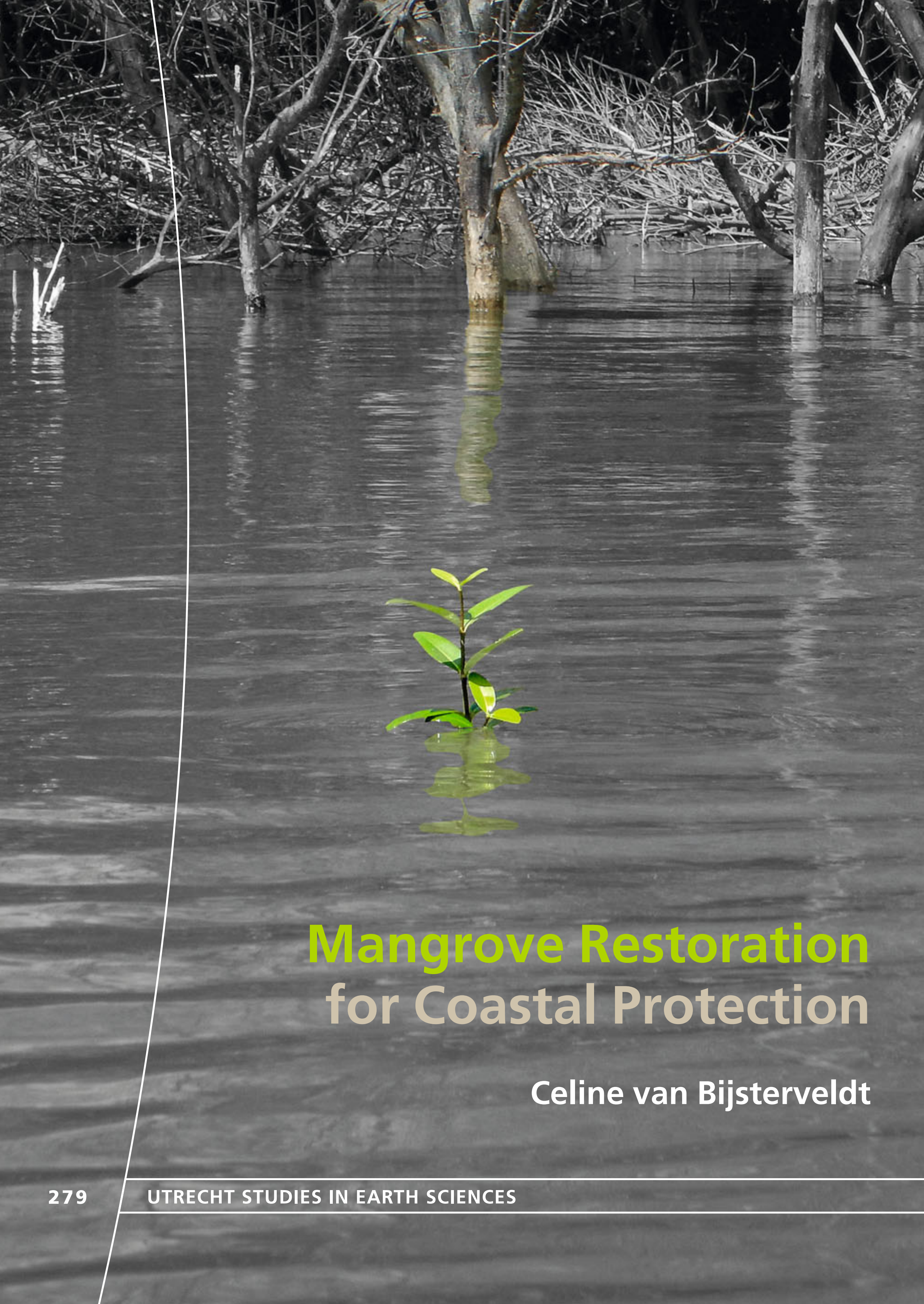

Mangrove Restoration for Coastal Protection

Celine van Bijsterveldt

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Mangrove Restoration for Coastal Protection

Celine Euphemia Johanna van Bijsterveldt

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for
Coastal Protection

Mangrovenherstel voor Kustbescherming
(met een samenvatting in het Nederlands)

