardless of the actual status of dredged material, it may still be perceived to be contaminated (ABP Research, 1998). However, this perception is gradually being changed (Park & Lee, 2007).

5.3 Trickle charge

5.3.1 Technique description

Trickle charge implies bringing sediments into the natural system, using natural processes to redistribute the material. At the side of placement, it does not intend to change the habitat. In essence, it functions as a slow recharge source for the foreshore. It can be done by placing sediment in the subtidal or intertidal zone, or through water column recharge (Fletcher, et al., 2000; Foster, et al., 2013). Figure 5.6 shows a conceptualized image of the effect of these three techniques on hydrological and morphological processes. In case of strong onshore currents, the sediments can be placed just above the low water mark (intertidal) or above normal wave base (subtidal) (UK CHM, 1999). The deposited materials are eroded and transported by the rising tide to increase the sediment load on the intertidal zone (Colenutt, 1999a; MAFF, 1993). For coasts with strong longshore currents, sediment can be placed up drift. The sediments will move naturally along the coast (UK CHM, 1999). Subtidal or intertidal placement can be at a single point or at a series of point along the shore (Foster,

> **Definition trickle charge:**
> *Bringing sediments into the natural system to form a slow recharge source for the foreshore.*

et al., 2013). Water column recharge entails discharging material into the water column at such a rate/dilution that the moving water column carries the recharged material away from the site of introduction, without impacting the sea bed. The impact depends on energy within the water column (tidal currents and turbulence) (Fletcher, et al., 2000).

Implementing trickle charge will imply loss of material outside the targeted area. Possibly the loss of sediment can be reduced through combining trickle charge with brushwood fencing. These permeable structures are composed of brushwood or similar material. Often a double row of wooden stakes is driven well into the mudflat. They allow water to pass through, dampen waves and reduce the water velocity sufficiently to allow sediments to settle and increase the mudflat level (Figure 5.6). This can be applied when plenty of fine sediment is in suspension. In case of strong longshore currents this can be done by placing shore normal brushwood structures. When there is also a strong across shore current, structures can also be placed parallel to the coast, thus creating a box or sedimentation field. The trickle charge is then placed offshore the fences (Colenutt, 1999c; Tonneijck, et al., 2015; Colenutt, 2001). Although no documented examples were found of this combined technique, brushwood fences have been observed to accrete sediments to such bed levels that lost mangrove forest and salt marshes could re-establish. Brushwood fences are currently being applied in the Mekong delta (Vietnam), in Demak (Indonesia) and along the coastline near Paramaribo (Suriname) to elevate bed levels to restore mangroves (Deltares, 2016; Colenutt, 2001).

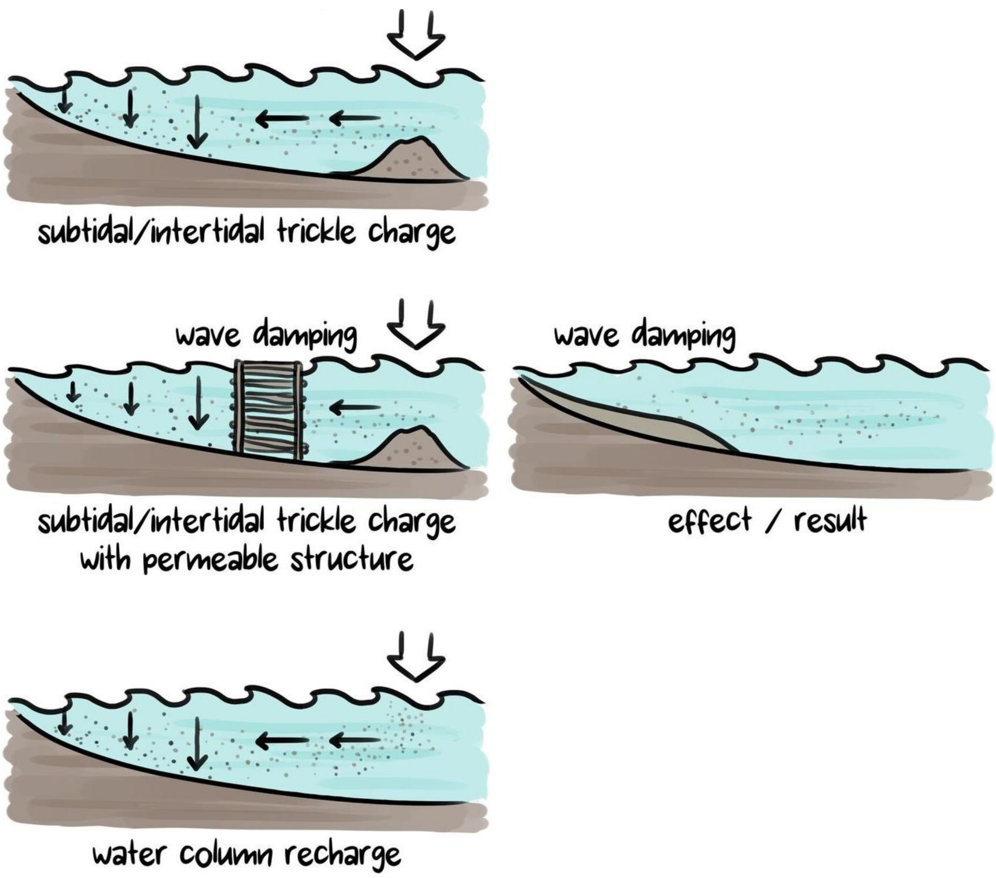


Figure 5.6: Vertical view of conceptualized influence of 3 trickle charge techniques on hydrological and morphological processes along coasts: subtidal and intertidal trickle charge (with and without permeable structures) and water column recharge. The figure on the right shows the targeted end results of all the three trickle charge techniques (Illustrated by Van Ginneken, 2017c).

5.3.2 Environmental conditions

Across-shore

Trickle charge can both be done subtidal and intertidal.

Coastal type / habitat requirements

Environmental conditions under which trickle charge can be applied are unknown. Based on the limited number of implemented projects found (one of each type of trickle charge, all located in bay/estuaries (Appendix 11.3) it might be suggested that trickle charge is suitable for the bay/estuary mud coast type. This could imply the requirement of sheltered conditions. However, it is unknown if trickle charge might be suitable for other coastal types. Brushwood fencing is not suitable in every environment. A trial at Zuidgors Salting in the Westerschelde (the Netherlands) showed that the brushwood groynes were too weak to hold against the strong currents (Colenutt, 1999c).

5.3.3 Erosion mitigation

Three projects were found which implemented muddy trickle charge, one of each type: intertidal, subtidal and water column recharge. Examples are described in Box 3. Figure 5.7 shows how the projects aided in mitigating erosion: all three different types of trickle charge schemes increased the intertidal bed level (ABP Research, 1998; UKMPA Centre, 2001; Mundy & Kelly, 2010). It is likely that this has also resulted in wave damping. However, no data is available to support this. Water column recharge was found more successful in elevating the bed than subtidal trickle charge (Mundy & Kelly, 2010). Findings are based on descriptive grey literature. Thus, quantitative data is unavailable.

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BOX 3 Trickle charge, implemented projects

 # Intertidal trickle charge: An example of a trickle charge experiment on lower intertidal area was performed at Medway Port (UK) in 1996 using 4000 m^3 of dredged silt. Results showed that trickle feeding can achieve intertidal recharge for relatively small infrequent volumes of fine material. It was estimated that approximately 50% of the material was retained at the recharge site (UKMPA Centre, 2001; ABP Research, 1998).

 # Subtidal trickle charge + Water column recharge: Within the Stour and Orwell estuary (UK) an ongoing sediment replacement program exists since 1994 to compensate for lost intertidal habitat due to increasing erosion rates of up to 5 hectares per year. Subtidal trickle charge was being executed 2 to 3 times a year, but replaced with water column recharge schemes since 2001 due to the inefficient dispersal of silt material. They found that undertaking water column recharge on a rising tide is more efficient in terms of feeding intertidal areas than undertaking subtidal trickle charge (Mundy & Kelly, 2010).

5.3.4 Success factors of implementation

Technical

1. Uncontaminated dredged material can be usefully employed for mud recharge, instead of it being regarded as waste. See Section 5.2.4, technical factor 1.

Environmental

1. Subtidal trickle charge avoids impacting the intertidal habitats due to the slow recharge rate (Fletcher, et al., 2000). However, the creation of sedimentation polders may result in accretion rates above that which is tolerable to infauna (UK CHM, 1999).

Economic & Social

Knowlegde gap.

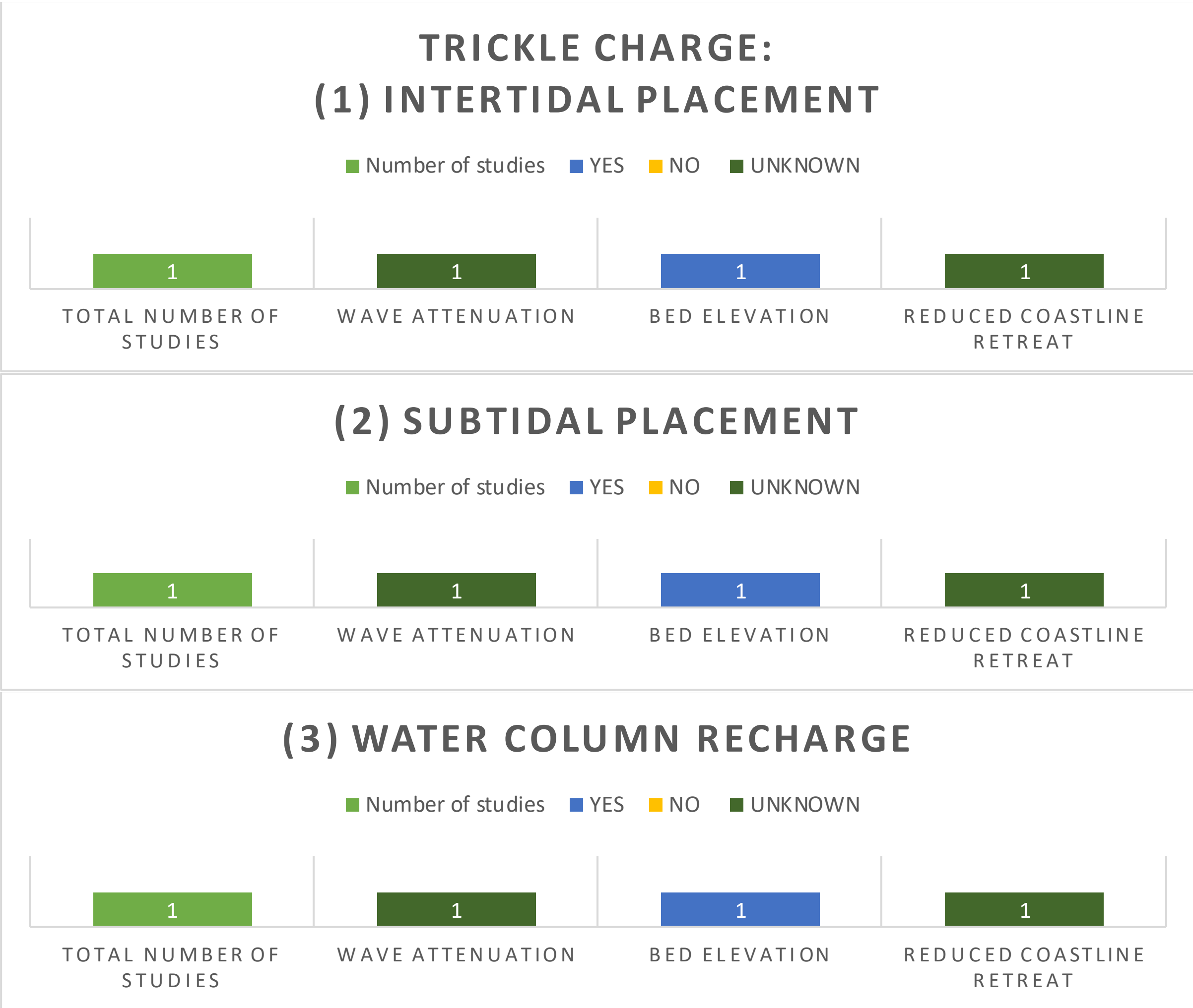


Figure 5.7: Erosion mitigation services which have been observed to be provided by implemented projects of three types of trickle charge: (1) intertidal trickle charge, (2) subtidal trickle charge, and (3) water column recharge. Note for graph (2): bed elevation through the subtidal trickle charge scheme was found inefficient. Graphs are based on Appendix 11.3.

5.3.5 Failure factors of implementation

Technical

1. <u>Sediment can be lost</u> to areas outside the targeted area, which for example occurred in the Stour and Orwell estuary (UK) when subtidal trickle charge was performed (Mundy & Kelly, 2010). Thus, more sediment is required to recharge the targeted area. This can increase costs and potentially temporarily impact marine life (see Section 5.2.5, environmental factor 2) and affect commercial interest (see Section 5.2.5, economic factor 2).
2. The <u>recharge process is slower</u> compared to direct placement (UK CHM, 1999).

Environmental

1. <u>Using different grain size to the existing mudflat can impact marine life</u> (see Section 5.2.4, technical factor 2).
2. <u>Potential physical and ecological impacts of removing sediments from a borrow pit (</u>see Section 5.2.5, environmental factor 3).

Economical

1. To minimize transport costs, the dredging area should be as close as possible, but not too close as to encourage immediate return of the material in case of river dredging (Burt, 1996).

Social

Knowledge gap.

5.4 Sediment stirring

5.4.1 Technique description

Definition sediment stirring:
 # Agitation dredging: seabed sediments are brought into suspension over the whole water column to form a recharge source.
 # Water injection: Water injected into the bed to fluidize mud. The fluid mud flows on the lower part of the water column, driven by density to form a recharge source elsewhere.

Sediment stirring can be referred to as agitation dredging. With agitation dredging the muddy seabed sediments are brought into suspension over the whole water column and transported away by tidal currents (Verhagen, 2000). There are different mechanical and hydraulic techniques to achieve agitation dredging (Fletcher, et al., 2000). A related technique is water injection dredging. By injecting water into the mud layer, the water content of the mud increases and it becomes fluid mud. This fluid mud will flow because its density is higher than the surrounding water. The material can flow over large distances in orders of kilometres (Verhagen, 2000) until it settles at a site of lower elevation within the river or estuary system. It differs from normal agitation dredging in the sense that the mud layer flows horizontally on the lower part of the water column along the waterbed (Bray, 2008). Although these techniques have been applied for maintenance dredging to remove muddy material from ports and channels (Sullivan, 2000; Fletcher, et al., 2000), no examples were found where these techniques were applied to mitigate erosion along mud coasts. Figure 5.8 schematically shows how the techniques influence hydrodynamic and morphodynamic processes. Similar as with trickle charge, permeable structures might be implemented together with sediment stirring to reduce the loss of sediments outside the targeted area (Section 5.3.1) (Tonneijck, et al., 2015).

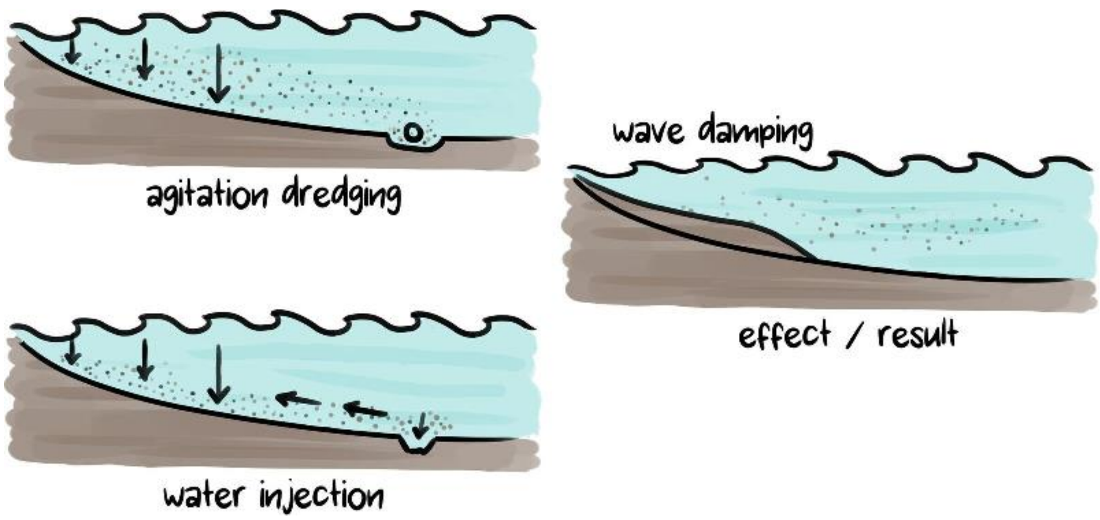


Figure 5.8: Vertical view of conceptualized influence of 2 sediment stirring techniques on hydrological and morphological processes along coasts: agitation dredging and water injection. Stirred sediments transported landward by the tide, might settle along the coast. The figure on the right shows the targeted end results of the sediment stirring techniques (Illustrated by Van Ginneken, 2017b).

5.4.2 Environmental conditions

Across-shore

Unknown.

Coastal type

Unknown.

Environmental conditions

Agitation dredging is mainly used in areas with a strong (tidal) current and high background turbidities. When currents are variable, agitation dredging can be restricted to periods when currents are high (Bray, 2008). This is not the case for water injection dredging, in which sediments are transported through a gravity-driven density current. The presence of tidal flow is not a precondition. However, an outward tidal flow will ease the process and an inward tidal flow will slow the process (Verhagen, 2000). Potential other environmental preconditions need to be researched.

5.4.3 Erosion mitigation

Since no implemented projects were found of sediment stirring for erosion control, observational knowledge related to the erosion mitigation success is lacking.

5.4.4 Success factors of implementation

Environmental

1. When sediment stirring is performed in an estuary, an advantage is that the dredged material remains in the sedimentary system. Thus, a sediment balance is maintained (Bray, 2008).

Economical

1. Advantage are the lower cost, compared to dredging techniques related to direct placement and trickle charge (Bray, 2008). This can be explained by absence of the need to transport dredging material and due to the lower energy demand compared to pumping mud (Verhagen, 2000).

Technical & Social

Knowledge gap.

5.4.5 Failure factors of implementation

Technical

1. With agitation dredging it is difficult to control the position where sediments will settle. Usually the mud settles in a thin layer spreads over a large area. This makes this technique less suitable for environmentally sensitive projects (Verhagen, 2000; Bray, 2008). Water injection dredging is more accurate; it is possible to deliver a flat bed surface with an accuracy of 0,10 meter (Bray, 2008).
2. Performance of water injection dredging reduces as cohesion and consolidation increases of muddy particles (Verhagen, 2000).

Environmental

1. The suspended sediments will temporary increase the turbidity of the water column. From an ecological point of view this may cause a temporary unacceptable turbidity level, which for example may affect fish or primary producers (Burt, 1996) (section 5.2.5, environmental factor 1). With water injection dredging most of the material moves close to the riverbed, therefore the effect on turbidity of the upper water layers is limited (Bray, 2008).
2. Stirring might affect the benthic community at the stirring site and reduce alongshore sediment transport down drift. The latter could aid in down drift erosion.

Economical & Social

Knowledge gap.

5.5 Compare techniques: direct placement, trickle charge & sediment stirring

5.5.1 Mitigating erosion & Environmental conditions

Based on the limited documentation available, Table 5.1 gives an overview of the previously discussed implemented recharge projects of which their contribution to erosion reduction is documented. No implemented schemes were found for sediment stirring. Direct unconfined intertidal recharge was unsuccessful in reducing erosion. Direct confined intertidal placement and all three types of trickle charge (intertidal, subtidal and water column recharge) have been observed to mitigate erosion. However, subtidal trickle charge was found less efficient than water column recharge, and is thus not advised to implement. Bed elevation was the erosion mitigation service observed most. However, wave attenuation and reduced coastline were also observed for direct confined intertidal placement. Direct unconfined subtidal placement has also been observed to attenuate waves. It remains unknown though if this is also true for small waves along the shoreline, the main contributor to erosion (when looking at wave influence). Thus, the potential of this erosion mitigating option has yet to be confirmed. Most studies only indicated whether one type of erosion mitigation service was provided (often bed elevation), not mentioning other services. This leaves large unknowns related to the erosion mitigation services provided.

All recharge schemes found in literature were performed in bays or estuaries, suggesting mud recharge is suitable for more sheltered conditions. However, due to the limited availability of studies, further research is needed to indicate if different recharge techniques can also be applied in other type of muddy coasts. However, since barrier coasts entail more sheltered conditions than bays and estuaries, it is likely mud recharge is suitable here too. Depending on the type of recharge scheme, recharge can both be performed subtidal and intertidal.