Restorasi Mangrove untuk Perlindungan Pesisir
(dengan ringkasan dalam Bahasa Indonesia)

Proefschrift

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General Introduction

CHAPTER 1

General Introduction

Mangrove forests may help protect vulnerable coastlines from erosion and flooding. Coastal systems, such as deltas and estuaries, are often dynamic yet densely populated. In the tropics, these coastlines are naturally fringed by mangroves, forests that live on the border between land and water. Mangrove forests can attenuate waves and keep pace with sea level rise under the right conditions. However, mangroves along vulnerable coastlines have often been degraded, for instance, by pollution, logging, or conversion to other land uses. This raises the question: how can we best restore mangroves for the protection of vulnerable coastlines?

1.1 Mangroves build their own habitat

Mangroves are a diverse group of tropical tree species that grow between mean sea level and mean high water (Perillo *et al.*, 2009). They have one thing in common: they are equipped to grow on the border of the land and sea, where the soil is saline and frequently inundated (Ball, 1988). Mangroves cope with this challenging environment through various adaptations, such as through the ability to exclude, store or secrete salt (Parida and Jha, 2010), through the production of (crypto)viviparous seeds (Hutchings and Saenger, 1987), and most notably: through the growth of aerial roots (P B Tomlinson, 2016). Aerial root morphology varies between genera and species (e.g. pneumatophores, stilt roots, plank roots and root knees), but always consists of an above ground part that facilitates oxygen penetration to the subsurface part of the root system (Scholander, Van Dam and Scholander, 1955). Together, above and belowground root systems are responsible for building mangrove habitat. The aboveground part of mangrove roots efficiently attenuates incoming waves, creating calm conditions in which the sediment in the water column can sink to the bed. The many looping stilts and meadows of pencil-like pneumatophores effectively function as a sediment trap, raising the bed with minerogenic soil. In systems with high suspended sediment concentrations, sediment trapping is a process through which mangroves engineer their habitat and can even keep up with sea level rise (Balke and Friess, 2016). When there is little sediment in the water column, mangroves grow on their own dead material. The underground part of the root system slowly generates organogenic soil through the build-up of dead roots and leaves. A process through which mangroves engineer their environment in microtidal systems with little sediment in the water column (Mckee, Cahoon and Feller, 2007).

1.2 Mangroves provide many useful ecosystem services

The soil building and wave attenuating characteristics of mangroves make them interesting ecosystems for various purposes (Barbier *et al.*, 2011): The still waters in between above ground roots provide shelter for fauna and act as a nursery for fish (Hutchison, Spalding and zu Ermgassen, 2014). Furthermore, the sediment trapping function of above ground roots promotes deposition of potentially contaminating particles, filtering the water before it enters the ocean (e.g. Wong et al., 1997). In addition, below ground roots take up (excess) nutrients, adding to the water purification process (Keuskamp, 2014). Mangrove trees also generate and fixate organic material as they grow, thereby sequestering large amounts of carbon dioxide in above and belowground tissues (Donato *et al.*, 2011). Unlike terrestrial forests, the waterlogged and anoxic state of mangrove sediment hampers decomposition of dead roots and leaves. This facilitates the storage of large amounts of carbon belowground for long periods of time, making mangroves one of the most carbon-rich tropical ecosystems (Donato *et al.*, 2011). In addition to these regulating services, mangroves also provide materials such as wood for building and burning and edible fruits (Barbier *et al.*, 2011). Furthermore, the forests have aesthetic value, providing shade, recreation, and a habitat for wildlife, making it an appreciated ecosystem for tourism (Spalding and Parrett, 2019). Last but not least, the wave attenuating properties of mangroves in combination with their ability to build their own habitat make them valuable for coastal protection (Menéndez *et al.*, 2020).