Sediment Recharge schemes	Environmental conditions					Erosion mitigation services		
	Across shore		Coastal type			*Bed elevation*	*Wave atte-nuation*	*Reduced coastline retreat*
	Inter tidal	*Sub tidal*	*Open coast*	*Bay/ estuary*	*Barrier*			
Direct placement								
• ***Confined intertidal***	X			4		YES (4)	YES (2)	YES (1)
• ***Un-confined subtidal***		X		1			MAYBE ** (1)	
• ***Un-confined intertidal***	X			1		NO (1)		
Trickle charge								
• ***Intertidal placement***	X			1		YES (1)		
• ***Subtidal placement***		X		1*		YES (1) ***		
• ***Water column recharge***	X			1*		YES (1)		
Sediment stirring								
• ***Agitation dredging***								
• ***Water injection dredging***								

*Table 5.1: Environmental conditions and erosion mitigation services observed to be provided by different type of mud recharge schemes. Only implemented project were included which indicated whether erosion mitigation was successful or not. Grey=indicates knowledge is lacking or unavailable. ( )= Numbers between brackets indicate the number of projects on which the results are based. X = based on literature. *=scheme occurred in one estuary, however, this has been performed for 2 to 3 times a year for several years in a row. **=wave damping was observed of 29% and 46%. However, unclear if this was true for small erosive waves along the shoreline. ***=less effective then water column recharge. Information is based on Appendix 11.3.*

5.5.2 Success & failure factors of implementation

When comparing success and failure factors related to the implementation of different techniques, the following factors can be identified.

Environmental & Technical

With all solutions, sediment is removed from a borrow pit which can cause physical or ecological impacts at the pit, depending on its location across shore. For example, removing sediments from the literal zone can alter alongshore sediment transport, potentially causing a sediment deficit down drift. For trickle charge and direct placement this can be avoided by using uncontaminated dredged material which is removed anyway. This is only possible when the recharge location is in proximity of the dredging areas. Dredged material should have similar characteristics to local sediments to reduce potential environmental impact. However, it should be considered that public opposition might arise due to the 'contaminated' perception of dredged material.

Direct placement smothers the benthic community, thereby reducing the food supply to fish and birds. Many intertidal areas are important international over-wintering grounds. Burial of infauna could have major implication since this deprives birds from their food (UK CHM, 1999). However, re-establishment of rich community is possible over time, although possibly not equal to the reference community. Frequent direct recharge might inhibit this. With sediment stirring and trickle charge the slow recharge rate will have less impact on marine life. However, combining the latter two solutions with brushwood fencing could also imply smothering of benthic life in case the sediment settling rate is too high. However, when the impact on marine life is of less importance, then direct placement could be preferred due to the higher recharge rate compared to trickle charge. The recharge rate of sediment stirring is likely also slower compared to direct placement. This is however not confirmed.

Furthermore, all solutions seem to be related to loss of sediments outside the targeted area. This could form a trickle source for other areas, which might be beneficial. However, it can also cause temporary unacceptable levels of turbidity which might be a threat to sensitive habitats like shellfish beds, spawning habitats, clear water estuaries etc. (IADC, 2013) and impact commercial interest along- and off-shore, such as siltation of shellfisheries and sedimentation of dredged channels. It is likely that sediment losses are lower with direct confined placement compared to unconfined direct placement, trickle charge and agitation dredging. Further research is needed to prove this. To reduce the loss of sediments related to direct placement, the pumped mud can be de-watered before placement. Trickle charge and sediment stirring might be combined with brushwood fencing to reduce sediment losses, however, this combined technique has not been implemented yet. Water injection dredging (aka sediment stirring) can be considered more environmentally friendly compared to agitation dredging in the sense that the latter induces sediment suspension in the whole height of the water column, while water injection dredging only induces sediment transport just above the bed. The overlaying water layers are not affected (IADC, 2013). Furthermore, with agitation dredging it is difficult to control the targeted location, which makes this technique less suitable for environmental sensitive areas.

Economical

The disadvantage of trickle charge and direct placement is that muddy material is obtained elsewhere. This implies relatively high transportation costs and a sediment composition which might differ from sediments on the recharge location. This might impact the lower biomass benthic community and subsequently fish and birds. It is likely that agitation dredging is less costly compared to direct placement and trickle charge because there is no need to transport sediments and because energy requirements related to stirring are lower compared to pumping mud. It is unknown if this is also true for water injection dredging.

5.6 Conclusion

The limited availability of implemented recharge projects, suggests mud recharge schemes of direct confined intertidal placement and trickle charge can aid in mitigating erosion in the coastal type estuaries and bays. However, the efficiency cannot be quantified due to lack of quantitative data. Bed elevation was observed most during projects. Data concerning the provision of other erosion mitigation services (wave attenuation and reduced coastline retreat) is limited. Potentially direct unconfined subtidal placement can also aid in mitigating erosion. However, it remains unknown whether the observed wave attenuation is true for small waves along the water lines which cause erosion. Due to the limited number of implemented projects and the large unknowns in the erosion mitigating services provided, these results should be taken with caution. Sediment stirring has also been suggested as a recharge technique, but not been implemented. Depending on the chosen techniques, mud recharge can both be done in the intertidal and subtidal area. Furthermore, since all implemented projects have been implemented in estuaries and bays, it is unknown whether mud recharge can be applied successfully along open and barrier coasts to mitigate erosion. However, since barrier coasts are more sheltered environments than estuaries and bays, mud recharge might be applicable along barrier coasts too. It has been suggested that direct placement techniques can be applied in low to moderate wave energy environments (definition unknown). Requirement of other environmental conditions have not been found. This lack in information is also true for trickle charge and sediment stirring. This indicates the need for more experimental research to indicate the effectiveness of direct placement, trickle charge and sediment stirring in mitigating erosion and under what environmental conditions these techniques can be applied.

When choosing a mud recharge technique, the following pros and cons could be taken considered. All sediment recharge schemes are related to loss of sediment outside the targeted area, thus increasing turbidity. Direct placement and potentially applying brushwood fencing with sediment recharge schemes, causes smothering of benthic life. Consequently, applying sediment recharge might be less suitable in areas with the presence of sensitive marine life, important overwintering grounds and commercial activities like shellfisheries. However, when chosen, trickle charge and water injection dredging (without brushwood fencing) are advised above direct placement and agitation dredging. This is due to the lower recharge rate of sediment stirring and trickle charge, compared to direct placement (UK CHM, 1999). Furthermore, injection dredging is advised above agitation dredging since controlling the position where sediments will settle is more accurate with water injection. Water injection dredging might even be more environmentally friendly compared to trickle charge because the dredged material remains in the sedimentary system, while trickle charge uses an external sediment source. However, no implemented projects of water injection dredging were found to be implemented to mitigate erosion, which makes the suitability of this technique yet unsure. When execution time is of importance, then direct placement is preferred over trickle charge and sediment stirring due to its higher recharge rate. From an economical point of view agitation dredging would be preferred above direct placement and trickle charge due its likeliness of lower transportation and energy costs during execution. Relative costs of water injection dredging are unknown. However, muddy sediment stirring (agitation dredging and water injection) has not been implemented yet for erosion mitigation. Although sediment recharge with mud has not yet been implemented together with brushwood fending, this might aid in reducing the loss of sediment material in less environmental sensitive areas.

Knowledge concerning the lifespan of the solutions was unavailable. The only mention was that underwater mud berms might need maintenance due to the erosive forces that work on the mud. This might be true for other solutions too. Findings are brought together in Table 5.2.

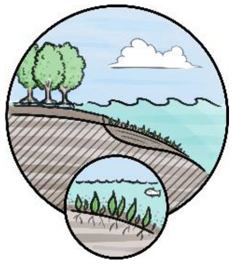
Direct placement	Trickle charge	Sediment stirring
Pros: • Higher recharge rate	*Pros:* • Potential public opposition when dredged material is used • Slower recharge rate (=positive for marine life)	*Pros:* • Potentially lowest cost • (Likely) slower recharge rate (=positive for marine life) • Water injection: only sediment transport just above bed (=positive for marine life • Sediments remains in system
Cons: • Potential public opposition when dredged material is used • Removing sediment from borrow pit (when dredged material is not available) → physical + ecological impact • High impacts marine life • Loss sediments outside targeted area (impact marine life and commercial interest)	*Cons*: • Removing sediment from borrow pit (when dredged material is not available) → physical + ecological impact • Loss sediments outside targeted area (impact marine life and commercial interest)	*Cons*: • Removing sediment from borrow pit → physical + ecological impact • Loss sediments outside targeted area (impact marine life and commercial interest) • Agitation dredging: difficult to control targeted location
Successful implementation on muddy substrate: • Across-shore: intertidal + potentially subtidal • Coastal type: bay/estuary	*Successful implementation on muddy substrate:* • Across-shore: intertidal + subtidal • Coastal type: bay/estuary,	*Successful implementation on muddy substrate:* • No examples found

Table 5.2: Pros and cons which can be considered when choosing a mud recharge technique. Successful implementation implies one or two erosion mitigation services have been observed. Implementation of mud recharge schemes was not found along open and barrier coast.

Seagrass bed in Bali (from Wetlands International, 2017b)

6 Seagrass

6.1 Introduction



Seagrass grows fully submerged and rooted in estuarine and marine environments. In many places they cover extensive areas, thus forming seagrass beds or meadows. Seagrasses play an important role in the functioning of coastal ecosystems like mangroves, oyster reefs and intertidal flats (Green & Short, 2003). Of the approximately 60 species of seagrasses, 31 can reside in tropical areas (see species in Appendix 11.4.1). Two tropical geographic bioregions can be distinguished which resembles the tropical region as defined in Section 2.1.1 (Figure 6.1). The Tropical Atlantic bioregion has clear water with a high diversity of seagrasses on reefs and shallow banks. The Tropical Indo-Pacific has the highest seagrass diversity in the world, mostly growing on reef flats between the reef break and the shore. However, seagrasses also occur in very deep waters up to 70 meters deep (Short, et al., 2007).

Seagrass typically grows on soft substrates such as sand or mud. It differs per species which substrate is preferred. However, often species occur on a variety of substrates (Green & Short, 2003; IUCN, 2016). For example, of the 6 most commons tropical seagrass species (Appendix 11.4.1), only 2 prefer sandy or coarser sediments, but they are also found on muddy substrates. The other 4 specie both thrive on sand and mud (IUCN, 2016). However, growth and survival rate of restored seagrass beds can potentially be influenced by soil type (Zhang, et al., 2015).

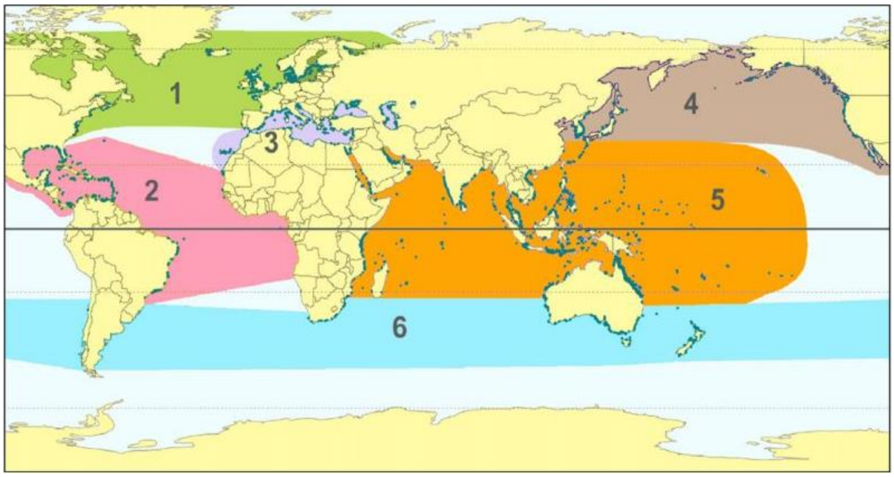


Figure 6.1: Global seagrass distribution shown as blue points and polygons (data from 2005 UNEP-WCMC) and geographic bioregions: 1. Temperate North Atlantic, 2. Tropical Atlantic, 3. Mediterranean, 4. Temperate North Pacific, 5. Tropical Indo-Pacific, 6. Temperate Southern Oceans (from Short, et al., 2007).

Section 6.2 will describe the general required environmental conditions for seagrass to grow. Restoration techniques and their related required environmental conditions and success and failure factor will be discussed and compared in Section 6.3. No information was found to indicate how effective different restoration techniques are in mitigating erosion. More important seems the influence of the different techniques on the efficiency of seagrass restoration. Therefore, the relation between seagrass restoration techniques and erosion mitigation can be considered indirect. Consequently, only the general ability of seagrass to aid in erosion reduction will be discussed (Section 6.4). General pros and cons of seagrass transplantation will be discussed in Section 6.5 and 6.6, followed by a conclusion in Section 6.7.

6.2 General environmental conditions

Across-shore

In the tropics, seagrass beds occur both on shallow reef flats in subtidal nearshore areas (Christianen, et al., 2013), as well as intertidal areas (De Fouw, 2016). In fact, more tropical species can grow in the intertidal area then temperate species, where they are often exposed to high irradiance levels and desiccating atmospheric conditions (Shafer, et al., 2007). Location preferences across-shore are species depended (NAOO, 2011)

Coastal type

Seagrass species have been observed to grow along open coasts, bays, estuaries and in lagoo ns behind coastal barriers (Appendix 11.4.2, Table 11.5). However, this may differ per specie. More important is whether habitat requirements for each species are met at the coastal location since these can be different at equal coastal type.

Habitat requirement

Environmental factors such as light availability (water depth and turbidity), temperature, salinity, tidal range, sediment stability and physical disturbances from waves and associated sediment movement, may be used to determine in advance whether seagrass growth could be supported at a given site (Fonseca, et al., 1987; Christianen, et al., 2013). Foraging of herbivores also influences growth (Christianen, et al., 2013). All these factors are important for initial establishment and long term survival (Turner & Schwarz, 2006). Most tropical species reside in water less than 10 meter deep (Short, et al., 2007). For restoration, sheltered locations with adequate light environments where seagrass beds historically resided, are preferred (Van Katwijk, et al., 2016). Preferred water depth is species depended (NAOO, 2011). Minimum water depth is mainly determined by wave orbital velocity, tide and wave energy. Maximum depth by light availability (De Boer, 2007). Furthermore, the water depth should be similar to nearby natural seagrass growth (Van Katwijk, et al., 2009). For more specifics on habitat requirements for seagrass see link below[1].

[1] https://publicwiki.deltares.nl/display/BWN1/Building+Block+-+Habitat+requirements+for+seagrass

6.3 Restoration techniques

6.3.1 Overview

6.3.1.1 Introduction

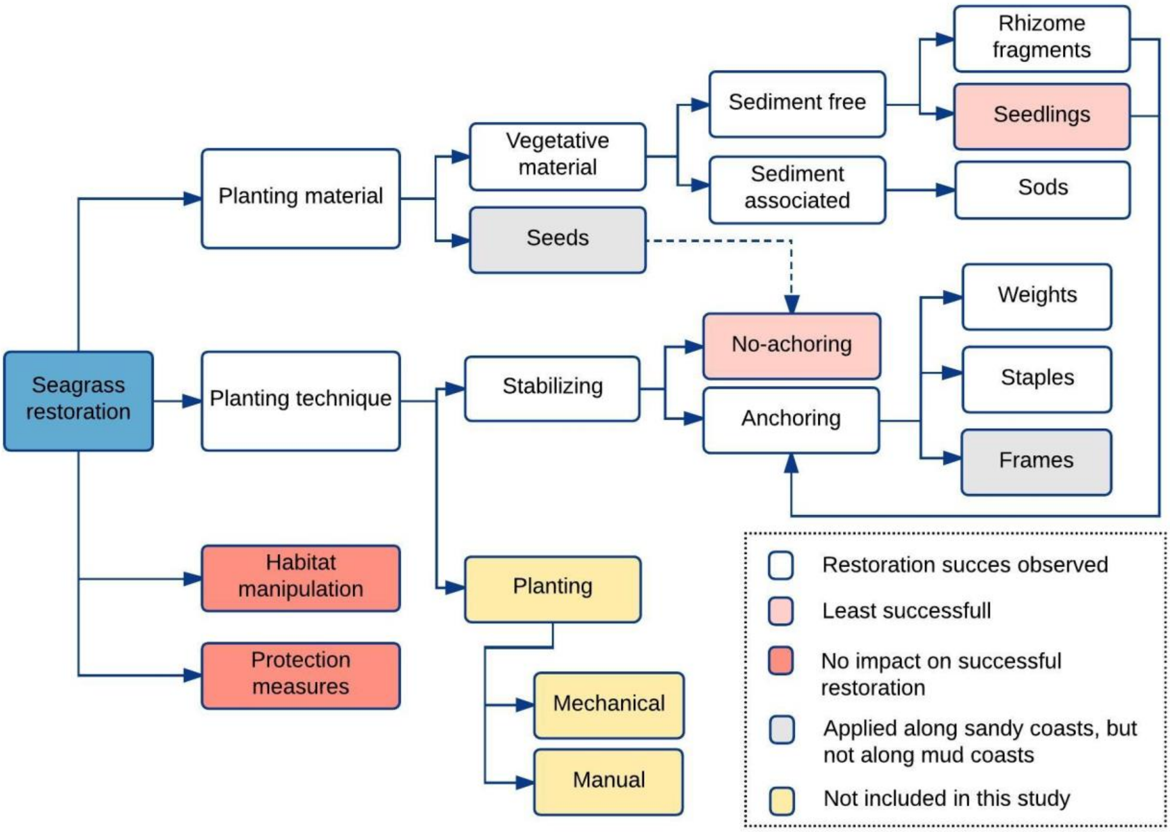


Figure 6.2: Potential restoration techniques of seagrasss (white boxes). Information is largely based on a review of best practices of seagrass restoration of seagrass restoration of Van Katwijk et al. (2016), which makes no distinction between soil types due to lack of data.

Worldwide the success of seagrass transplantation and restoration is uncertain and the experiences among species vary enormously. The most widely transplanted specie is *Z. marina*, a non-tropical specie. Survival rates vary with planting method, but a compilation of 53 planting experiences in the USA (not specified which soil type) showed a mean planting unit survival of 42% after one year. Only 5% of the planting projects with Z. Marina had a 100% survival rate (Ondiviela, et al., 2014). This indicates the difficulty of successful restoration.

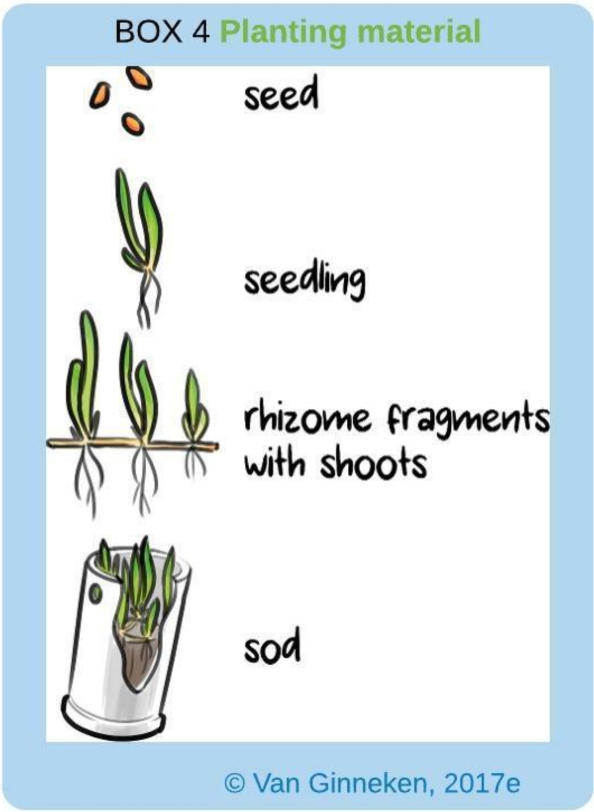
Figure 6.2 shows techniques to restore seagrass based on the results of a global systematic analysis of seagrass restoration of Van Katwijk et al. (2016). Van Katwijk et al. (2016) identified best practices based on the evaluation of 1289 trials of both tropical and temperate species. Soil type at the transplantation location was not considered due to lack of data (Van Katwijk, 2017, personal communication). However, according to Van Katwijk et al. (2009), sediment composition seems not to be vital for seagrass transplantation and is probably not a habitat requirement. This could be because sediment composition often reflects the prevailing water dynamics at a location, which is important to seagrass. This study therefore assumes that restoration techniques are independent of soil type. Local and regional expertise for seagrass restoration however remain important since Van Katwijk et al.'s (2016) global analysis only provides generalities.

Van Katwijk et al. (2016) indicated that planting material and the application of anchoring were the most important factors influencing restoration success when planting procedures were considered. Whether planting was done manual or mechanical was of lesser influence and will therefore not be elaborated on in this study. However, manual planting is the most common used practices compared to mechanical planting. Manual planting has been found to reduce initial survival, but somewhat improve later success scores as compared to mechanical planting (Ibid.). Habitat manipulation and implementation of protection measures with seagrass restoration had no positive effect on transplantation. Habitat manipulation refers to anti-bioturbation measures and sediment stabilization. Protection measures to creating shelter against hydrodynamics or grazing (Ibid.). Therefore, this section will only discuss planting material (Section 6.3.2) and stabilizing techniques (Section 6.3.3).