1.3 Mangroves are in decline worldwide

Despite the recognized value of mangrove ecosystems, they have been declining for decades, with global mangrove losses of 2.1 % per year in the 1980s and 1990s (Valiela, Bowen and York, 2001). Global mangrove decline continued further in the 2000s, but at a lower rate of 0.18% per year (Richards and Friess, 2016). South-East Asia, the region with historically the most extensive (Bunting *et al.*, 2018) and diverse mangrove forests (Polidoro *et al.*, 2010), is also the region with the most significant mangrove loss (Polidoro *et al.*, 2010). Up to 50% of the original mangrove area has been converted to other land-uses, including urban encroachment, rice farming and oil palm plantations (Thomas *et al.*, 2017). However, commercial aquaculture has been, and remains, the main culprit of large-scale mangrove conversion, with the proportion of mangroves converted to commercial aquaculture reportedly ranging between 30 and 80% since the 1970s (Wolanski *et al.*, 2000; Hamilton, 2013; Thomas *et al.*, 2017). Mangroves are ideal locations for aquaculture, as the ecosystem naturally facilitates flushing and nutrient input through a network of creeks. The presence of small-scale aquaculture farms in mangroves have been reported as far back as 1400 (Herre and Mendoza, 1929), but large-scale conversions to aquaculture only came with a spike in the global shrimp market in the 1970s (Hamilton, 2013). The industrialization of aquaculture led to clear-cutting of entire mangrove ecosystems, often right down to the shoreline.

1.4 Degraded mangrove coastlines are vulnerable

The removal of mangroves from low-lying lands leaves the resulting landscape extremely vulnerable to erosion. The construction of aquaculture ponds often diverts existing rivers and canals, and cuts-off natural sediment input into the system. In addition, by removal of mangroves, the land-retaining function of below-ground root systems is lost (Sidik and Lovelock, 2013). The clearcutting of mangroves directly along the coast further weakens shoreline stability, as this removes the wave-attenuating buffer between land and sea. The loss of protective ecosystems services in combination with the absence of fresh sediment input has resulted in ongoing erosion of many former mangrove-mud coasts (Winterwerp *et al.*, 2013). Moreover, the typical gentle slope (1:1000) of (converted) mangrove-mud coasts makes that, when a big storm occurs, kilometres of exposed converted land can be lost to the sea at once (Van Wesenbeeck *et al.*, 2015). Sea level rise and increasing storm frequency and intensity all increase the risk of flooding and large-scale erosion along degraded mangrove-mud coasts. This not only increases the risk of losing arable land, but it also jeopardizes villages and cities, as deltas and coasts are among the most densely populated areas in the world (McGranahan, Balk and Anderson, 2007).

1.5 Mangrove restoration for nature-based flood defence

Seventeen percent of the world’s coastline is currently inhabited and at risk of flooding (van Zelst *et al.*, 2021). Many urban coastlines are protected by hard flood defences such as levees and breakwaters. Levees are designed in such a way that they can protect the land from the high water levels of a storm surge by a once in a 50- or 100- year storm (Figure 1-1). However, sea level rise is increasing the maintenance costs of these “grey” flood defences as water levels continue to rise. Furthermore, heightening levees with sea level rise is not always possible, as the width of a dike often needs to be increased as well. There is not always space for that type of maintenance if buildings rim the edge of the levee.