6.3.2 Planting material

6.3.2.1 Description

Planting material can be seeds based or vegetative material (seedling, rhizome fragments with shoots or sods (Box 4)) (Park & Lee, 2007; Van Katwijk, et al., 2016). Van Katwijk et al.'s (2016) review indicated that seedlings consistently perform worse compared to other planting material and is therefore not considered further.



Seeds can be planted into the bed or released into the water through hand-broadcasting or buoy-deployed seeding. The latter implies putting harvested flowering shoots suspended in mesh bags buoyed above the sediment of a targeted restoration area (Marion & Orth, 2010). Sods imply intact units of native sediment with roots, rhizomes and leaves (Van Katwijk, et al., 2016). Various materials have been used to extract plugs/cores from the donor site, including PVC pipes, small metal cans, sod pluggers and shovels (Bulseco-McKim, 2012). Removing a sod with a shovel is most applicable for hard, compact substrates and deep-rooted and large species (Perrow & Davy, 2002). Due to the weak structure of mud, it seems unlikely that this method is suitable for mud sediments. However, for small species the sediment with seagrass can for example be removed from the donor bed with a PVC pipe with caps on both sides and transported to the rehabilitation site (Perrow & Davy, 2002).

6.3.2.2 Successfulness of restoration

Van Katwijk et al.'s (2016) global analysis shows that the average success rate of seagrass transplanting (regardless of soil type) is highest when sods or rhizome fragments with anchoring are applied as planting material (Figure 6.3) and lowest for seeds or rhizome fragments without anchoring. Anchoring will be discussed in Section 6.3.3.

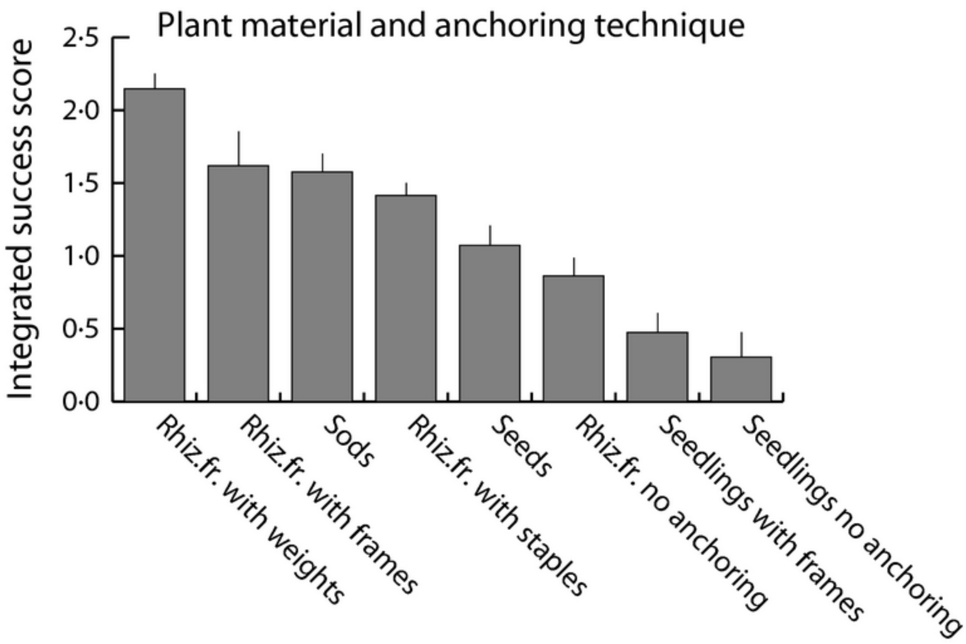


Figure 6.3 Relative performance of seagrass restoration trials in relation to plant material and anchoring techniques. The semi-quantitative integrated success score and its standard error of the mean were calculated from initial survival and long-term performance after initial survival of 1289 seagrass transplantation projects. Results included studies on both on sandy and muddy beds (from Van Katwijk, et al., 2016).

Eighteen seagrass restoration projects were identified through a literature search (Appendix 11.4.3) which implemented sods or rhizome fragments as planting material for seagrass restoration in a muddy environment. Most species were Z. Marina. Only one project was in a tropical area. No projects were found which applied seeds. All projects which implemented sods and two third of the projects with rhizome fragments showed some level of restoration success (Figure 6.4). However, it is difficult to assess a standard level of success over a certain length of time since studies do not always define what 'successful' means and monitoring is often only a few months. Only 4 of the 18 studies monitored over a period of 2 years or longer. Only 6 studies indicate a percentage of survival and some studies only see success for one growing season (for temperate species). Therefore, this study only makes a distinction between two categories: unsuccessful and successful. Successful referring thus to all studies indicate some form of success, whether temporary (one season only) or low (25%) or high (100%) survival percentage. Comparison on the level of success of muddy seagrass restoration projects is thus not possible with such a limited amount of data. It should be noted that seasonal effects are considered of lesser importance to tropical species compared to temperate species.

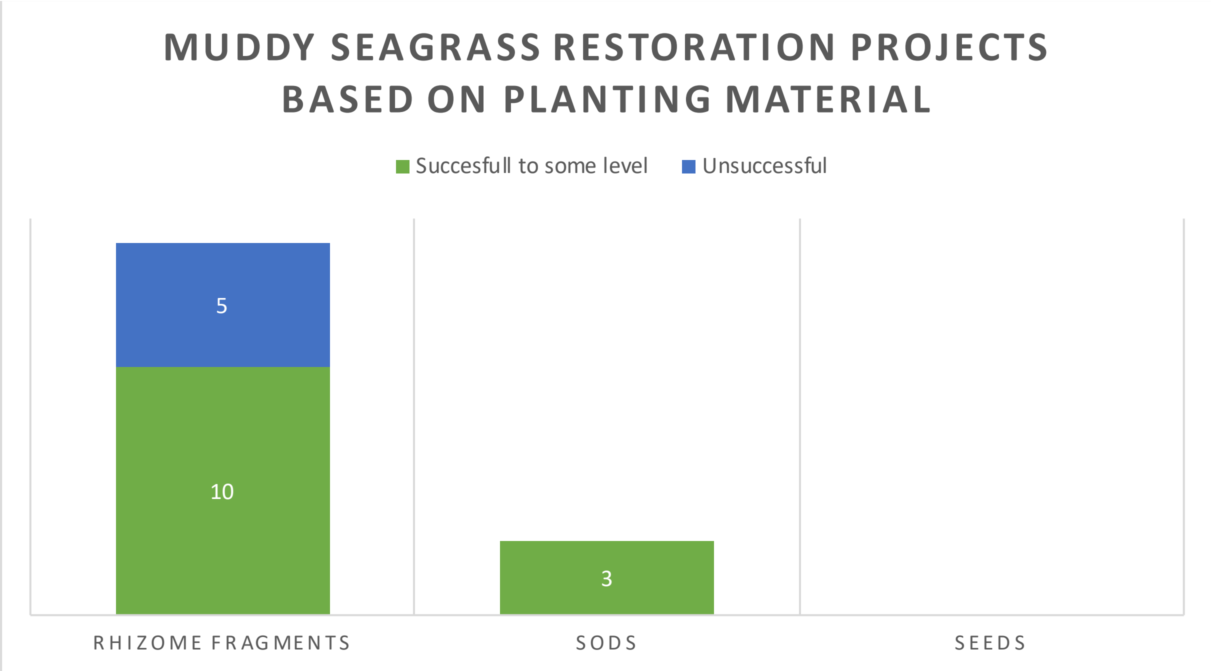


Figure 6.4: Level of success for muddy seagrass restoration projects found in literature.

6.3.2.3 Environmental conditions

Across-shore

15 of the 18 studies indicated whether restoration was done subtidal or intertidal. Figure 6.5 shows the number of projects which were placed subtidal or intertidal per planting material. It shows most restoration projects used rhizome fragments as planting material. Restoration projects were mainly performed intertidal. Successful restoration with rhizome fragments both has been performed subtidal and intertidal. Sods project were only implemented in the intertidal area. However, sods have been planted subtidal in sandy environments (Suykerbuyk, et al., 2016) and might therefore also be applicable in muddy substrate.

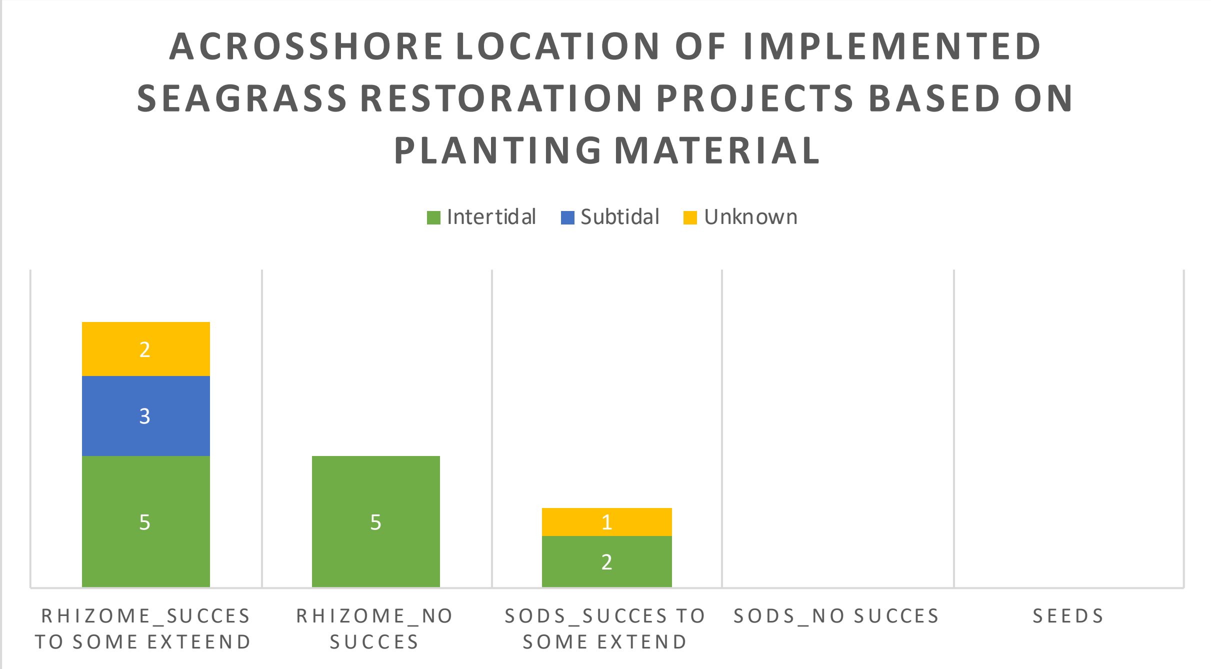


Figure 6.5: Number of seagrass restoration projects in muddy environments which were performed in the subtidal or intertidal area, based on planting material and successfulness of restoration. Two categories of successful restoration are considered: no success and successful to some extent (results based on Appendix 11.4.3).

Coastal type

Figure 6.6 shows along what type of coast the 18 found seagrass restoration projects were located. Restoration with some form of success in plant establishment was both observed along bay/estuaries and barrier coasts for both the planting materials rhizome fragments and sods. However, especially for sods the number of projects is limited.

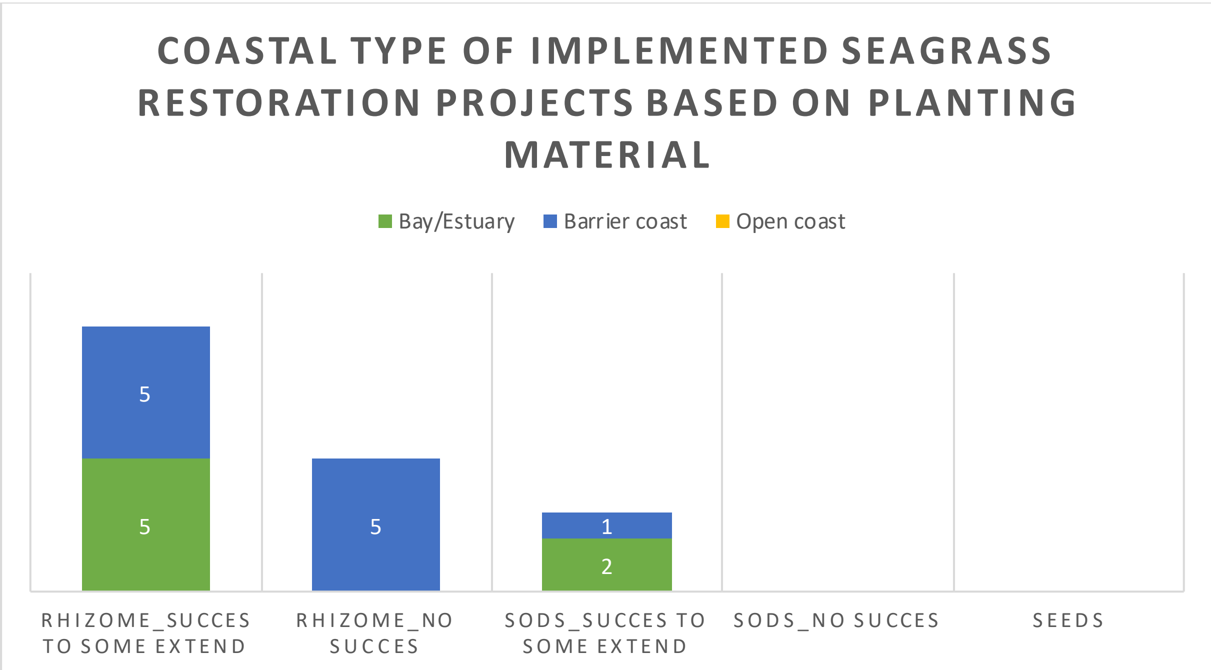


Figure 6.6: Number of seagrass restoration projects in muddy environments which were performed in three different coastal types, based on planting material and successfulness of restoration. Two categories of successful restoration are considered: no success and successful to some extent (results based on Appendix 11.4.3).

Habitat requirements
Environmental parameters like light availability and tidal range etc. (as mentioned in Section 6.2) of the restoration site must closely match those of the donor site if restoration is to be successful (Turner & Schwarz, 2006). This implies that donor material cannot be retrieved from every location.

When applying seeds as planting material it should be considered that large numbers of seeds can be consumed by seed predators (Park & Lee, 2007) and settling might be difficult in energetic environments with strong currents and high waves (Fonseca, et al., 1998). Applying seed as planting material might therefore only be applicable in low energy environments (Short & Coles, 2001).

Sods have the advantage that they are less susceptible to erosion and bioturbation than bare root planting units (Boyer & Wyllie-Echeverria, 2010). For more hydrodynamically rigorous settings planting large sods might be most appropriate. They may have sufficient integrity in comparison too small cores or bare root planting to prevent them from being quickly eroded away (Fonseca, et al., 1998).

6.3.2.4 Success & failure factors of implementation

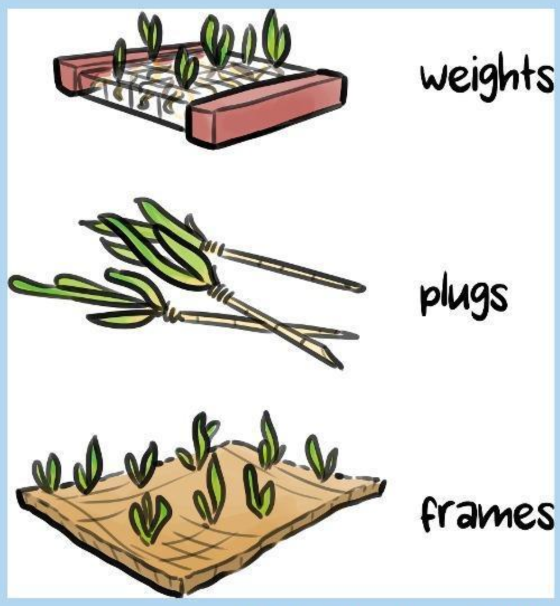
Sods
Although the root-rhizome-sediment system remains intact with the sod method, it will significantly impact the donor bed and holes must be filled to avoid erosion of adjacent areas of the bed. Furthermore, when the donor site is far away, transporting the material may present a problem as the weight poses a physical burden (Perrow & Davy, 2002; Boyer & Wyllie-Echeverria, 2010). This method is considered the most labor and cost intensive (compared to rhizome and seeds). Especially in the case of subtidal planting which requires SCUBA diving (Bulseco-McKim, 2012). Furthermore, in case of manual planting inexperience of planting personnel has been observed to reduce the success of seagrass establishment. Likewise, with mechanical planting when the planting installation is poor (Statton, et al., 2012).

Rhizome fragments with shoots
Adult plants must be removed from the donor bed to retrieve the rhizome. This is less damaging to the donor site compared to the sod method but still labor intensive, among others due to necessity of SCUBA diving in case of subtidal planting (Bulseco-McKim, 2012). Like with sods, inexperience of planting personnel and poor planting installation can reduce the success of seagrass establishment (Statton, et al., 2012).

Seeds
Seeds can be retrieved from a donor bed or laboratory. However, producing seeds in a laboratory is costlier compared to collection from donor beds (OCEANA, n.d.) and removing them from the donor bed will reduce natural recruitment at the donor bed (Bulseco-McKim, 2012). Furthermore, not all species produce seeds. For example, H. wrightii, one of the six most common tropical species, produces seeds only under extremely rare circumstance (Garvis, 2012). Using seeds might therefore not be applicable for every specie and be most suitable for species that produce seeds in large quantities on annual basis (Marion & Orth, 2010). Once seeds are collected, they can be sown quickly and easily over large areas. Limited experience to utilize seed material has been observed to cause failure in seagrass establishment (Statton, et al., 2012).

6.3.3 Planting technique: stabilizing

BOX 5 **Examples of anchoring devices for seagrass transplantation**



\# Example weights: seagrass attached to weighted TERFS.

\# Example plugs: seagrass attached to bamboo stakes.

\# Example frames: seagrass is attached to coconut hair mats

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Seagrass (rhizome fragments or seedlings) can be planted directly into the bed or anchored using a variety of devices of different types of weights, staples and frames (Table 6.1 and examples in Box 5) (Perrow & Davy, 2002).

Anchoring device	Implies
Weights	sand bags, stones/rocks, bricks, shells, TERFS
Staples	rods, bamboos, pegs, sprigs and washers
Frames	attaching the planting material to frames, girds, quadrats, nets, mats or meshes that are not weighted

Table 6.1: Different types of anchoring devices for seagrass restoration. TERFS=Transplanting Eelgrass Remotely with Frame System (Van Katwijk, et al., 2016).

6.3.3.1 Successfulness of restoration

The seagrass restoration review performed by Van Katwijk et al. (2016) showed that any anchoring (weights, staples, frames or using sods) will improve the initial survival of plants by 84% on average. The application of weights improved later success scores by 45%. Other anchoring methods like staples and frames did not contribute to later success scores. However, these findings only indicate averages and are non-specific for muddy environments. For example, a study performed by Park and Lee (2007) on a non-tropical specie in a muddy environment, showed different survival rates for different anchoring devices (2 years after planting) (Table 6.2). They found that using metal staples as an anchoring device gave the quickest initial establishment and highest long term seagrass survival rates compared to using weights (Oyster shells and TERFS[2]). This is in contrast with Van Katwijk et al.'s (2016) global analysis in which it was found that staples did not seem to contribute to long term survival and showed lower survival rates compared to weights (Figure 6.3). Van Katwijk et al.'s (2016) results concerning anchoring therefore might not be true for muddy environments. This might in part be

[2] Transplanting Eelgrass Remotely with Frame Systems

because applying weights might not be suitable on every muddy bed since heavy structures might sink on soft, muddy bottoms, prohibiting the transplants to survive (Almela, et al., 2004). Further research should give more insight in this for muddy substrates.

Method	Initial establishment after (average)	Survival rate after 2 years	Characteristics
Metal staple	1,5 months	75-95%	Labor intensive + need scuba diving for transplanting
Weight: TERFS	2,6 months	60-75%	Reduced amount of diving time + need removing frames after rooting time
Weight: Oyster shells	3,2 months	60-95%	Shells are dropped of boats

Table 6.2: Site-specific success of three transplanting methods of Zostera marina in the muddy Kosung Bay (Korea) (Park & Lee, 2007).

6.3.3.2 Environmental conditions

Four seagrass restoration studies were found which implemented anchoring with rhizome fragments as a restoration technique. Survival rates varied between 60-100% after respectively 4 months for one study and 2 years for the other three studies. Two studies applied staples, two weights (Appendix 11.4.3).

Across-shore

Figure 6.7 shows anchoring rhizome fragments have both led to successful restoration of seagrass beds in the subtidal and intertidal area.

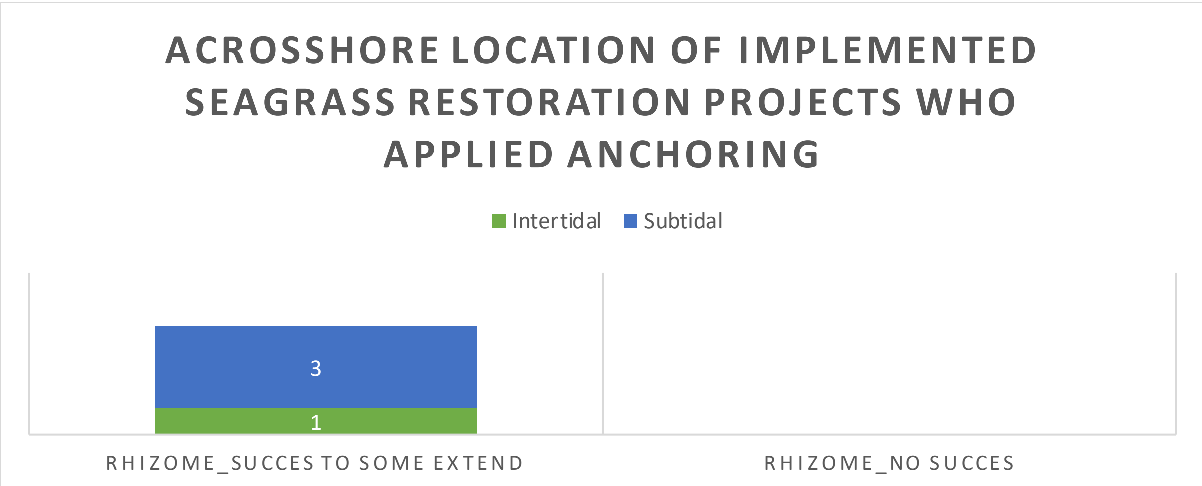

Figure 6.7: Number of seagrass restoration projects in muddy environments which applied anchoring with transplantation in the subtidal or intertidal area. Two categories of successful restoration are considered: no success and successful to some extent (results based on Appendix 11.4.3).

Coastal type

Figure 6.8 shows anchoring rhizome fragments has led to successful restoration in the coastal type bays/estuary and along barrier coasts. No projects were found along muddy open coasts.

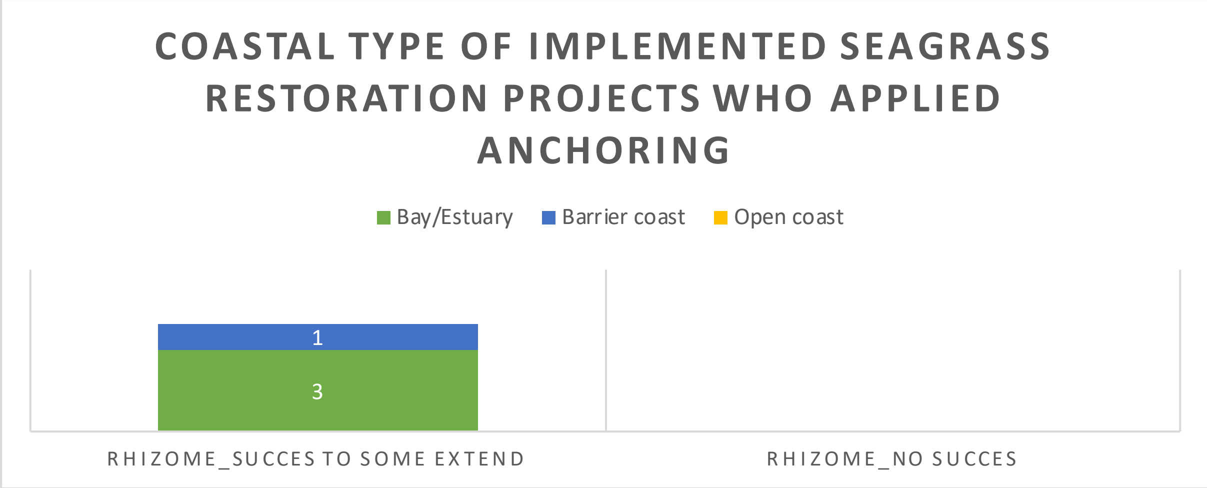

Figure 6.8: Number of seagrass restoration projects in muddy environments which applied anchoring with transplantation along different coastal types. Two categories of successful restoration are considered: no success and successful to some extent (results based on Appendix 11.4.3).

Habitat requirements

Knowlegde gap.

6.3.3.3 Success & failure factors of implementation

Park and Lee (2007) compared the implementation of staples, TERFS and shells. They found that applying seagrass with metal staples was most labor intensive and required scuba diving for transplanting, thereby increasing costs. They found that the TERFS method minimizes the amount of time and related costs of diving, however, the frames need to be removed after rooting time. Using shells as anchoring device was found to be most labor and cost effective. TERFS, and especially shells are thus considered more suitable for large scale rehabilitation. However, the success of TERFS is highly site-specific because it has been observed to attract bioturbators (Bulseco-McKim, 2012). Considering shells, the application is restricted by local availability. In case staples are chosen, bamboo staples (U- or L-shaped) might be more desirable because they are biodegradable, less expensive than metal staples (US $0,01 for each metal staple, US $0,006 for each bamboo staple) and often available in large quantities in tropical areas (Fonseca, et al., 1982; Thangaradjou & Kannan, 2008; Perrow & Davy, 2002). Furthermore, in calm areas, plants can be stapled to the bottom without attaching them to the staples beforehand, saving time (Fonseca, et al., 1998). In general, it thus seems that using weights as anchoring devices is less labor intensive then applying staples, among others due to the reduced amount of scuba diving time. No documentation was found of implementation of frames as anchoring devices on muddy sediments.

6.3.4 Compare restoration techniques

6.3.4.1 Restoration success & Environmental conditions

The average success rate of seagrass transplanting seems to be highest when sods or rhizome fragments with anchoring are applied as planting material, and lowest for rhizome fragments without anchoring, followed by seeds (Figure 6.3). Planting rhizome fragments without anchoring should therefore be avoided. Further research in muddy environments should give insight in which anchoring material (weights, staples or frames) is most suitable in terms of restoration success in different environmental settings. However, one study does suggest staples are more effective then weights.

Table 6.3 gives an overview of the limited number of muddy restoration projects in different environmental settings. No projects were found which implemented seeds as planting material or anchored rhizome fragments with non-weighted frames. Most projects include the planting material rhizome fragments. Seagrass has successfully been restored in the coastal types bays/estuaries and barrier, suggesting seagrass restoration is suitable for more sheltered conditions. However, further research is needed to indicate if seagrass restoration can also be successful along open coasts. Restoration with rhizome fragments was successful in the intertidal and subtidal area. Restoration with sods was successful in the subtidal area. Due to lack of data it is unknown if this is also true for the intertidal area. It is stated that applying seeds as planting material might only be applicable in low energy environments. For more hydrodynamically rigorous settings planting large sods might be most appropriate.

Seagrass restoration techniques	Environmental conditions					
	Across shore			*Coastal type*		
	Intertidal	*Subtidal*	*Unknown*	*Open coast*	*Bay/ estuary*	*Barrier*
Planting material (18 projects in total)						
• ***Seeds***						
• ***Sods***		2 (2)	1 (1)		2 (2)	1 (1)
• ***Rhizome fragments with shoots***	10 (5)	3 (3)	2 (2)		5 (5)	10 (5)
Rhizome fragments with shoots with anchoring						
• ***Weights***		2 (2)	-		2 (2)	
• ***Staples***		2 (2)	-		1 (1)	1 (1)
• ***Frames (non weighted)***						
• ***Unknown if anchoring was used***	9 (4)					9 (4)

Table 6.3: Overview of number of seagrass restoration projects in muddy environments implemented in different coastal types and locations across-shore. Of the total 18 seagrass beds, only one project was tropical, other studies were located in temperate areas. Numbers between brackets indicate a certain level of success was achieved during restoration (other projects were unsuccessful. Grey implies data is lacking (data based on Appendix 11.4.3).

6.3.4.2 Success & failure factors of implementation

Although restoration success is higher for seagrass projects which choose rhizome fragments and sods as planting material, the impact on the donor bed is lower for seeds as planting material (Seddon, 2004). Using sods and rhizome both rely on the use of adult seagrass plants which may lead to a possible loss in genetic diversity when removed in large quantities, making the donor beds more vulnerable to disturbances and climate change (Bulseco-McKim, 2012). Potentially plants can also be reared and grown in laboratories from plant fragments, however, this is costly. When donor seagrass is scarse and/or in case of large scale planting, seeds may thus be preferred as planting material (Ibid.; Seddon, 2004). However, this is only true when seeds are harvested from the donor bed, not when plants are harvested to release its seeds through buoy-deployed seeding. Furthermore, using seeds might only be applicable for species that produce seeds in large quantities on annual basis. Not all species produce seeds. In case ample of donor bed is available sods and rhizome fragments with anchoring can be chosen as planting material.

In terms of labor intensity and costs, seeds are the least labor intensive to implement and sods the most labor and cost intensive (among others due to the physical burden of the sods and the necessity of scuba diving in case of subtidal planting). Concerning anchoring, using weights is less labor intensive then using staples to attach seagrass to the bed. Mainly since weights can be dropped off a boat or placed on the soil, while staples need to be attached to the bed. Regardless of the planting material chosen, inexperience of planting personnel has shown to reduce the success of seagrass establishment (Statton, et al., 2012).

6.4 Erosion mitigation

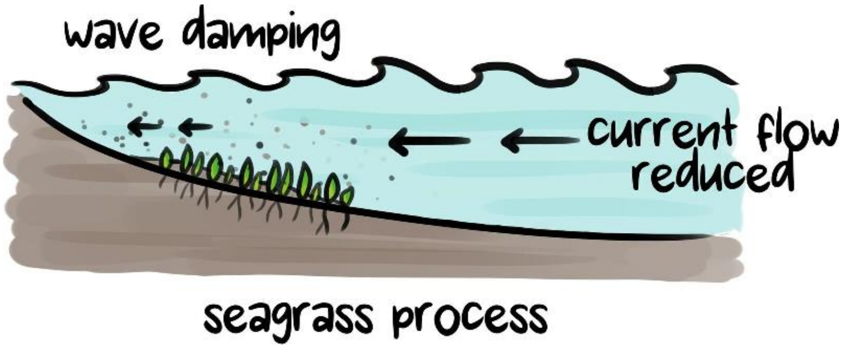

Figure 6.9; Conceptualized contribution of seagrass to erosion mitigation along mud coasts (Illustrated by Van Ginneken, 2017g).

In general, it is assumed in literature that submerged seagrass beds significantly influence the hydrodynamic environment by reducing current velocity within the bed and dissipating wave energy, thereby altering sediment dynamics. Sediment deposition and stabilization in the near- or foreshore is influenced by sediment trapping and flow velocity reduction provided by the above-ground biomass, rhizoidal and root system (Ondiviela, et al., 2014) (Figure 6.9). However, documentation of the efficiency of the erosion mitigation services provided by different seagrass species is poorly done (De Boer, 2007), especially for seagrass growing on muddy beds.

Figure 6.10 shows erosion mitigation services of different seagrass beds that have been observed along mud coasts. It is based on 5 studies with different species. 2 studies (5 seagrass beds) are tropical. No mention was made of whether these beds were natural or transplanted. Studies included qualitative and quantitative data from field, flume and model studies. Studies only indicated the level of one or maximum two erosion mitigation services provided. No study exists which measured all 5 erosion mitigation services which seagrass beds have been suggested to offer. The limited available data suggests seagrass can aid in mitigating erosion. However, not in every situation and time of the year. For example, the contrasting results for bed elevation can be explained by a difference in specie and biomass: structural small species with low biomass were unable to trap and accumulate sediments. Large species with high biomass were (Mellors, et al., 2002). Mellors et al. (2002) field measurements however do not give an indication of absolute bed elevation. Halley et al (2000) indicated that seagrass bed accumulated sediments of up to 2,5 cm/year in sheltered parts of Florida Bay (USA). Although the small species in Mellors et al. (2002) study do not contribute to sediment accumulation, this does not exclude their potential in stabilizing mud beds. Christianen et al. (2013) for example found that even intensively grazed subtidal seagrass meadows with a very short canopy (growing on sandy substrate) could stabilize sediments effectively compared to bare soil conditions. Further research should give insight in whether this is also true for seagrass on muddy soils. The contrasting results concerning bed stabilization cannot be explained due to incomparable settings of the locations. Flow reduction only occurred at low velocities. One flume study measured a 18% reduction in current velocities at 0,05 m/s, and 8% reduction at 0,25 m/s (Prager & Halley, 1999). Field measurements indicated a reduction of up to 39% at 0,64 m/s (Hasegawa, et al., 2008). Only one study observed wave attenuation. Wave measurements and modelling indicated wave damping of up to 80% in the presence of intermediate and dense seagrass along the mudflat's upwind edge (Prager & Halley, 1999). However, the study does not mention if this influences the erosion inducing small waves along the shoreline. No documentation was found to indicate whether seagrasses can aid in reduced coastline retreat. In general, the provision of erosion mitigation services, might be seasonal due to differences in biomass, which was true for the non-tropical studies which considered bed stabilization and current velocity. This effect might be eliminated in the subtropical areas (Figure 2.1). However, seasonal effects might be observed in coastal areas located in the alternating tropical zone (Figure 2.1), as even near the Equator seagrasses have shown seasonal growth variation (Short, et al., 2007).

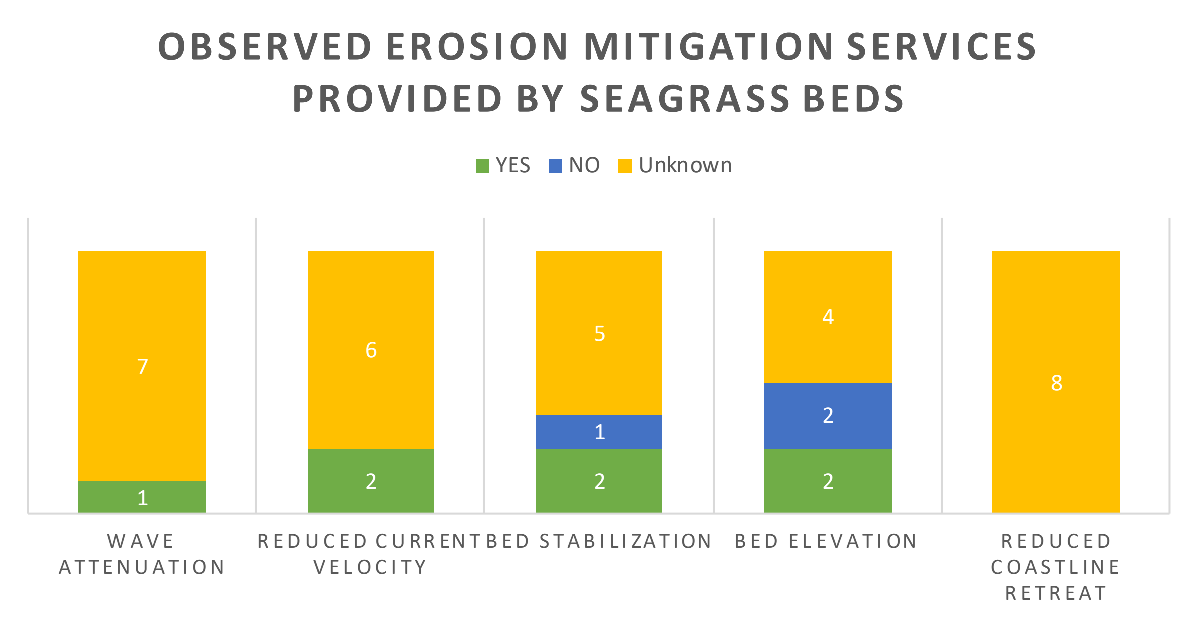

Figure 6.10: Erosion mitigation services provided by 8 seagrass beds based on the results of 5 studies. 5 Seagrass beds are tropical, 3 non- tropical. Studies include field and flume studies and modelling. Results are based on Appendix 11.4.4. Results of one study which indicated reduced current velocity mainly show reduced current velocities at low flow. Wave attenuation was observed in general, but not specified if this also referred to onshore small waves which induce erosion.

Due to the limited data, it is difficult to say whether seagrass species can provide all services. In general, it is found that seagrass beds cannot protect shorelines in every location and/or scenario. The efficiency of the services provided depends largely on the incident hydrodynamic energy flux, density, standing biomass, plant stiffness and leave length (Ondiviela, et al., 2014). Furthermore, current velocities are more efficiently reduced when seagrass occupies the entire water column. When water depth is greater than the maximum meadow height, wave attenuation is less efficient, and sediment both is deposited and resuspended (Widdows, et al., 2008). The optimal conditions for enhancing the erosion mitigation services seagrasses provide might therefore be in shallow waters and low to moderate wave energy environments. Combined with high interaction surface in the vertical and horizontal dimension between water flow and seagrasses (Ondiviela, et al., 2014). Other factors which influence the defense services include the seagrass distance from the shore, the beach slope, the reproductive stage and the tidal stage (Barbier, et al., 2011). Furthermore, seasonality of seagrass growth or random variation of standing biomass modifies wave attenuation (Ondiviela, et al., 2014). In general, it can be stated that large, long living and slow growing seagrass species, with biomass and density largely independent on seasonal fluctuations and with the maximum standing biomass achieved under the highest hydrodynamic forcing might be most favorable for mitigating coastal erosion (Ondiviela, et al., 2014). However, although seagrass meadows might aid the best in coastal protection in shallow waters and low to moderate wave energy environments, Van Katwijk et al. (2016) analysis on the relation between planting depth and restoration success, showed that lowest success rates are found for shallow depth (<0,5 m), especially in intertidal areas, due to wave dynamics.

6.5 General success factors

Technical

1. Restoration success increases with proximity to and recovery of donor beds. This indicates the suitability of the environment for seagrass growth and increases its recovery potential. The closer the distance from the donor site, the higher the chance of successful restoration (see Figure 6.11) (Van Katwijk, et al., 2016).
2. Large scale planting has been observed to be more successful in restoration due to the positive relationship between the number of plants or seeds initially transplanted and the tri al survival and seagrass population growth rate. There seems to exist a threshold scale required for restoration progress between 1000 and 10.000 shoots/seeds. However, large scal e planting is costly due to extracting of donor material and operational costs. Regained ecosystem services may compensate these investments costs (Ibid.).

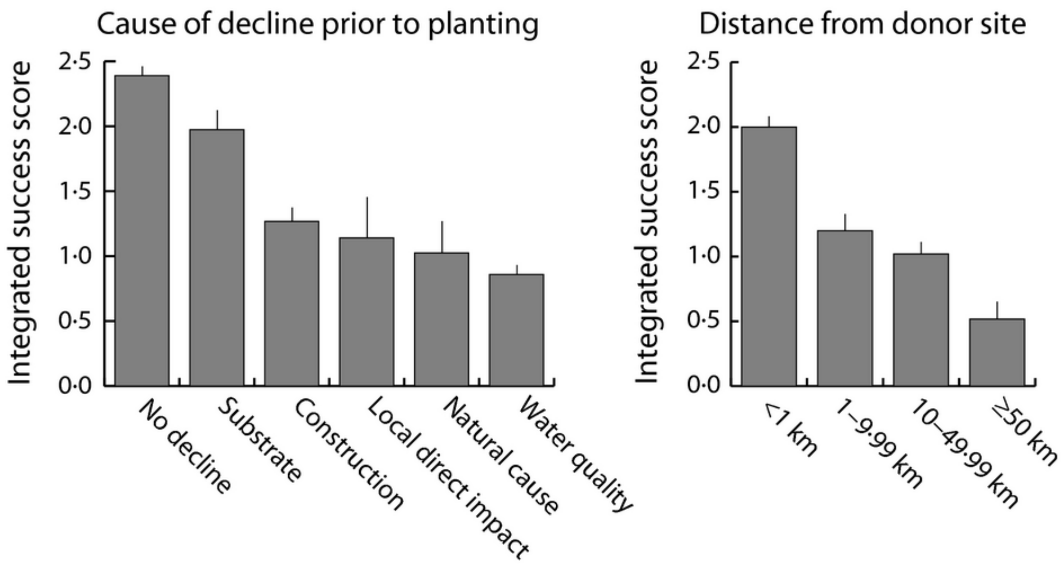

Figure 6.11: Performance of seagrass restoration trials in relation to degradation prior to planting and distance from donor site. Cause of degradation include among others substrate manipulation like dredging and filling, construction activity or reduced water quality (from Van Katwijk, et al., 2016).

Environmental

Seagrasses are often the dominant primary producers in coastal areas, forming a direct source of food for dugons, sea turtles and parrot fish and are closely linked with high fisheries production due to the critical nursery habitat they provide (Unsworth, et al., 2014)

Social

Knowlegde gap.

Economical

Knowlegde gap.

6.6 General failure factors

Technical

1. Both successful regrowth of seagrass species and wave attenuation requires crossing a minimum density of reintroduced individuals, which must be determined per specie (Van Katwijk, et al., 2016; Ondiviela, et al., 2014).