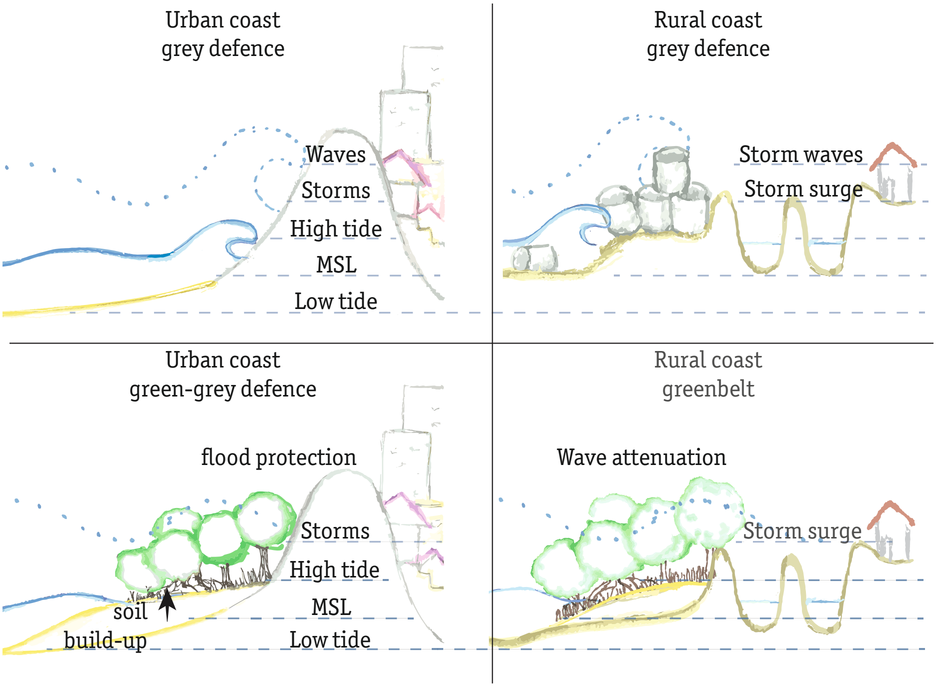

Figure 1-1: *Schematic cross-shore representation of urban and rural tropical coastlines (columns) and their vulnerability to storm surges and storm waves under traditional (grey) flood defence and (green) nature-based flood defence. Where grey defence structures need frequent heightening and widening to withstand rising water levels, green defence structures have the potential to grow with sea level rise.*

If a vegetated foreshore is present along the seaward edge of a levee, this can help strengthen the traditional coastal defence structures; vegetated foreshores, such as mangroves, tend to be able to eco-engineer themselves and outgrow rising water levels. In addition, the vegetation can reduce the wave height of relatively high frequency waves such as wind and swell waves, ensuring that relatively small waves arrive at the existing flood defence structure. For instance, a 100 m wide vegetated foreshore can cause a significant wind- and swell wave reduction (of between 13 and 66%) of (A. McIvor *et al.*, 2012). This means that levees behind a vegetated foreshore can be built lower or heightened less frequently than levees along an exposed shoreline. Van Zelst et al. (2021) estimate that along 27.6% of the world's populated coastlines that are prone to flooding, levee-height for protection against a once in a 100 years storm could be reduced by at least 25 cm when vegetated foreshores are present. On a global level, this would decrease construction costs with a total of 320.2 billion USD if levees of this standard would be constructed along all flood prone coastlines (van Zelst *et al.*, 2021). Construction of conventional flood defence is of course not feasible along all mangrove coastlines, and especially rural tropical regions often do not have the means to construct traditional flood defence structures to prevent villages from flooding.

In rural areas without conventional flood defences, mangrove forests alone have also shown to be effective flood defence buffers. For instance, the 2004 tsunami revealed how villages behind wide mangrove buffers were sheltered from destructive floods (Dahdouh-Guebas *et al.*, 2005; Danielsen *et al.*, 2005). A later study in India showcased how mangroves also shelter from storm surges, as the death toll after a cyclone in 1999 was much lower in villages behind wide mangrove forests compared to villages sheltered by narrower fringes (Das and Vincent, 2009). On a global scale, existing mangroves are estimated to protect at least 15 million people from flooding, and protect 65 billion USD worth of property from flood and storm damage on an annual basis (Menéndez *et al.*, 2020). However, for mangroves to provide significant coastal protection, the buffering greenbelts need to be significantly wider than the decametres wide fringes that currently border many of the converted mangrove coastlines. For instance, storm surges, which are very low frequency waves, are relatively poorly attenuated by mangrove forests. At least 1 km is required to cause a significant reduction in a storm surge (Montgomery *et al.*, 2019), whereas only 5% of the world's coastline is currently fringed by a greenbelt of that width (van Zelst *et al.*, 2021). Fortunately, both the significant land losses along many aquaculture coasts (Winterwerp *et al.*, 2013) and the declining productivity of remaining aquaculture ponds (Stevenson, 1997) have renewed an interest in restoring mangroves with their ecosystem services. The challenge is therefore to restore mangroves fringes that are wide enough to improve (existing) coastal protection and decrease the shoreline's vulnerability.