2. Seagrass meadows are vulnerable to external impacts, which makes their durability questionable. Trends of climate change related factors like seawater warming, increasing storms and sea level rise, together with growing populations and anthropogenic threats in coastal areas may impact seagrasses to an extent that disables them to adapt and maintain their erosion mitigation services. Sea level rise for example will cause an increase in water depth which among others reduces the availability of light at the bed. Anthropogenic threats include for example mechanical damage of meadows (e.g. dredging and construction activities), deterioration of water quality due to urban/industrial/agricultural runoff or introduction of invasive species (Ondiviela, et al., 2014). Anthropogenic actions form the greatest threat (Grech, et al., 2012). Van Katwijk et al. (2016) found that especially reduced water quality (mainly eutrophication) and construction activities led to poorer restoration success than factors like dredging, local direct impacts and natural causes (see Figure 6.11). This indicates the vulnerability of seagrass to external factors. It is therefore recommended to remove threats which caused the degradation of habitat prior to restoration (Ibid.). But also to avoid these impacts on the long term to ensure the durability of this erosion mitigation measure. These activities are largely terrestrially based, which implies the importance of combining coastal planning with adjacent watershed planning (Grech, et al., 2012).
3. Other observed reasons causing poor seagrass establishment during transplantation project are numerous. They include among others slow growth and poor root development of some seagrass species, bioturbation, grazing, hydrodynamics, salinity fluctuations, erosion, sediment deposition, insufficient anchorage, disease, poor water quality etc. (Statton, et al., 2012; Paling & van Keulen, 2002).

Economical

1. Restoring seagrass meadows is expensive. Cost of restoration are determined by different components of restoration, like planning, purchasing, land acquisition, planting, maintenance, monitoring, and equipment repair/replacement (Bayraktarov, et al., 2016) . Cost may vary with planting technique, project area, project duration, and increase due to factors like inappropriate site selection, inexperience in planting, disturbance events like bio perturbation and storms, low water visibility, increased water depth (related to SCUBA diving), etc. (Calumpong & Fonseca, 2001). Consequently, costs of seagrass restoration projects vary largely. A systematic review of the cost of coastal restoration by Bayraktarov et al. (2016), found average total cost of 700.000 US$ per hectare and median cost of 384.000 US$/hectare for seagrass restoration projects (Table 6.4). All projects were small scale (<20 ha) and located in developed countries. Cost may be lower in developing countries and for large scale projects. No distinction was made concerning the soil type of the restoration project. However, these findings due give an indication of average costs.

Restoration cost		Total restoration cost	
N	2010 US $	N	2010 US $
64	107.000 (400.000)	22	384.000 (700.000)
64	107.000 (400.000)	22	384.000 (700.000)

Table 6.4: Median (and average in brackets) values of restoration/rehabilitation cost per hectare represented in 2010 US dollars. N= number of observations. 'Total restoration cost' implies projects which both included capital and operating costs. 'Restoration cost' include observations which did not specify what costs included. Findings indicate general restoration cost, regardless of soil type (from Bayraktarov, et al., 2016).

Environmental

Knowledge gap.

Social

Knowledge gap.

6.7 Conclusion

6.7.1 Successful Nature Based Solution?

6.7.1.1 Mitigating erosion

Implementing seagrass as a NBS implies both successful restoration as well as the provision of erosion mitigation services. Seagrass beds on muddy substrate have been observed to contribute to mitigating erosion by providing the following services: wave attenuation, reduced current velocity at low flow and bed stabilization and elevation. However, it is unclear if this wave attenuation also occurred along the shoreline, and thus if it contributed to mitigating erosion. No documentation was found to indicate whether seagrass beds can aid directly in reducing coastline retreat. However, these findings are based on a limited amount of studies (8) and should thus be considered with care. Services provided might be subjected to seasonal variation, even in tropical areas, and not applicable for every specie in every location and/or scenario due to a wide variability in factors both related to plant and bed characteristics, hydrodynamic conditions and physical settings. Large, long living and slow growing seagrass species, not subjected to seasonal fluctuations, growing in shallow waters and low to moderate wave energy environments might be most successful in protecting shorelines. However, lowest restoration successes are found for shallow depths (<0,5m), especially in intertidal areas, due to wave dynamics. All this indicates the uncertainty of implementing seagrass as erosion mitigation solution and the need for further research.

6.7.1.2 Restoring seagrass

Seagrass has been restored with some level of success in the muddy coastal type bay/estuary and barrier coast. No restoration projects were identified for open mud coasts. Potential restoration techniques imply three different planting materials: seeds, sods or anchored rhizome fragments with shoots. Due to a lack of implemented projects (18), successful restoration in mud areas has only been observed for rhizome fragments in the intertidal and subtidal area, and for sods in the intertidal area. No projects were found which used seeds as planting material or applied sods in subtidal muddy areas, making it unclear if this option can be successful along muddy coasts. Anchoring can be done with weights, staples or non-weighted frames made of different materials. Anchoring rhizome fragments to staples seems to be more successful in terms of survival rate compared to weights. However, transplanting seagrass with staples is more labor intensive than weights. No studies were found which implemented frames. A non-soil specific relative success scale of seagrass restoration, suggests restoration with anchored rhizome fragments and sods give higher success rates compared to seeds. Confirmation of this for muddy coasts is yet needed. Because monitoring was mostly less than a year, no statement can be made of the sustainability of this NBS solution on the long run.

6.7.2 Success and failure factors of implementation

6.7.2.1 General

The chance of successful seagrass restoration increases with proximity to and recovery of donor beds, with large scale planting when a threshold scale between 1000 and 10.000 shoots/seeds is exceeded and when a minimum density of reintroduced individuals (species depended) is crossed.

General failure factors include the vulnerability of this NBS to external impacts like climate change and in especially anthropogenic threats. Climate change related impacts include seawater warming, increasing storms and sea level rise. Anthropogenic threats include mechanical damage of meadows (e.g. dredging and construction activities), deterioration of water quality due to urban/industrial/agricultural runoff and/or introduction of invasive species. When these impacts are not removed (before restoration and on the long term), they might unable seagrass to adapt and maintain their erosion mitigation services. Because these activities are largely terrestrially based, it is recommended to combine coastal planning with adjacent watershed planning. Other observed reasons causing poor seagrass establishment during transplantation project are numerous. They include among others slow growth and poor root development of some seagrass species, bioturbation, grazing, hydrodynamics, salinity fluctuations, erosion, sediment deposition, insufficient anchorage, etc.

Restoring seagrass meadows is expensive and vary widely due to variation in planting techniques, project area and project duration, and increase due to factors like inappropriate site selection, inexperience in planting, disturbance events like bio perturbation and storms, low water visibility, increased water depth (related to SCUBA diving), etc. Average total cost (non-soil specific) are 700.000 US$/hectare and median cost 384.000 US$/hectare for small scale projects (<20 ha) located in developed countries. Cost may be lower in developing countries and for large scale projects. Costs of seagrass restoration projects vary largely

6.7.2.2 Restoration techniques

The pros and cons of the three type of planting materials are shown in Table 6.5. Choice of material can be made on local conditions or preference. However, when the energetic conditions are not restricting and ample of donor bed is available, it is advised to choose anchored rhizome fragments as restoration technique since this appears to have a higher restoration success rate than seeds (and sods, depending on anchoring technique), and is less labor intensive, costly and impacting to the donor site compared to sods. No information is available concerning success and failure factors related to the lifespan of different techniques.

Seeds	**Anchored rhizome fragments with shoots**	**Sods**
Pros: • Least labor and cost intensive • Least impact donor bed • Potentially best option for large scale planting	*Pros:* • Intermediate labor and cost intensive • Most implemented, thus most experience	*Pros:* • Large sods can be applied in more energetic environments
Cons: • Lowest survival rate • Only in low energy environments • Not all specie produce seeds (in large enough quantities)	*Cons*: • Only possible when ample donor bed available • Intermediate impact donor bed	*Cons*: • Most labor and cost intensive • Only possible when ample donor bed available • Highest impact donor bed
Successful restoration on muddy substrate: • No implemented projects found	*Successful restoration on muddy substrate:* • Across-shore: intertidal + subtidal • Coastal type: bay/estuary, barrier coast	*Successful restoration on muddy substrate:* • Across-shore: subtidal + intertidal? • Coastal type: bay/estuary, barrier coast

Table 6.5: Pros and cons which can be considered when choosing a technique to restore seagrass beds. No quantitative definition of low and more energetic environments is available. Successful restoration implies some form of success has been achieved. Seagrass restoration project where not executed along open coast.

7 Comparing NBS

7.1 Erosion mitigation

Table 7.1 shows the erosion mitigation services that have been observed to be provided by muddy seagrass beds and mud recharge schemes. Both are based on a limited number of studies. For mud recharge, the differences in success in erosion mitigation services provided can potentially be explained by an unsuccessful restoration technique (unconfined intertidal) and for seagrass in part due to differences between species characteristics. However, reasons remain uncertain due to the limited number of studies.

All erosion mitigation services applicable to mud recharge have been observed (wave attenuation, bed elevation and reduced coastline retreat). Available quantitative field data indicated wave attenuation of 29% and 46% and stable bed elevations of 1,1m and 0,03-0,3m after 2 years. Concerning seagrass, all erosion mitigation services applicable to this NBS were observed except reduced coastline retreat. Available quantitative data indicate bed elevation rates of up to 2,5 cm/year and reduced current velocity at low flows of 18% at 0,05 m/s, 8% at 0,25 m/s and up to 39% at 0,64 m/s. Wave damping was observed of up to 80%, However, it is unknown if the observed wave attenuation also referred to the erosive inducing small waves along the shoreline. Making a quantitative comparison between the mitigation services provide by mud recharge schemes and seagrass is difficult due to the incomparable settings. For example, it is unknown under what wave conditions the observed wave attenuations were measured. Also, differences in bed elevation levels might be explained by differences in sediment delivery to locations.

Concerning seagrass beds, it should be considered that services provided might be subjected to seasonal variation and that they are not applicable for every specie in every location and/or scenario. Furthermore, the provision of mitigating services provided by seagrass is only possible when restoration is successful on the long run. This is not always achieved. This suggests that mud recharge might be a more reliable NBS to mitigate erosion than seagrass. However, maintenance might be necessary for mud recharge, while a healthy seagrass bed may remain for years when not degraded by external impacts.

NBS	Wave attenuation	Reduced current velocity	Bed stabilization	Bed elevation	Reduced shoreline/coastline retreat
Mud recharge	YES (1), MAYBE (1)	n/a	n/a	YES (7) NO (1)	YES (1)
Seagrass	MAYBE (1)	YES (2, at low flow)	YES (2) NO (1)	YES (2) NO (2)	?

Table 7.1: Erosion mitigation services observed by two NBS. Numbers between brackets indicate the number of studies the results are based on. MAYBE=wave attenuation has been observed, but no mention was made if this referred to small waves along the shoreline which mainly induce coastal erosion. ?= unknown

7.2 Environmental conditions

An overview is given of the restoration techniques of mud recharge and seagrass for the coastal type bays and estuaries in Figure 7.1 and barrier coasts in Figure 7.2. These solution frameworks give insight in whether techniques were implemented with some level of success on a specific coastal type. For seagrass restoration, success revers to successful transplantation. For mud recharge this revers to successful erosion mitigation. Implemented NBS along open coasts were not found. Thus, this study cannot provide a solution framework for this muddy coastal type. All seagrass restoration projects were located in bays and estuaries or along barrier coasts. All implemented mud recharge projects were in bays or estuaries. However, since back-barrier areas can be considered less exposed to waves than estuaries and bays (Daidu, et al., 2013), it could be assumed that mud recharge projects might be successfully implemented along barrier coasts too.

Depending on the restoration technique chosen, mud recharge and seagrass restoration have both been done successful in the subtidal and intertidal zone. It has been suggested that direct placement techniques can be applied in low to moderate wave energy environments (definition unknown). Requirement of other environmental conditions have not been found. The latter is also true for trickle charge and sediment stirring. This makes comparison difficult. However, seagrass species might also be most successful in protecting shorelines in low to moderate wave energy environments. These results suggest mud recharge and seagrass as NBS might be most suitable in more sheltered conditions. However, this should be taken with caution since experimentation along open mud coasts and under more energetic environments have not been performed yet.

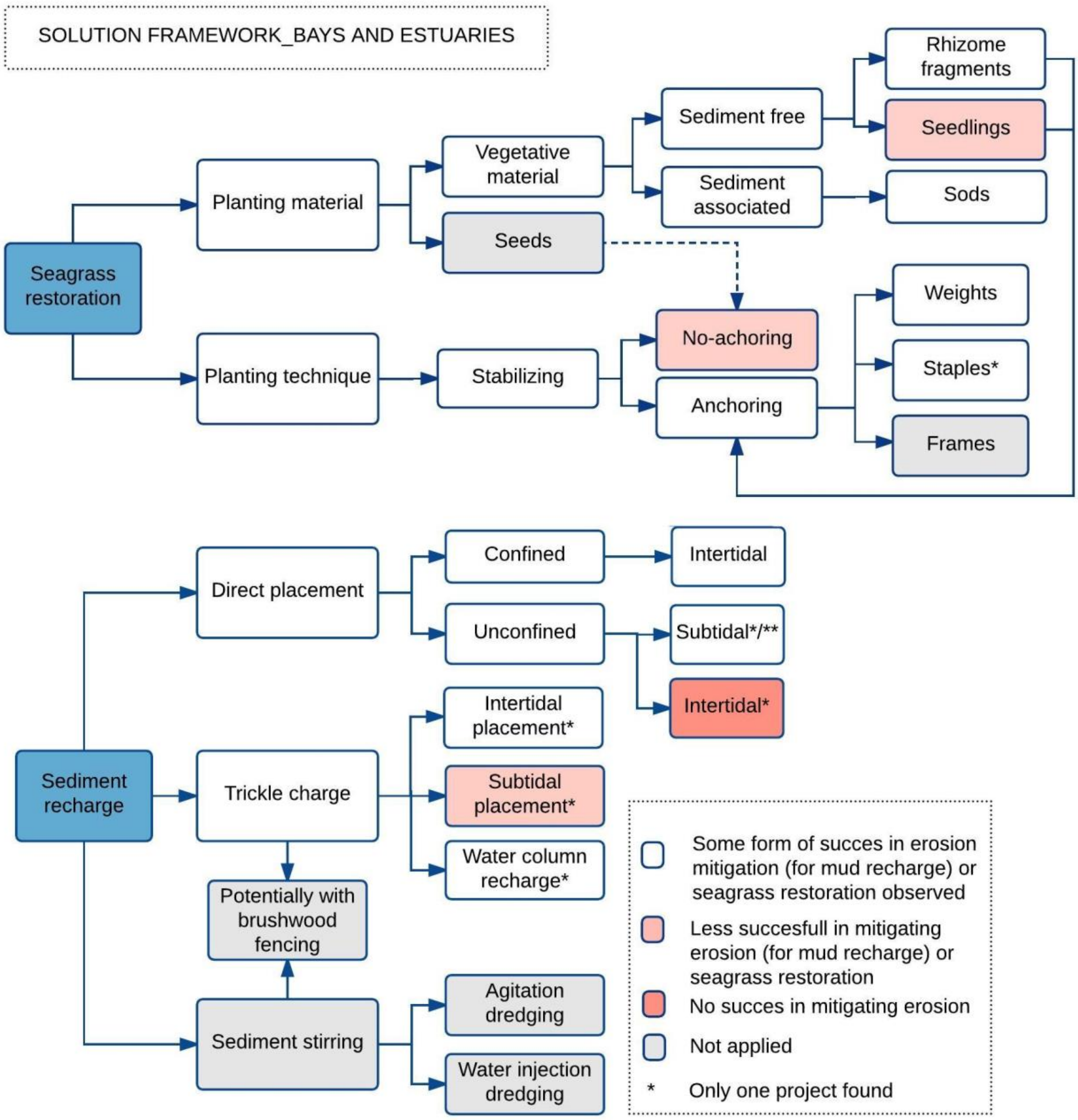

*Figure 7.1: Solution framework showing restoration techniques of mud recharge and seagrass for the coastal type bays and estuaries. Pros and cons of mud recharge techniques can be found in Table 5.2 in Section 5.6. Pros and cons of seagrass restoration techniques can be found in Table 6.5 in Section 6.7. **=wave attenuation observed. However, it is not specified if this also referred to onshore small waves which mainly induce erosion.*

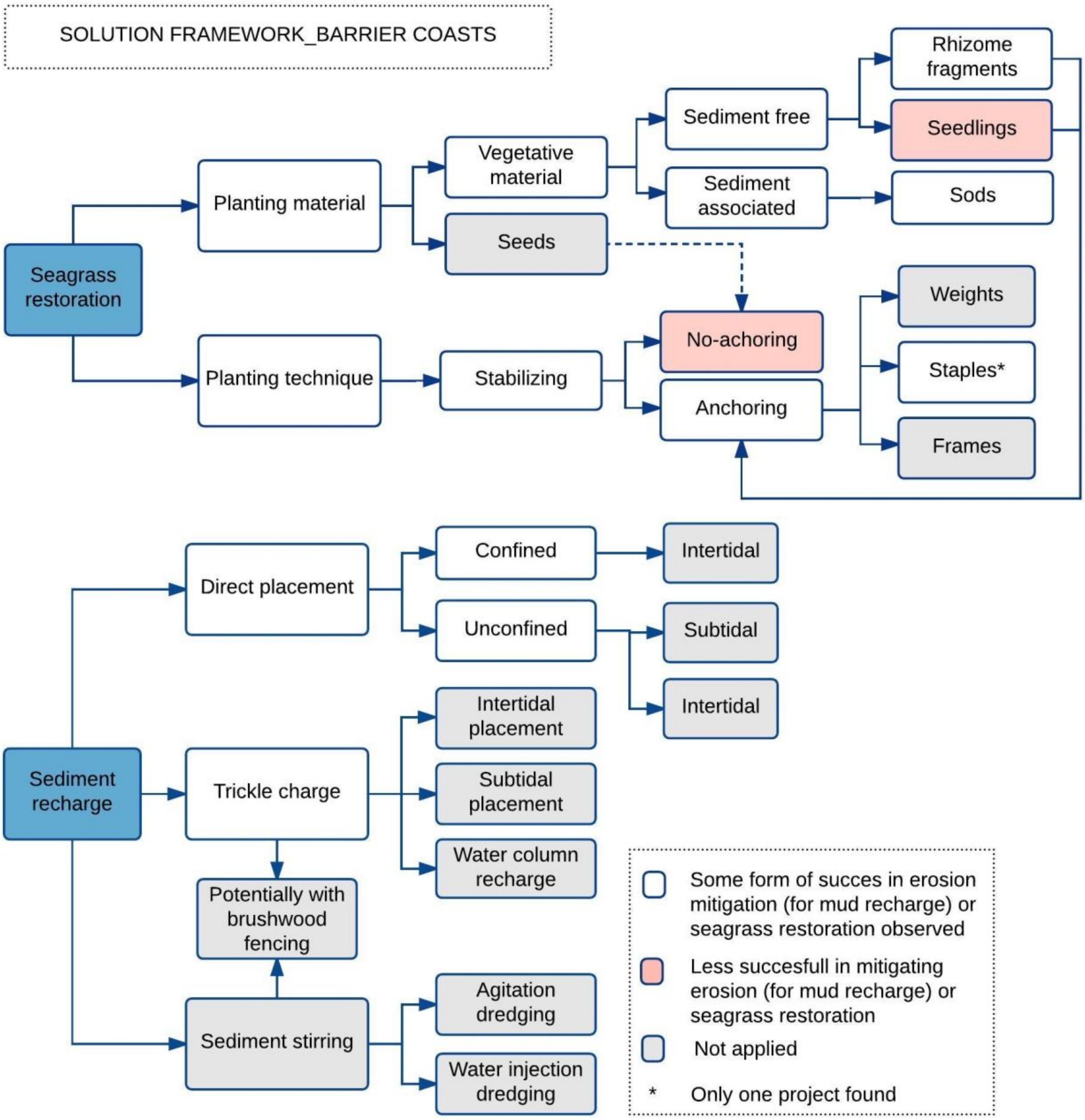

Figure 7.2: Solution framework showing restoration techniques of mud recharge and seagrass for the coastal type barrier coast. Pros and cons of mud recharge techniques can be found in Table 5.2 in Section 5.6. Pros and cons of seagrass restoration techniques can be found in Table 6.5 in Section 6.7.

8 Discussion

8.1 Reliability of the results

8.1.1 Definitions

Terms used in documentation are not always clearly defined. For example, mud has no precise definition in the sense of the percentage of cohesive sediments present in the soil. However, so called mud often contains a high percentage of sand (Burt, 1996). Consequently, implemented NBS projects found in this study do not pertain equal percentages of mud particles in the soil. Morphological processes might therefore differ, which might impact the successfulness of the erosion mitigation potential of the restoration techniques. However, data is considered adequate for a general overview study like this.

Also, in indicating under what environmental conditions NBS can be implemented, terms like 'low' and 'moderate' energy environments are used in literature. Not specifying what this means in for example tidal current speeds or wave energy and/or height.

Furthermore, in this study restoration techniques were identified with the search terms 'restoration', 'rehabiliation' and 'creation'. However, in case a difference exists in techniques between these three terms, this could reduce the reliability of the results. However, no indication was found in literature to assume this, and it is therefore considered of having little influence.

Another discussion point of unreliability of the sources are the non-scientific (and not peer reviewed) sources. These 'grey' sources can provide useful information but are not always completely objective.

8.1.2 Erosion hotspots

A few short comings can be identified related to the identification of hotspots with the Aquamonitor. First, when relative sea level rise is the driver of the conversion of land into water, then this can be caused both by erosion processes and long term inundation. Since it is impossible yet to make distinction between inundation and erosion this might have caused an overestimation of the number of erosional hotspots in this study. Church and White (2011) for example, found an average global sea level rise for 1993–2009 of 3.2 ± 0.4 mm/year from satellite data and 2.8 ± 0.8 mm/year from in situ data. Although modern satellite records have shown that the sea level does not rise uniformly around the globe, this might have caused inundation along shallow coasts, a common characteristic of mud coasts (Healy, et al., 2002). Local natural or anthropogenic induced land subsidence only adds to this. Whether this observed inundation is significant compared to erosion trends depends on the gradient of the coast. However, no representative data was found concerning mud coasts gradients. A simplified calculation is given in Table 8.1 to indicate the potential significance of inundation on identified erosion hotspots in this study. Identification of coastal gradients and sea level rise at coastal sites could give insight in this relation.

It is unlikely that he suitability of mud recharge and seagrass as a NBS is affected by whether the conversion of land into water is driven by inundation and/or erosion. These NBS can aid in all scenarios due to the bed elevation services they provide.

Knowledge concerning the presence of mud along coasts is not always known. For example, except for some detailed local studies, little is known concerning the coast of Cambodia and Indonesia (Flemming, 2002). Also, the scale on which hotspots are identified on the Aquamonitor (1:1.000.000) implies hotpots could been missed which might have been visible on a larger scale. It is likely these factors have caused an underestimation of the number of hotspots.

Potential gradients of mud coast	Horizontal inundation (perpendicular to the coast)	Inundation % of a 10 km coastal erosion stretch (in alongshore direction)
0,5%	18 m	1,8 %
0,2%	45 m	4,5 %
0,1%	90 m	9 %
0,05%	180 m	18 %

Table 8.1: Simplified calculation of the percentage of alongshore areal extent of erosion hotspot which could be caused by inundation, depending on the coastal gradient. It assumes a 3 mm global sea level rise per year over a 30 years period. This implies a 90 mm level of inundation over this period. This 30 year period is chosen since data from the Aquamonitor indicates 30 years of data. The 3 mm per year is taken as simplified average from Church and White (2011).

For the northern South-American coast between the Orinoco and Amazon river, data was only available for a 15 to 23 year period. However, this coastline is characterized by strong erosion and accretion phases, influencing the location of the coastline. Consequently, the 17 hotspots located along this coast have been excluded from this study. However, this coast is the largest muddy coastline of the world. Not including this coast for identification of erosion hotspots implies comparison between the number of hotspots per continent could be skewed. This will not change the outcome that along the African and Australian coast the number of hotspots are lower compared to the other continents. However, it makes comparison between North- and South-America and Asia difficult. The overall outcome of comparing the number of hotspots between different coastal types is not influenced. Most hotspots (51%) are already located along open coast. Potential erosion hotspots which would appear along the South-American coast when 30 year satellite data would be available would increase the number, but not change the overall outcome: that most hotspots are located along the open coast type in comparison to bay/estuaries and barrier coasts.

Furthermore, the extent of the mud migrating bank systems in other coastal areas will not reach the magnitude of the Amazon related banks, creating some unreliability in identified hotspots. This depends however on the extent of the banks, their periodicity and calculation method of land-water conversions in the Aquamonitor. However, no documentation was available explaining how the calculation of the Aquamonitor was performed. For example, are the land-water conversion averaged over time or do they show a difference between 1985 and 2015? Also, did they account for the tide? Or consider the potential presence of large storms which can cause erosion on short time scales (Wong, et al., 2014)? All these factors should be taken into account when indicating long-term land-water conversions. This indicates the need for further insight in the setup of the Aquamonitor, but also the importance of looking at specific hotspots and all the potential drivers of coastal erosion working on this area to determine long term erosion trends.

8.1.3 Mud recharge

Although only 9 implemented projects of mud recharge were found in this study, it is likely more mud schemes have been executed. The US Army Corps of Engineers for example has 30 years of experience of using dredged material for wetland restoration and creation. However, it is not always mentioned what type of sediments are used in documentation of recharge schemes (ABP Research, 1998; Colenutt, 1999c). If so, most recharge schemes seem to be done with sandy material. Recharge projects which applied muddy material might therefore have been missed. Furthermore, according to Fletcher (2008) the most widely applied beneficial use of muddy dredged material to create or enhance mudflats (within the UK) is trickle charge and agitation dredging. It is also stated that small scale experiments of fine-grained intertidal recharge have been undertaken (Schratzberger, et al., 2006; Fletcher, et al., 2000). However, documentation of these schemes is mostly not available or accessible and monitoring is often lacking. Fletchers (2008) findings are in contrast with this studies results in which it was found that direct placement was performed most (6 times), trickle charge 3 times and no implemented projects were found of sediment stirring (agitation dredging and water injection dredging). It seems therefore likely that more muddy recharge schemes have been executed than results suggest in this study. Furthermore, in case monitoring is performed, it often only includes ecological effects. This reflects the phenomena that most recharge schemes are undertaken for habitat and wetland restoration purposes, and not necessarily for coastal protection (ABP Research, 1998). Consequently, documentation does not always mention whether the recharge scheme was successful in mitigating erosion.

Thus, little is known concerning the implementation, effectiveness and impacts of mud recharge for erosion mitigation. In the Netherlands, pilot studies are still being performed to better understand physical and ecological processes of mud nourishment. For example, a pilot project started in 2014 along the port of Harlingen in the Wadden Sea, where mud dredged from the port is disposed close to the shore as a subtidal trickle charge to increase coastal protection and/or prevent drowning of salt marshes under relative sea level rise. It is excepted that tidal flows will transport the material to the intertidal and salt marsh zones, forming a 'Mud motor'. Dredging and dumping will be done over a period of 8 months for three consecutive years which differs from the 'Sand engine' along the Dutch North Sea coast, where a large volume of sand was placed once (Eekelen, et al., 2016).

8.1.4 Seagrass restoration

In this study, it is assumed that the success of restoration techniques for seagrass transplantation is independent of soil type. However, this should be taken with caution. A study for example by Park and Lee (2007) showed that anchoring seagrass to shells was effective for muddy seabed's (survival rate after 2 years approximately 60-95%), but ineffective for sandy beds (survival rate <5%). However, for other restoration techniques (attaching seagrass to staples and frames) successful transplantation was both observed on muddy and sandy soil. No mention was made of the hydrodynamic regime in the different experimental settings, making it difficult to indicate whether the ineffectiveness of the shells as anchoring was related to the soil type, the hydrodynamic conditions, a combination of both or some other factor. Due to lack of data concerning this dependency, Van Katwijk et al. (2016) global analysis is considered a good starting point to indicate restoration techniques for muddy coasts. However, further research is needed to confirm this.

Like with mud recharge, the number of implemented seagrass restoration projects found in this study is probably an under estimation of reality. Two reasons can be distinguished. Firstly, soil type is often not mentioned in seagrass studies. Secondly, due to time constrains specific species names were not entered in search engines when looking for data.

It is important to note that this study only provides generalities and that local and regional expertise for seagrass restoration are important to achieve greater success (Van Katwijk, et al., 2016). Especially since natural variability among locations, the local biology and ecology of the restored species, and environmental conditions during the restoration process all have a strong influence on the success of rehabilitation projects, such that the success of a projects in a given area cannot be guaranteed (Ganassin & Gibbs, 2008). Furthermore, seagrass restoration techniques have still only been documented to successfully replace small areas of seagrasses. Thus, the restoration of large areas of seagrass is more uncertain. Also, optimal restoration techniques might differ per specie. However, seagrass restoration techniques have not been developed so far that methods can be recommended for different species in different habitats (Ibid.).

8.2 Management implications

For coastal managers, two useful 'products' resulted from this study. The first implies the maps indicating the locations of erosion hotspots (Figure 4.1 and Figure 4.2). Abating erosion should be prime focus in these areas. Although comparison in the severity of erosion between continents is only in part possible due to lack of data, it seems that muddy erosion problems are occurring most often in Asia and North and South America. Relatively speaking, erosion occurring along mud coasts in Africa and Australia is of little importance. Coastal manager should therefore focus on muddy erosion hotspots in Asia and America.

Furthermore, this study provides coastal managers with an overview of potential restoration techniques which have been implemented for mud recharge and seagrass restoration along the coastal type bays and estuaries and barrier coasts (Figure 7.1 and Figure 7.2). Together with the related success and failure factors of implementation (see Section 5.6 and Section 6.7.2 for an overview), these findings can aid coastal managers in choosing a technique based on economical, technical and environmental preferences. However, no techniques were applied along open coasts where most erosion occurs. Results suggest that implementation of mud recharge and seagrass seems to be most suitable in less energetic environments like barrier coasts, bays and estuaries. The same might be true for the NBS oysters and mangroves. For example, coastal protection offered by oyster reefs in sandy environments has been documented to be most effective in low energy environments (Beck, et al., 2014). The best locations for mangroves to grow include more sheltered environments like bays, lagoon, estuaries and shores behind barrier islands. Establishment of mangrove seedlings is difficult along open exposed coasts due to wave action. However, mangroves modify the local wave climate and can therefore grow out from a sheltered environment, progressively growing into medium-high energy environments (Saenger, 2002). When sheltered from wave action, mangroves seedlings might therefore establish along open coast. This suggesting the mangroves might be a suitable NBS for more energetic environments. Research is needed to confirm this.

Although seagrass and mud recharges schemes have been shown to provide erosion mitigation services along bays, estuaries and barrier mud coasts, these solutions are still in the experimental phase and data availability is limited. Consequently, success is not guaranteed and they seem to be most suitable in less energetic environments like barrier coasts, bays and estuaries. Especially implementing seagrass as an NBS is uncertain due to varying restoration successes and vulnerability to external impacts like climate change and anthropogenic threats. When seagrass is chosen as a NBS it is therefore important to combine coastal planning with watershed management. Mud recharge and seagrass meadows should therefore not be seen as a single solution (yet) to mitigating erosion, but more as part of an integrated mitigation plan. Funding for more research is needed to quantify the erosion mitigation potential of all NBSs and their individual technique under different hydrodynamic conditions. In this, especially for the living NBS, long term monitoring is essential. Particularly because monitoring is often only done for a comparatively short time frame (1-3 years), making evaluation of restoration successes difficult (Statton, et al., 2012). In terms of execution costs of mud recharge, sediment stirring might pertain lowest cost compared to direct placement and trickle charge. Thus, although sediment stirring has not been executed yet in muddy environments to mitigate erosion, experimental research related to this technique might be useful for coastal managers. Relative costs of seagrass restoration techniques are unknown.

Besides seagrass planting, it is also important to preserve existing seagrass beds. This is because while seagrass die-off tends to be rapid, natural recovery of disturbed seagrass habitats is comparatively slow. Furthermore, the success of seagrass transplantation and restoration is uncertain and the experiences among species vary enormously (Ondiviela, et al., 2014). Also, from an economic viewpoint, it is far more cost effective to preserve a seagrass habitat from damage than to restore an area after its degradation (Paling, et al., 2009).

Beside implementing mud recharge, seagrass, oyster or mangroves along a coastline as NBS, a different type of solution could also be applied: coastal realignment aka managed retreat. Managed retreat can be applied on eroding mud coasts which are backed by low value land. With this approach a buffer zone is created by setting back the defence works and breaching the existing wall. A temporary bund can be created behind which mud sediments can be placed to raise the backshore area where vegetation can grow (Burt, 1996).

This study only included on site erosion control solutions. However, formulating a complete advice would also imply considering the broader hydrological and morphological system in which the erosion hotspot is located. In this it is important to tackle the drivers of coastal erosion which might be located further up in the drainage basin.

8.3 Scientific implications

This study is an overview study. New insights were obtained by combining knowledge of grey and scientific sources with satellite data. Thus, this study gives the first global overview of locations of strong erosion along muddy tropical coasts. Also, the first state of the art overview of potential restoration techniques for mud recharge and seagrass beds has been created. Showing that knowledge related to these NBS in muddy environments is still in its infancy. Since mud recharge and seagrass restoration seem to be most suitable for less energetic environments, this study shows a knowledge gap related to potential NBS for erosion mitigation along open mud coasts.

8.3.1 Further research

Future research is recommended to address the knowledge gaps identified in this study. Further research should give insight in which hotspots should be of prime concern to coastal managers of the 112 found. To make comparison between hotspots possible the physical extent of erosion should be determined for each hotspot, combined with social and economic knowledge to indicate the level of societal importance of each coastal hotspot. In this, a distinction should be made between inundation and erosion. Furthermore, more research needs to be executed to indicate potential long term erosion along the South American coast between the Amazon and Orinoco river. To separate potential long term erosion from natural erosion phases along this coast, satellite data must be available for at least 60 years (total period of mudbank migration and coastal erosion along Guyana coast).

All documentation found in this study show implementation of mud recharge and seagrass restoration in bays, estuaries and along barrier coasts. Experimentation is thus needed for planting of seagrass and implementation of mud recharge along open coasts in high energy environments (Fonseca, et al., 1998). Furthermore, comparison with the NBS mangrove forests and oyster reefs might be useful in finding solutions for erosion problems. In this, different environmental settings should be considered and hydrodynamic conditions and changes should be quantitatively measured (wave height, tidal current speed, bed elevation etc.). This will give insight in what terms like 'low' and 'moderate' energy environments imply. Research should also include implementation costs of different restoration techniques and their lifespan. Further seagrass restoration projects must indicate if planting seagrass beds can aid in reducing coastline retreat.

Specifically, for direct mud placement, more experimental research is needed related to different potential types of retaining structures and their successfulness in retaining sediment particles. Furthermore, research should show if direct placement can also be applied successfully in subtidal muddy environments and if sediment stirring (agitation dredging and water injection) is a successful technique for mitigating mud erosion. Experimental research is also needed to confirm if combining brushwood fencing with trickle charge and/or sediment stirring will increase the recharge abilities of these restoration techniques.

For seagrass restoration, large scale transplantation in muddy environments is yet to be executed. Furthermore, research is needed to indicate if soil type is of influence to seagrass restoration. Also, the potential of seagrass to attenuate waves and reduce coastal retreat in muddy environments is yet to be proven. When looking at restoration techniques, the successfulness of restoration with seed material in muddy environments and sods in subtidal areas still needs to be proven. Also, more restoration projects need to be executed to find relative success of the different types of anchoring material for planting vegetative material, both on the long run as well as initial survival. It is also recommended to look at the best restoration techniques per specie, since this might differ.

9 Conclusion

This study looked for erosion hotspots along tropical mud coast and nature based solutions to mitigate this erosion. The first research question implies: *Where is erosion on mud coasts occurring strongest in tropical regions?*

- Globally 112 erosion hotspots were identified along tropical mud coasts, mostly in Asia and North and South America. More erosion hotspots are found along open coasts then bays, estuaries and barrier coasts.

The second research question was: *What restoration techniques of mud recharge and seagrass can be implemented to mitigate erosion along tropical mud coasts and how effective are they?*

- Mud recharge schemes of direct confined intertidal placement and trickle charge (intertidal trickle charge and water column recharge) have been observed to aid in erosion mitigation in estuaries and bays in the intertidal zone. Potentially direct unconfined subtidal placement and sediment stirring techniques (with or without permeable structures) might be applied too.
- Independent of restoration techniques, seagrass beds on muddy substrate have been observed to mitigate erosion. This might however be subjected to seasonal variation, even in tropical areas, and not applicable for every specie in every location and/or scenario. However, successful restoration is a precondition. Some level of transplantation success has been achieved in the muddy coastal type bay/estuary and barrier coast. It has been done by planting sods in the subtidal area or anchored rhizome fragments to staples or weights in the inter- or subtidal area. Survival rates of transplantation range between 0-100%. Possibly seeds might be implemented as planting material also. Consequently, the potential of introducing seagrass as an erosion mitigation solution is still uncertain.
- Due to the limited availability of data and lack in long term monitoring, the efficiency of erosion mitigation provided by mud recharge techniques and seagrass cannot be given. Findings should thus be taken with caution.

The third research question refers to: *What success and failure factors can be identified concerning the implementation and lifespan of the erosion mitigation solutions and their related restoration techniques, along tropical muddy coasts?*

- Reasons causing poor seagrass transplantation are numerous and seagrass is vulnerable to anthropogenic and climate change related impacts. Consequently, long term survival of seagrass is uncertain. However, seagrasses are often the dominant coastal primary producers and closely linked with high fisheries production due to the nursery habitat they provide. When energetic conditions are not restricting, choosing anchored rhizome fragments as restoration technique is advised. This is less labor intensive, costly and impacting to donor sites compared to sods.
- Implementation of mud recharge might negatively affect the surrounding area due to sediment losses outside the targeted area and potential smothering of benthic life. Consequently, mud recharge is not advised to be implemented in areas with sensitive marine life or in the proximity of commercial activities like shellfishery. When mud recharge is chosen, trickle charge is thus advised above direct placement due to its lower recharge rate. Data concerning the lifespan of recharge techniques is lacking.

The main insight in this study is that although strong erosion is occurring most along open mud coasts, mud recharge and seagrass have not been implemented along this coastal mud type. Due to a lack of experimental data it is unknown if these NBSs could also be applied along these more energetic coasts.

10 References

ABP Research, 1998. *Review of coastal habitat creation and recharge schemes. Report No. R.909,* Southampton, Hampshire, UK: ABP Research & Consultancy Ltd.

Albers, T., San, D. & Schmitt, K., 2013. *Shoreline management guidelines. Coastal protection in the Lower Mekong Delta,* s.l.: GIZ.

Almela, E., Diekman, O. & Bondo-Christensen, P., 2004. Can transplanting accelerate the recovery of seagrasses?. In: J. Borum, C. Duarte, D. Krause-Jensen & T. Greve, eds. *European seagrasses: an introduction to monitoring and management.* s.l.:The M&MS project, pp. 77-82.

Amos, C. L. et al., 2004. The stability of tidal flats in Venice Lagoon—the results of in-situ measurements using two benthic, annular flumes. *Journal of Marine Systems, 51(1),* pp. 211-241.

Anthony, E. J. et al., 2010. The Amazon-influenced muddy coast of South America: A review of mud-bank–shoreline interactions. *Earth-Science Reviews, 103(3),* pp. 99-121.

Atkinson, P., Crooks, S., Grant, A. & Rehfisch, M., 2001. *The success of creation and restoration schemes in producing intertidal habitat suitable for water birds. No. 425, part 1,* Norfolk, UK: English Nature Research Reports.

Barbier, E. B. et al., 2011. The value of estuarine and coastal ecosystem services. *Ecological monographs, 81(2),* pp. 69-193.

Bayraktarov, E. et al., 2016. The cost and feasibility of marine coastal restoration. *Ecological Applications, 26(4),* pp. 1055-1074.

Beck, M. et al., 2014. The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. *Ocean Coastal Management, 90,* pp. 50-57.

Bolam, S. G. & Whomersley, P., 2005. Development of macrofaunal communities on dredged material used for mudflat enhancement: a comparison of three beneficial use schemes after one year. *Marine Pollution Bulletin, 50(1),* pp. 40-47.

Borsje, B. et al., 2011. How ecological engineering can serve in coastal protection. *Ecological Engineering, 37(2),* pp. 113-122.

Boyer, K. E. & Wyllie-Echeverria, S., 2010. *Eelgrass Conservation and Restoration in San Francisco Bay: Opportunities and Constraints. Final report for the San Fransico Bay Subtidal Habitat Goals Project.,* Tiburon, CA, USA: San Fransico State University.

Bray, R. N., 2008. *Environmental aspects of dredging.* Leiden, the Netherlands: Taylor & Francis/Balkema.

Bridges, T. et al., 2015. *Use of Natural and Nature-Based Features (NNBF) for Coastal Resilience. North Atlantic Coast Comprehensive Study: Resilient Adaptation to Increasing Risk.* Vicksburb, USA: US Army Corps of Engineers: Engineer Research and Development Center.

Bulseco-McKim, A., 2012. *Eelgrass restoration. Phase 2: Restoring Eelgrass to the Neponset River Estuary,* s.l.: s.n.

Burt, T., 1996. *Guidelines for the beneficial use of dreged material. Report SR 488 ,* Oxon, UK: HR Wallingford.