1.6 Challenges of mangrove restoration along aquaculture shorelines

With many of the shorelines in South-East Asia converted to aquaculture right to the shoreline, restoration of any significant mangrove forest width poses a major challenge. Mangrove restoration can be targeted seaward or landward. When looking for seaward restoration, the main challenge might be that many of these muddy shorelines currently show erosion (Winterwerp *et al.*, 2013; Van Wesenbeeck *et al.*, 2015; Luijendijk *et al.*, 2018)(Figure 1-2). In natural systems, coastlines that are subject to a period of erosion generally lack locations that are suitable for seedling establishment along the seaward edge.

When looking for mangrove restoration in landward direction, mangrove rehabilitation generally needs to be done inside (former) aquaculture ponds or agricultural fields

(Figure 1-2). This poses challenges in terms of inundation time, sediment availability, connectivity to seed sources, and all other issues related to the original conversion from mangroves to aquaculture. In addition, aquaculture ponds, though often unproductive and even abandoned, still have legal owners. Land tenure issues can therefore be problematic in any project attempting to restore mangroves into old aquaculture ponds.

On top of the challenges related to the rehabilitation of mangroves, there are challenges related to conservation of existing and restored mangroves as well. Of course, there are the threats that caused mangrove conversion in the first place (such as clear-cutting for land-use change), but new threats are arising with growing population densities in coastal regions. Increasing pollution, global sea level rise, and regional land subsidence may all affect remaining mangroves, especially where population densities are high. To successfully restore mangroves in densely populated regions where they are needed the most for flood defence, we need to obtain fundamental understanding of the pressures hampering the persistence and preventing the expansion of mangrove ecosystems (Figure 1-2).

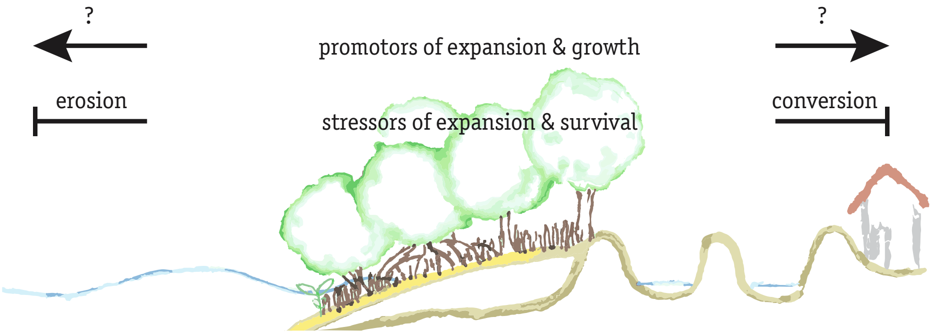

Figure 1-2: *Challenges to mangrove greenbelt restoration for coastal resilience in rural areas. Identifying and understanding the importance of different stressors to mangrove expansion and survival will help design successful restoration measures based on promotors of mangrove expansion and growth.*

1.7 The study area: a degraded, polluted and sinking mangrove-mud coast

A prominent example of such a mangrove system that is needed for coastal resilience, yet needs to be restored despite anthropogenic stressors, is the mangrove fringe of Demak regency, Central Java, Indonesia. Here, the ancient alluvial plane used to be covered with extensive mangrove forests, but most of these have been converted to rice culture and later aquaculture. The few mangroves that remain are under heavy anthropogenic pressure.

The government is attempting to restore a mangrove greenbelt along the shore to halt massive shoreline erosion that preceded with tens of meters per year. A consortium of international and local NGO's, companies, universities and governmental organizations has attempted to address two of the stressors in the system to aid natural mangrove expansion in the so called "Building with Nature Indonesia" project (Ecoshape, 2021). The first issue that was addressed, was the continued conversion of (restored) mangroves to aquaculture. This was addressed through coastal field schools (Roel H. Bosma *et al.*, 2020), implementation of sustainable mixed-mangrove aquaculture (R. H Bosma *et al.*, 2020) and a bio-rights approach to fund alternative livelihoods in exchange for steps taken towards mangrove conservation by the local communities (Wilms *et al.*, 2019). The second stressor was the ongoing coastal erosion and thereby mangrove retreat. This problem was addressed through the placement of permeable brushwood structures parallel to the coastline. These structures attenuated the oncoming waves, thereby creating calm water and subsequent sediment accretion (Winterwerp *et al.*, 2020). This way, the bed level seaward of the shoreline was raised with decimetres per season, building out elevated mudflats fit for mangrove recruitment with varying success.