Butler, R., MacVicar, R. & Foster, R., 2011. *Attempts to Restore Eelgrass (Zostera marina) in Port Moody Inlet, British Columbia. Pacific WildLife Foundation Technical Report Number 1,* Barnet Highway, British Columbia, USA: Pacific Wildlife Foundation.

Byron, D., Heck, K. L. J., Powers, S. P. & Scyphers, S. B., 2011. Oyster Reefs as Natural Breakwaters Mitigate Shoreline Loss and Facilitate Fisheries. *PLoS ONE, 6(8),* p. e22396.

Cai, F. et al., 2009. Coastal erosion in China under the condition of global climate change and measures for its prevention. *Progress in Natural Science, 19(4),* pp. 415-426.

Calumpong, H. P. & Fonseca, M. S., 2001. Seagrass transplantation and other seagrass. *Global seagrass research methods, 33,* p. 425.

Carlson Jr, P. R., Yarbro, L. A. & Barber, T. R., 1994. Relationship of sediment sulfide to mortality of Thalassia testudinum in Florida Bay. *Bulletin of Marine Science, 54(3),* pp. 733-746.

Carminati, E. & Martinelli, G., 2002. Subsidence rates in the Po Plain, northern Italy: the relative impact of natural and anthropogenic causation. *Engineering Geology, 66(3),* pp. 241-255.

Chinnarasri, C. & Kittirart, S., 2012. *Mud coastal protection using triangular modules of breakwater made of bamboo.* Rhodes, Greece, International Society of Offshore and Polar Engineers, pp. ISOPE-I-12-507.

Christianen, M. J. et al., 2013. Low-canopy seagrass beds still provide important coastal protection services. *PloS one, 8(5),* p. e62413.

Church, J. A. & White, N. J., 2011. Sea-level rise from the late 19th to the early 21st century. *Surveys in Geophysics, 32(4-5),* pp. 585-602.

Chu, Z., Sun, X., Zhai, S. & Xu, K., 2006. Changing pattern of accretion/erosion of the modern Yellow River (Huanghe) subaerial delta, China: Based on remote sensing images. *Marine Geology, 227(1),* pp. 13-30.

Clark, J., 1995. *Coastal Zone Management Handbook.* s.l.:CRC Press.

Colenutt, A. J., 1999a. Intertidal recharge techniques [Online]. In: *Beneficial Use of Dredged Material for Inter-tidal Recharge: Management options for the Lymington Saltmarshes.* Lymington, ZG, USA: NFDC, pp. Cited on 14-02-2017: http://nfdc.gov.uk/CHttpHandler.ashx?id=2608&p=0.

Colenutt, A. J., 1999b. Environmental impacts of intertidal recharge [Online]. In: *Beneficial Use of Dredged Material for Inter-tidal Recharge: Management options for the Lymington Saltmarshes.* Lymington, ZG, USA: NFDC, pp. Cited on 20-02-2017: http://nfdc.gov.uk/.

Colenutt, A. J., 1999c. Evaluation of inter-tidal recharge techniques: a discussion [Online]. In: *Beneficial Use of Dredged Material for Inter-tidal Recharge: Management options for the Lymington Saltmarshes.* Lymington, ZG, USA: NFDC, pp. Cited on 20-02-2017: http://nfdc.gov.uk/CHttpHandler.ashx?id=2610&p=0.

Colenutt, A. J., 2001. *Saltmarsh management techniques, a review,* s.l.: New forest district council coast protection group.

Costa-Pierce, B. & Weinstein, M., 2002. Use of dredge materials for coastal restoration. *Ecological Engineering, 19(3),* pp. DOI: 10.1016/S0925-8574(02)00076-9.

Daidu, F., Yuan, W. & Min, L., 2013. Classifications, sedimentary features and facies associations of tidal flats. *Journal of Palaeogeography, 2(1),* pp. 66-80.

Dankers, P., 2002. *The behaviour of fines released due to dredging. A literature review,* Delft, the Netherlands: Faculty of Civil engineering and Geoscience, Delft University of Technnology.

Dawes, C., n.d. *The dynamic seagrasses of the Gulf of Mexico and Florida coast,* Tampa, Florida, USA: University of South Florida.

De Boer, W. F., 2007. Seagrass–sediment interactions, positive feedbacks and critical thresholds for occurrence: a review. *Hydrobiologia, 591(1),* pp. 5-24.

De Fouw, J., 2016. *Bottom-up and top down forces in a tropical intertidal ecosystem. The interplay between seagrasses, bivalves and birds. PhD Thesis,* Groningen, the Netherlands: University of Groningen.

De Vriend, H. & Van Koningsveld, M., 2012. *Building with Nature: Thinking, acting and interacting differently,* Dordrecht, the Netherlands: EcoShape.

De Vriend, H., Van Koningsveld, M. & Aarninkhof, S., 2014. Building with nature: the new Dutch approach to coastal and river works. *Civil Engineers, 167(CE1),* pp. 18-24.

Deltares, 2016. *Mangrove restoration: to plant or not to plant?,* s.l.: Ecoshape Consortium.

Deltares, n.d.1. *Building with Nature [Online],* Cited on 20-12-2016: https://publicwiki.deltares.nl/display/BWN1/Building+with+Nature.

Deltares, n.d.2. *Five basic steps for generating Building with Nature designs [Online],* Cited on 17-01-2017: https://publicwiki.deltares.nl/display/BWN1/Guideline.

Díaz, S., Fargione, J., Chapin III, F. S. & Tilman, D., 2006. Biodiversity loss threatens human well-being. *PLoS Biol, 4(8),* p. e277.

Dineen, J., 2001. *Indian River Lagoon Species Inventory [Online],* Cited on 15-03-2017: http://www.sms.si.edu/irlspec/Halodu_beaudet.htm.

Donchyts, G. et al., 2016. Earth's surface water change over the past 30 years. *Nature Climate Change, 6(9),* pp. 810-813.

Dugan, J. et al., 2011. Estuarine and Coastal Structures: Environmental Effects, A Focus on Shore and Nearshore Structures. In: *Treatise on Estuarine and Coastal Science.* Waltham, USA: Acadamic Press, pp. 17-41.

Dunton, K. H., 1990. Production ecology of Ruppia maritima L. sl and Halodule wrightii Aschers, in two subtropical estuaries. *Journal of Experimental Marine Biology and Ecology, 143(3),* pp. 147-164.

Dunton, K. H., 1994. Seasonal growth and biomass of the subtropical seagrassHalodule wrightii in relation to continuous measurements of underwater irradiance. *Marine Biology, 120(3),* pp. 479-489.

EA, n.d. *Science Report – R&D Technical Report (PFA – 076/TR) [Online],* Cited on 16-02-2017, on: http://evidence.environment-agency.gov.uk/FCERM/Libraries/FCERM_Project_Documents/Appendix_D.sflb.ashx.

Eekelen, E. v. et al., 2016. *Muddy waters and the Wadden Sea Harbours,* Miami, Florida, USA: Proceedings of the Twenty-First World Dredging Congress, WODCON XXI, June 13-17.

Eisenhardt, K. M., 1989. Building Theories from Case Study Research. *Acadamy of Management Review, 14(4),* pp. 532-550.

Eisma, D., 1998. *Intertidal deposits: river mouths, tidal flats, and coastal lagoons (Vol. 16).* Boca Raton, USA: CRC Press LLC.

European Commission, 2004. *Living with coastal erosion in Europe. Sediment and space for sustainability. Results from the erosion study,* Schiedam, the Netherlands: Van Rossum Rijngroep B.V..

FAO, The World Bank & African Development Bank, 2016. *African Package for Climate-Resilient Ocean Economies,* s.l.: FAO.

Flemming, B. W., 2002. Chapter Six Geographic distribution of muddy coasts. *Proceedings in Marine Science, 4,* pp. 99-201.

Fletcher, C., 2008. *Large scale use of muddy dredged materials for sustainable flood defences and habitat management-AE0260. Summary report. Report No. SR685,* s.l.: HR Wallingford limited.

Fletcher, C. A. et al., 2000. Habitat creation and the beneficial use of muddy dredged material in the United Kingdom. In: J. Pederson & E. Adams, eds. *Dredged material management options and environmental considerations, proceedings of a conference.* Cambridge, MA, USA: Massachusetts Institute of Technology, pp. 94-100.

Fonseca, M. S., Kenworthy, W. J. & Thayer, G. W., 1982. *A Low-Cost Planting Technique for Eelgrass (Zostera marina L.),* USA: NATIONAL MARINE FISHERIES SERVICE BEAUFORT .

Fonseca, M. S., Kenworthy, W. J. & Thayer, G. W., 1998. *Guidelines for the conservation and restoration of seagrasses in the United States and adjacent waters,* Silver Spring, USA: NAOO.

Fonseca, M. S., Thayer, G. W. & Kenworthy, W. J., 1987. The use of ecological data in the implementation and management of seagrass restorations. *Fla. Mar. Res. Publ, 42,* pp. 175-187.

Foster, N. M., Hudson, M. D., Bray, S. & Nicholls, R. J., 2013. Intertidal mudflat and saltmarsh conservation and sustainable use in the UK: A review. *Journal of environmental management, 126*, pp. 96-104.

French, J. R. & Burningham, H., 2009. Restoration of an Eroded Estuarine Foreshore Using Cohesive Dredge Material, Orwell Estuary, UK. *Journal of Coastal Research,* pp. 1444-1448.

French, P., 2011. *Coastal Defences: Processes, Problems and Solutions,* New York, USA: Routledge Inc.

Ganassin, C. & Gibbs, P. J., 2008. *A review of seagrass planting as a means of habitat compensation following loss of seagrass meadow. Fisheries final report Series No. 96,* Cronulla, Australia: NSW Department of Primary Industries.

Gardel, A. G. N., 2005. A Satellite Image–Based Method for Estimating Rates of Mud Bank Migration, French Guiana, South America. *Journal of Coastal Research, 39,* pp. 720-728.

Garvis, S. K., 2012. *Quantifying the impacts of oyster reef restoration on oyster coverage, wave dissipation and seagrass recruitment in mosquito Lagoon, Florida ,* Orlando, Florida, USA: University of Central Florida Orlando.

Ghazali, N., 2006. Coastal erosion and reclamation in Malaysia. *Aquatic Ecosystem Health & Management, 9(2),* pp. 237-247.

Gratiot, N. et al., 2008. Significant contribution of the 18.6 year tidal cycle to regional coastal changes. *Nature Geoscience, 1(3),* pp. 169-172.

Grech, A. et al., 2012. A comparison of threats, vulnerabilities and management approaches in global seagrass bioregions. *Environmental Research Letters, 7(2),* p. 024006.

Green, E. & Short, F., 2003. *World Atlas of Seagrasses.* Berkeley, USA: University of California Press.

Halley, R. B., 2000. *Florida Bay Watch Report. A Synthesis document of the Florida Bay Adjacent Marine Systems Science Program [Online],* Cited on 23-03-2017: http://www.floridabay.org/pub/bw_report/murkypast.shtml.

Halley, R. B., Holmes, C. & Prager, E. J., 1997. Florida Bay mudbanks: Relatively new piles of mostly old sediment (abstract). *US Geological Survey program on the South Florida ecosystem-Proceedings of the technical symposium in Ft. Laudedale, Florida*.

Hamer, F., 2007. *ComCoast. Projects Abbotts Hall Farm and Horsey Island (UK),* Delft, the Netherlands: Rijkswaterstaat DWW.

Harvey, M., Gauthier, D. & Munro, J., 1998. Temporal changes in the composition and abundance of the macro-benthic invertebrate communities at dredged material disposal sites in the Anse à Beaufils, Baie des Chaleurs, Eastern Canada. *Marine Pollution Bulletin, 36,* pp. 41-55.

Hasegawa, N., Hori, M. & Mukai, H., 2008. Seasonal changes in eelgrass functions: current velocity reduction, prevention of sediment resuspension, and control of sediment–water column nutrient flux in relation to eelgrass dynamics. *Hydrobiologia, 596(1),* pp. 387, 399.

Healy, T., Wang, Y. & Healy, J. A., 2002. *Muddy coasts of the world: processes, deposits and function (Vol. 4).* Amsterdam, the Netherlands: Gulf Professional Publishing.

Hsu, T., Lin, T. & Tseng, I., 2007. Human impact on coastal erosion in Taiwan. *Journal of Coastal Research, 23(4),* pp. 961-973.

IADC, 2013. Water injection dredging [Online]. *Facts about, 1*, Cited on 14-02-2017: https://www.iadc-dredging.com/ul/cms/fck-uploaded/documents/PDF%20Facts%20About/facts-about-water-injection-dredging.pdf.

IUCN, 2016. *The IUCN Red list of Threatened Species [Online],* Cited on 01-03-2017: www.iucnredlist.org.

Jokiel, D. & Detering, M., n.d. *An Innovative Sediment Removal Solution - Application and Project Experiences,* Cologne, Germany: Cologne University of Applied Science.

Jones, C., Lawton, J. & Shachak, M., 1994. Organisms as ecosystem engineers. *Oikos, 69,* pp. 373-386.

Julianus, E., 2016. *The development of a fine sediment nourishment strategy for the rehabilitation of an eroding mangrove-mud coast,* Delft, the Netherlands: Civil Engineering and Geoscience faculty of TU Delft.

Kamali, B. & Hashim, R., 2010. Bamboo foundation mat for rubble mound breakwaters on mud deposits. *International Journal of Physical Sciences, 5(9),* pp. 1406-1410.

Kelletat, D. H., Scheffers, A. M. & May, S. M., 2014. Coastal environments from polar regions to the tropics: a geographer’s zonality perspective. *Geological Society, London, Special Publications, 388(1),* pp. 33-57..

Ke, X. & Collens, M., 2002. Saltmasrhes in the West Solent (southern England); their morphodynamics and evolution. In: *Muddy coasts of the world: processes, deposits and function (Vol. 4).* Amsterdam, the Netherlands: Gulf Professional Publishing, pp. 411-437.

Kirby, R., 1995. *Tidal flat regeneration - a beneficial use of muddy dredged material. 14 'the World dredging congress 'Dredging Benefits'. 14-17 November.* Amsterdam, the Netherlands, WODA.

Kirby, R., 2013. Managing industrialised coastal fine sediment systems. *Ocean & coastal management, 79,* pp. 2-9.

Kjerfve, B. et al., 2002. Morphodynamics of muddy environments along the Atlantic coast of North and South America. In: *Muddy coast of the World processes, Deposits and Functions.* s.l.:Elsevier Science B.V., pp. 479-532.

Kumar, V. S. & George, J., 2016. Influence of Indian summer monsoon variability on the surface waves in the coastal regions of eastern Arabian Sea. *Annales Geophysicae, 34(10),* pp. 871-885.

LievenseCSO, 2017. *Integrated coastal zone management, Suriname [Online],* Cited on 25-07-2017: http://www.lievensecso.com/Home/Projects/Integrated-coastal-zone-management%2C-Suriname.

Lipcius, R. N. et al., 2015. Overcoming restoration paradigms: Value of the historical record and metapopulation dynamics in native oyster restoration. *Frontiers in Marine Science, 2(65),* p. doi: 10.3389/fmars.2015.00065.

Li, X. et al., 2015. GIS-based methodology for erosion risk assessment of the muddy coast in the Yangtze Delta. *Ocean & Coastal Management, 108,* pp. 97-108.

MAFF, 1993. *Strategy for Flood and Coastal Defence in England and Wales,* London, UK: MAFF.

Marion, S. R. & Orth, R. J., 2010. Innovative Techniques for Large-scale Seagrass Restoration Using Zostera marina (eelgrass) Seeds. *Restoration Ecology, 18(4),* pp. 514-526.

McGranahan, G., Balk, D. & Anderson, B., 2007. The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. *Environment and urbanization, 19(1),* pp. 17-37.

Mehta, A. J., 2002. Mudshore dynamics and control. In: *Muddy Coasts of the World: Processes, Deposits.* Florida, USA: Elsevier Science B.V., pp. 19-60.

Mehta, A. J. & Jiang, F., 1993. *Some observations on water wave attenuation over nearshore underwater mudbanks and mud berms,* Gainesville, Florida, USA: University of Florida.

Mehta, A. J. & Jiang, F., 1994. *Water Wave Attenuation Over Nearshore Underwater Mudbanks and Mud Berms. Dredging Research Technical Notes. DRP 1-14,* Vicksburg, USA: US Army Engineer Waterways Experiment Station.

Mellors, J., Marsh, H., Carruthers, T. J. & Waycott, M., 2002. Testing the sediment-trapping paradigm of seagrass: Do seagrasses influence nutrient status and sediment structure in tropical intertidal environments?. *Bulletin of Marine Science, 71(3),* pp. 1215-1226.

Mundy, E. & Kelly, R., 2010. *Mitigation and monitoring for the Stour and Orwell Estuaries SPA and Hamford Water SPA. Annual Review 2009. Final report 9T2703,* Peterborough, UK: Harwich Haven Authority.

Murali, R., Boora, P. & Vethamony, P., 2013. *Coastal sensitivity mapping of Gulf of Kachchh and Gulf of Cambay, India. Proceedings of the 7th Conference on Asian and Pacific Coasts (APAC 2013). September 24-26.* Bali, Indonesia, s.n.

NAOO, 2011. *A Concise Environmental Assessement (EA) for emergency restoration of seagrass impacts from the deepwater horizon oil spill response,* s.l.: NAOO.

National Research Council, 2007. *Mitigating shore erosion along sheltered coasts.* Washington, DC, USA: National Acadamy of Sciences.

Nicholls, R. et al., 2007. Coastal systems and low-lying areas. In: M. Parry, et al. eds. *Climate change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the fourth assessment.* Cambridge, UK: Cambridge University Press, pp. 315-356.

OCEANA, n.d. *Restoration of seagrass meadows. Sustainable development manuals.,* s.l.: OCEANA.

Ondiviela, B. et al., 2014. The role of seagrasses in coastal protection in a changing climate. *Coastal Engineering, 87,* pp. 158-168.

Paling, E., Fonseca, M., Van Katwijk, M. & Van Keulen, M., 2009. Seagrass restoration. In: G. Perillo, E. Wolanski, D. Cahoon & M. Brinson, eds. *Coastal wetlands: An integrated ecosystem approach.* s.l.:Elsevier B.V., pp. 687-713.

Paling, E. & van Keulen, M., 2002. *Seagrass restoration in Australia. In: Proceedings of the seagrass restoration workshop for Gulf St Vincent, 15 - 16 May, 2001.* Adelaide, Australia, Murdoch University.

Park, J. I. & Lee, K. S., 2007. Site-specific success of three transplanting methods and the effect of planting time on the establishment of Zostera marina transplants. *Marine Pollution Bulletin, 54(8),* pp. 1238-1248.

Paul, B. & Rashid, H., 2017. *Climatic Hazards in Coastal Bangladesh, non-structural and structural solutions.* Oxford, UK: Elsevier Inc.

Perrow, M. R. & Davy, A. J., 2002. *Handbook of ecological restoration (Vol. 2),* s.l.: Cambridge University Press.

Pilarczyk, K. W., 2005. Coastal stabilization and alternative solutions in international perspective. *Arabian Coast,* pp. 1-26.

Pinet, P. R., 2011. *Invitation to oceanography.* 5th ed. s.l.:Jones & Bartlett Publishers.

Pontee, N., Narayan, S., Beck, M. W. & Hosking, A. H., 2016. Nature-based solutions: lessons from around the world. *Maritime Engineering, 169(1),* pp. 29-36.

Prager, E. J. & Halley, R. B., 1999. The influence of seagrass on shell layers and Florida Bay mudbanks. *Journal of Coastal Research, 15(4),* pp. 1151-1162.

Ranwell, D. S. et al., 1974. Zostera transplants in Norfolk and Suffolk, Great Britain. *Aquaculture, 4,* pp. 185-198.

Ray, G. L., 2000. Infaunal assemblages on constructed intertidal mudflats at Jonesport, Maine (USA). *Marine Pollution Bulletin, 40(12),* pp. 1186-1200.

Saenger, P., 2002. *Mangrove ecology, silviculture and conservation.* Dordrecht, the Netherlands: Kluwer acadamic publishers.

Saengsupavanich, C., 2013. Erosion protection options of a muddy coastline in Thailand: stakeholders' shared responsibilities. *Ocean & coastal management, 83,* pp. 81-90..

Schratzberger, M., Bolam, S., Whomersley, P. & Warr, K., 2006. Differential response of nematode colonist communities to the intertidal placement of dredged material. *Journal of Experimental Marine Biology and Ecology, 334(2),* pp. 244-255.

Seddon, S., 2004. Going with the flow: Facilitating seagrass rehabilitation. *Ecological Management & Restoration, 5(3),* pp. 167-176.

Shafer, D., Sherman, T. & Wyllie-Echeverria, S., 2007. Do desiccation tolerances control the vertical distribution of intertidal seagrasses? *Aquatic Botany, 87(2),* pp. 161-166.

Shepard, C., Crain, C. & Beck, M., 2011. The Protective Role of Coastal Marshes: A Systematic Review and Meta-analysis. *PLoS ONE, 6(11),* p. e27374.

Short, F., Carruthers, T., Dennison, W. & Waycott, M., 2007. Global seagrass distribution and diversity: a bioregional model. *Journal of Experimental Marine Biology and Ecology, 350(1),* pp. 3-20.

Short, F. & Coles, R., 2001. *Global seagrass research methods (Vol. 33).* Amsterdam, the Netherlands: Elsevier.

Silverster, R. & John, R., 1997. *Coastal Stabilization.* River Edge, USA: World Scientific Publishing Co. Pte. Ltd.

Statton, J., Dixon, K. W., Hovey, R. K. & Kendrick, G. A., 2012. A comparative assessment of approaches and outcomes for seagrass revegetation in Shark Bay and Florida Bay. *Marine and Freshwater Research, 63(11),* pp. 984-993.

Steel, R. J. & Milliken, K. L., 2013. Major advances in siliciclastic sedimentary geology, 1960–2012. *Geological Society of America Special Papers, 500,* pp. 121-167.

Sullivan, N., 2000. The Use of Agitation Dredging, Water Injection Dredging and Sidecasting: Results of a Survey of Ports in England and Wales. *Terra et Aqua, 78*, pp. 11-20.

Suykerbuyk, W. et al., 2016. Unpredictability in seagrass restoration: analysing the role of positive feedback and environmental stress on Zostera noltii transplants. *Journal of Applied Ecology, 53,* pp. 774-784.

Syvitski, J., Vörösmarty, C., Kettner, A. & Green, P., 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science, 308(5720),* pp. 376-380.

Temmerman, S. et al., 2013. Ecosystem-based coastal defence in the face of global change. *Nature, 504(7478),* p. 79–83.

Thangaradjou, T. & Kannan, L., 2008. Survival and growth of transplants of laboratory raised axenic seedlings of Enhalus acoroides (Lf) Royle and field-collected plants of Syringodium isoetifolium (Aschers.) Dandy, Thalassia hemprichii (Ehrenb.). *Journal of Coastal Conservation, 12(3),* p. 135.

Tonneijck, F. et al., 2015. *Building with Nature Indonesia Securing Eroding Delta Coastlines. R1.5 Design and Engineering Plan. Report version V7.0,* s.l.: Ecoshape.

Turner, S. & Schwarz, A. M., 2006. Management and conservation of seagrass in New Zealand: an introduction. *Science for Conservation, 264,* pp. 1-90.

UK CHM, 1999. *Soft engineering techniques for high and low energy coasts,* s.l.: s.n.

UKMPA Centre, 2001. *Beneficial use case studies [Online],* Cited on 14-02-2017: http://www.ukmarinesac.org.uk/activities/ports/ph19.htm#a3.

Unsworth, R. K., Van Keulen, M. & Coles, R. G., 2014. Seagrass meadows in a globally changing environment.. *Marine Pollution Bulletin, 83(2),* pp. 383-386.

Valle, M. et al., 2015. Increasing the chance of a successful restoration of Zostera noltii meadows. *Aquatic Botany, 127,* pp. 12-19.

Van Ginneken, J., 2017a. *Cover drawing eroding tropical mud coasts.* [Art].

Van Ginneken, J., 2017b. *Conceptualized influence of direct placement techniques of morphological and hydrological proccess along coasts.* [Art].

Van Ginneken, J., 2017c. *Conceptualized influence of trickle charge techniques of morphological and hydrological proccess along coasts.* [Art].

Van Ginneken, J., 2017d. *Conceptualized influence of 2 sediment stirring techniques on hydrological and morphological processes along coasts.* [Art].

Van Ginneken, J., 2017e. *Planting material seagrass rehabilitation.* [Art].

Van Ginneken, J., 2017f. *Examples of anchoring devices for seagrass transplantation.* [Art].

Van Ginneken, J., 2017g. *Conceptualized contribution of seagrass to erosion mitigation.* [Art].

Van Katwijk, M. M. et al., 2009. Guidelines for seagrass restoration: importance of habitat selection and donor population, spreading of risks, and ecosystem engineering effects. *Marine pollution bulletin, 58(2),* pp. 179-188.

Van Katwijk, M. M. et al., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. *Journal of Applied Ecology, 53(2),* pp. 567-578.

Van Wesenbeeck, B. K. et al., 2015. Aquaculture induced erosion of tropical coastlines throws coastal communities back into poverty. *Ocean & Coastal Management, 116,* pp. 466-469.

Verhagen, H., 2000. *Facts about water injection dredging,* Delft, the Netherlands: IADC.

Vichkovitten, T., Intarachart, A., Khaodon, K. & Rermdumri, S., 2016. Transplantation of Tropical Seagrass Enhalus acoroides (L.) in Thai Coastal water: implication for habitat restoration. *GMSARN International Journal 10,* pp. 113-120.

Walker, H., 2012. *Artificial structures and shorelines (Vol. 10).* Dordrecht, the Netherlands: Kluwer academic publishers.

Wang, Y. et al., 2002. Definition, properties and classification of muddy coasts.. In: *Muddy Coasts of the World: Processes, Deposits*. s.l.:s.n.

Wetlands International, 2017a. *Mudflat along mangrove coast.* [Art].

Wetlands International, 2017b. *Seagrass along the coast of Nusa Dua in Bali.* [Art].

Widdows, J. et al., 2008. Effects of seagrass beds (Zostera noltii and Z. marina) on near-bed hydrodynamics and sediment resuspension. *Marine Ecology Progress Series, 358,* pp. 125-136.

Wightlink Ltd, 2014. *Lymington to Yarmouth Ferries: Third Annual Mitigation Review. Progress report 3 and 8th report for the Environmental Management Panel. Report No. R.2313,* Southampton, Hampshire, UK: ABP Marine Evironmental Research Ltd.

Wightlink Ltd, 2015. *Lymington to Yarmouth Ferries; Mitigation and Monitoring. 9th Report for the Environment Management Panel. Report R.2512,* Southampton, Hampshire, UK: ABP Marine Environmental Research Ltd.

Winterwerp, H. et al., 2014. *A sustainable solution for massive coastal erosion in Central Java. Towards Regional Scale Application of Hybrid Engineering. Discussion paper,* s.l.: Deltares & Wetlands International.

Winterwerp, J. C., Borst, W. G. & De Vries, M. B., 2005. Pilot study on the erosion and rehabilitation of a mangrove mud coast. *Journal of Coastal Research, 21(2),* pp. 223-230.

Winterwerp, J. C., Erftemeijer, P., Suryadiputra, N. & Zhang, L., 2013. Defining Eco-Morphodynamic Requirements for Rehabilitating Eroding Mangrove-Mud Coasts. *Society of Wetlands Scientists,* pp. 515-526.

Wong, P. et al., 2014. Coastal systems and low-lying areas. In: *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, UK and New York, USA: Cambridge University Press, pp. 361-409.

Xue, Y. Q. et al., 2005. Land subsidence in China. *Environmental geology, 48(6),* pp. 713-720.

Yozzo, D. J., Wilber, P. & Will, R. J., 2004. Beneficial use of dredged material for habitat creation, enhancement, and restoration in New York–New Jersey Harbor. *Journal of Environmental Management, 73(1),* pp. 39-52.

Zanuttigh, B., 2011. Coastal flood protection: What perspective in a changing climate? The THESEUS approach. *Environmental Science & Policy, 14(7),* pp. 845-863.

Zhang, Q. et al., 2015. Effect of silt and clay percentage in sediment on the survival and growth of eelgrass Zostera marina: transplantation experiment in Swan Lake on the eastern coast of Shandong Peninsula, China. *Aquatic Botany, 122,* pp. 15-19.

11 Appendix

11.1 Search terms

Table 11.1 indicates search terms entered in different online search engines to find grey and scientific literature to answer research question 2 and 3.

Tropical mud coast	Projects/organizations
• (Sub)tropical coast/zone • Mud • Silt • Clay • Silt – clay percentage > 70% (when mentioned) • Cohesive sediment	• GIZ • The nature conservancy • Ecoshape
Category NBS	Mitigating erosion

• Seagrass
    ○ Restore / rehabilitate / transplant
    ○ Silt / clay / mud / cohesive
• Nourishment/recharge
    ○ Mud/silt/clay
    ○ Fine/cohesive sediment
    ○ Dredged material
• General terms
    ○ Nature based solution/defence
    ○ Natural/green infrastructure
    ○ Building with (living) nature
    ○ Living shorelines
    ○ Engineering with nature
    ○ Green infrastructure
	• Abating/mitigating erosion
	• Coastal resilience
	• Abating coastal risks
	• Shoreline/coastline stabilization/protection
	• Wave attenuation / reduction
	• Current velocity
	• Soil / bed elevation
	• Biostabilization
	• Sediment trapping/stabilization/ accumulation/ accretion/ deposition/ stability
	• Accretion / elevation rate

Table 11.1: Search terms to find indicative studies to answer the research questions 2 and 3.

11.2 Identifying hotspots: literature sources

Table 11.2 indicates literature sources that have been used to indicate whether the erosion hotspots found in the Aquamonitor are located along mud coasts. All sources provide qualitative information.

Coastal location	Source
Worldwide	Flemming (2002)
Australia	Eismee (1998)
India, Gulf of Camba	Murali et al. (2013)
Indonesia	Walker et al. (2012)
Malaysia	Ghazali (2006)

Table 11.2: Qualitative literature sources consulted to identify whether coasts are muddy or not.

11.3 Mud recharge: implemented projects

Table 11.3 gives on overview of all the implemented mud recharge projects found in literature. Of each project information concerning the environmental conditions and the erosion mitigation services observed are provided. Except one project on wave attenuation, all information is qualitative.

Sediment recharge schemes	Implemented project	Environmental conditions		Erosion mitigation services			Reference implemented project
		Across shore	**Coast-al type**	**Bed ele-vation**	**Wave ate-nuation**	**Reduced coastline retreat**	
# Direct placement							
Confined intertidal	Horsea Island in the Walton Backwaters (UK)	Inter-tidal	BE				(Fletcher, et al., 2000)
	Orwell estuary (UK)	Inter-tidal	BE				(Schratzberger, et al., 2006)
	Orwell estuary (UK)	Inter-tidal	BE	1,1 m	YES*		(French & Burningham, 2009)
	Horsey Island (UK)	Inter-tidal	BE	YES			(Hamer, 2007) (Fletcher, et al., 2000)
	Lymington Estuary (UK)	Inter-tidal	BE	0,03-0,3 m**		YES	(Wightlink Ltd, 2015)
	Parkstone Yacht club (UK)	Inter-tidal	BE	YES			(ABP Research, 1998)
Unconfined subtidal	Mobile Bay	Sub-tidal	BE		29% and 46%		(Bray, 2008) (Mehta & Jiang, 1993)
Unconfined intertidal	Horsey Island (UK)	Inter-tidal	BE	NO			(EA, n.d)
# Trickle charge							
Intertidal placement	Medway Port (UK)	Inter-tidal	BE	YES			(UKMPA Centre, 2001) (ABP Research, 1998)
Subtidal placement	Stour and Orwell estuary (UK)	Sub-tidal	BE	YES, but inefficient			(Mundy & Kelly, 2010)
Water column recharge	Stour and Orwell estuary (UK)	Sub-tidal + Inter-tidal?	BE	YES			(Mundy & Kelly, 2010)

# Sediment stirring						
Agitation dredging						
Water injection dredging						

*Table 11.3: Implemented mud recharge projects found in literature, categorized based on restoration technique. When available data is given concerning the environmental conditions in which the projects were located and the observed erosion mitigation services provided. BE=coastal type bay and estuary. *=refers to reduced wave-erosion along seawall. **=elevation variation after 3 years depending om location on the mudflat after.*

11.4 Seagrass

11.4.1 Tropical seagrass species

Table 11.4 givers an overview of seagrass species occurring in tropical areas. In this a division is made between two bioregions based on the bordering global oceans: the tropical Atlantic bioregion and the tropical Indo-Pacific bioregion (Figure 2.1).

Bioregion	Description	Species
Tropical Atlantic (including the Caribbean Sea, Gulf of Mexico, Bermuda, the Bahamas, and both tropical coasts of the Atlantic)	High diversity tropical seagrasses (10 species) growing on back reefs and shallow banks in clear water	Halodule beaudettei, *H. wrightii* (H. bermudensis, H. emarginata), Halophila baillonii, Halophila decipiens, Halophila engelmanni, Halophila johnsonii, R. maritima, *Syringodium filiforme, Thalassia testudinum*, Halophila stipulacea+
Tropical Indo-Pacific (East Africa, south Asia and tropical Australia to the eastern Pacific)	Largest and highest diversity bioregion; tropical seagrasses (24 species) predominantly on reef flats but also in deep waters.	Cymodocea angustata, Cymodocea rotundata, Cymodocea serrulata, Enhalus acoroides, Halodule pinifolia, *Halodule uninervis*, H. wrightii, Halophila beccarii, Halophila capricorni, H. decipiens, Halophila hawaiiana, Halophila minor, H. ovalis, Halophila ovata, Halophila spinulosa, H. stipulacea, Halophila tricostata, R. maritima, *Syringodium isoetifolium*, *Thalassia hemprichii*, Thalassodendron ciliatum, Zostera capensis+, Z. japonica+, Zostera muelleri+ [Zostera capricorni]

Table 11.4: Seagrass species in tropical areas, divided into two geographic bioregions related to different worlds oceans. The skewed species indicate the most common species of the bioregions (Short, et al., 2007). Species per country can be found in the World Atlas of Seagrasses according to Green & Shorts (2003).

11.4.2 Coastal type occurence of dominant tropical species

Table 11.5 indicates along what coastal types the four most common tropical seagrass species (according to Short et al. (2007)), which can grow on muddy beds, have been observed to grow.

Specie	Coastal type			Reference
	Open coast	Bay/ Estuary	Barrier	
H. wrightii	x			(Dawes, n.d.)
		x		(Dunton, 1994) (Dunton, 1990)
			x	(Dineen, 2001)
Syringodium filiforme	x			(Dawes, n.d.)
		x		(Dunton, 1994)
			x	(Dineen, 2001)
Thalassia testudinum	x			(Dawes, n.d.)
		x		(Carlson Jr, et al., 1994)
			x	(Dineen, 2001)
Halodule uninervis		x		(IUCN, 2016)
			x	(IUCN, 2016)

Table 11.5: Coastal types where the four most common tropical species (which both thrive on muddy and sandy beds) have been observed to grow.

11.4.3 Implemented projects: planting techniques

Table 12.6 gives on overview of the 21 muddy seagrass restoration projects found in literature. For each project the specie name, the environmental conditions, planting techniques and successfulness of survival is given as far as knowledge is available. Often no definition of 'successful' was given in documentation. Three projects used seedlings as planting material (number 19 till 21), which is not further considered in this study.

No.	Location implemented project	Specie	Tropical	Environmental conditions		Planting technique		Survival of shoots	Reference
				Coastal type	Across-shore	Planting material	An-choring		
1	Kosung Bay, Korea	Zostera Marina	No	Bay	Subtidal	Rhizome fragments with shoot	TERFS (1S)	Approximately 60-75%, after 2 years	(Park & Lee, 2007)
2	Kosung Bay, Korea	Zostera Marina	No	Bay	Subtidal	Rhizome fragments with shoot	Shells (1S)	Approximately 60-95%, after 2 years	(Park & Lee, 2007)
3	Kosung Bay, Korea	Zostera Marina	No	Bay	Subtidal	Rhizome fragments with shoot	Staple (1S)	Approximately 75-95%, after 2 years	(Park & Lee, 2007)
4	Sriracha Bay, Thailand	Enhalus Acoroides	Yes	Bay	Intertidal	Rhizome fragments with shoot	No	26%, after 8 months	(Vichkovitten, et al., 2016)
5	Terschelling, Wadden Sea, the Netherlands	Zostera Noltii (Perennial, small seagrass)	No	Barrier	Intertidal	Rhizome fragments	?	No success	(Van Katwijk, et al., 2009)
6	Sylt, Wadden Sea, Denmark	Zostera Noltii	No	Barrier	Intertidal	Rhizome fragments	?	Successful	(Van Katwijk, et al., 2009)
7	Balgzand, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	Successful for one growing season	(Van Katwijk, et al., 2009)
8	Balgzand, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	sods	No	Successful for one growing season	(Van Katwijk, et al., 2009)
9	Balgzand, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	Successful for one growing season	(Van Katwijk, et al., 2009)

10	Texel, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	No success	(Van Katwijk, et al., 2009)
11	Terschelling, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	No success	(Van Katwijk, et al., 2009)
12	Terschelling, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	No success	(Van Katwijk, et al., 2009)
13	Terschelling, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	Successful for one growing season	(Van Katwijk, et al., 2009)
14	Friesland, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Rhizome fragments	?	No success (probably desiccation)	(Van Katwijk, et al., 2009)
15	Norfolk and Suffolk, England	Zostera Noltii	No	Estuary	?	Sods	No	Successful	(Ranwell, et al., 1974)
16	Port Moody Inlet, British Columbia, USA	Zostera Marina	No (sea-sonal)	Bay	?	Rhizome fragments	No	Sudden disappearance after what appeared to be successful transplants	(Butler, et al., 2011)
17	Butroe estuary in Bay Biscay, Spain	Zostera Noltii	No	Estuary	Intertidal	Sods	No	25% after 5,5 years (increase 8 times in extent)	(Valle, et al., 2015)
18	Swan Lake, China	Zostera Marina	No	Barrier (Lagoon)	?	Rhizome fragments	staples	100% survival after 4 months	(Zhang, et al., 2015)
19	Terschelling, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Seedlings	?	Successful	(Van Katwijk, et al., 2009)
20	Terschelling, Wadden Sea, the Netherlands	Zostera Marina	No	Barrier	Intertidal	Seedlings	?	Successful	(Van Katwijk, et al., 2009)

21	Balgzand, Wadden Sea, the Netherlands	Zostera Marina (Annual, big seagrass)	No	Barrier	Intertidal	Seedlings	?	Successful for 8 years (after that extinct)	(Van Katwijk, et al., 2009)

Table 11.6: Seagrass restoration projects implemented on muddy beds. ? = unknown. Non-tropical species show seasonal growth.

11.4.4 Implemented projects: erosion mitigation services

Table 12.7 gives an overview of the 5 studies found in literature which indicated the observation of an erosion mitigation service provided by a seagrass bed on a muddy coast. For each study information is given (as far as knowledge reaches) of the following factors: % of mud in the soil, coastal type, type of study, species names and characteristics like bed density and occurrence across-shore, and the erosion mitigation services provided.

	Variables	Study 1	Study 2	Study 3	Study 4	Study 5
Location characteris tiscs	*Location*	Italy, Venice Lagoon	Japan, Akkeshi-ko estuary	Germany, Konigshafe n	Australia, Great Barrier Reef Lagoon (4 locations)	USA, Florida Bay
	% mud		65			
	Coastal type	Barrier	Estuary	Bay	Barrier	Bay
Research type		Field	Field	Flume + field	Field	Field + modelling+ descriptive
Specie characteris tics (dominant specie)	*Name*	Zostera noltii	Zostera marina	Zostera marina	Z. capricorni (1, 2)/ Halodule. Uninervis* ** (3) / Halophila minor*** (4)	Thalassia Testudinum
	Tropical / Non-tropical	Non - Tropical	Non - Tropical	Non - Tropical	Tropical	Tropical
	Annual/ Perennial	Perennial				Perennial
	Intertidal/ subtidal	Intertidal		Intertidal	Intertidal	
	Density/ covering/ density	grass covers 20-60%	Biomass fluctuation over seasons from 10 – 258 g/m^2	200 shoots/ m^2	Biomass mean (g DW/ m^2) 1: 252,16 2:72,3 3: 5,01 4: 0,20	
Erosion Mitigation	*Wave attenuatio n*					Up to 80%

	Reduction current velocity		YES (spring + summer), ratio vegetated - unvegetated current velocities: from 0.25 ± 0.09 to 0.64 ± 0.59 (mean ± standard deviation).	18% ↓ of low flow (0.05 m s–1), 8% ↓ at higher flows (0.25 m s–1)		
	Bed stabilization	YES (summer)	YES** (spring + summer)	NO**		
	Bed elevation				YES (Z. capricorni) NO*** (Halodule. Uninervis / Halophila minor)	YES (inside + outside bed) (2,5 cm/year)
	Reduced coastline retreat					
Reference		(Amos, et al., 2004)	(Hasegawa, et al., 2008)	(Widdows, et al., 2008)	(Mellors, et al., 2002)	(Prager & Halley, 1999; Halley, 2000; Halley, et al., 1997)

*Table 11.7: Studies which indicated the performance of erosion mitigation services provided by seagrass species growing on muddy coasts. Research type indicates whether the study was based on field measurements or flume studies. Non-tropical species are subjected to seasonal change in biomass (low in winter). *Bed elevation includes terms like 'increased sedimentation' or 'sediment trapping'. **Bed stabilization includes term like 'reduced erosion of bed' or 'prevention of sediment bed resuspension'. *** structural species of low biomass do not trap sediments. ↓= reduction. DW=dry weight.*