1.8 Scope and aims

In this dynamic system, with all its negative and positive anthropogenic influences, I sought to elucidate the different drivers of mangrove dynamics at play, to gain a broader understanding of how mangroves may be restored and maintained along degraded and vulnerable tropical coasts worldwide. I divided this overarching aim into three main research questions: (1) What are drivers of natural mangrove dynamics along degraded mangrove-mud coasts? (2) How can that knowledge be used to assist mangrove expansion? (3) How do indirect anthropogenic stressors affect survival of established mangrove vegetation?

Question 1 is addressed in the next two chapters of this dissertation, in which I explore the natural forces that drive mangrove recruitment and retreat along degraded mangrove-mud coasts. I discuss this topic at an individual seedling level in chapter 2, where we looked at local sediment characteristics and topography in the field, and how that correlated with seedling presence. In chapter 3, I zoom out to landscape scale drivers of mangrove dynamics, using remote sensing techniques and cross-shore wave information to identify landscape characteristics, such as mudflats and isolated sand lenses (cheniers), that determine the tipping point between seaward mangrove expansion and mangrove retreat. Question 2 is addressed in chapter 4, in which we investigated how mangrove establishment may be assisted in such a degraded system through various restoration techniques. We assessed the effectiveness of ecological mangrove restoration through the placement of permeable structures, but also the effectiveness of past mangrove planting efforts. We also compared the two methods in a field experiment. Finally, Question 3 is discussed in chapter 5 and 6 of this dissertation, in which I investigate the anthropogenic forces at play on already established mangrove trees, to understand the fate of mangroves after restoration but under continued anthropogenic stress. In chapter 5, we discuss the effect of plastic pollution, which was abundant in the research area and a major problem in South East Asia in general (Jambeck *et al.*, 2015). We quantified pre-existing plastic on and in the forest floor and did an experiment to investigated if plastic pollution can suffocate mangrove trees when more than half of a mangrove tree's roots are covered by plastic. In chapter 6, we discuss the effect of land subsidence, which presented itself as one of the main drivers for coastal erosion during the project. Severe land subsidence is often caused by subsurface fluid extraction. It occurs in many delta's worldwide (Syvitski *et al.*, 2009) and can be multiple times faster than global sea level rise (Nicholls *et al.*, 2021). We therefore investigated how both mangroves and villages are impacted by this severe increase in water level to shed a light on the long-term persistence of mangroves as flood defence buffer in the face of accelerated sea level rise. Finally, in chapter 7, the synthesis, I discuss how factors discussed in the various chapters may influence each other; I present data and experiments that were outside the scope of the chapters, and I present gaps in knowledge that require or merit further research. Finally, I discuss how the information presented in this dissertation can aid mangrove restoration efforts.

My research was part of the BioManCo research project, in which the larger scope was to understand the existing sediment- and hydrodynamics in the area with their influence on the mangrove fringe. Besides the work on mangroves presented in this dissertation, the dissertations from Silke Tas (Tas, 2022), and Alejandra Gijón Mancheño (Mancheño, 2022) discuss the dominant physical processes in the area (Figure 1-3). Silke Tas investigated which processes govern cross-shore chenier migration. She modelled how cheniers will change and move under different conditions. Her work has a strong link to chapter 3 in this dissertation, in which I discuss the effect of cheniers on mangroves. Alejandra Gijón Mancheño focussed on the functioning of permeable structures constructed in the research area. She developed modelling tools to improve their design. Her work links to chapter 4, and chapter 6 in this dissertation, in which the effect of permeable structures on mangrove establishment and survival are discussed. Ultimately, the aim of the overall BioManCo project, and thus the collection of our dissertations was to achieve the understanding necessary to break the cycle of ongoing erosion along degraded mangrove-mud coasts.

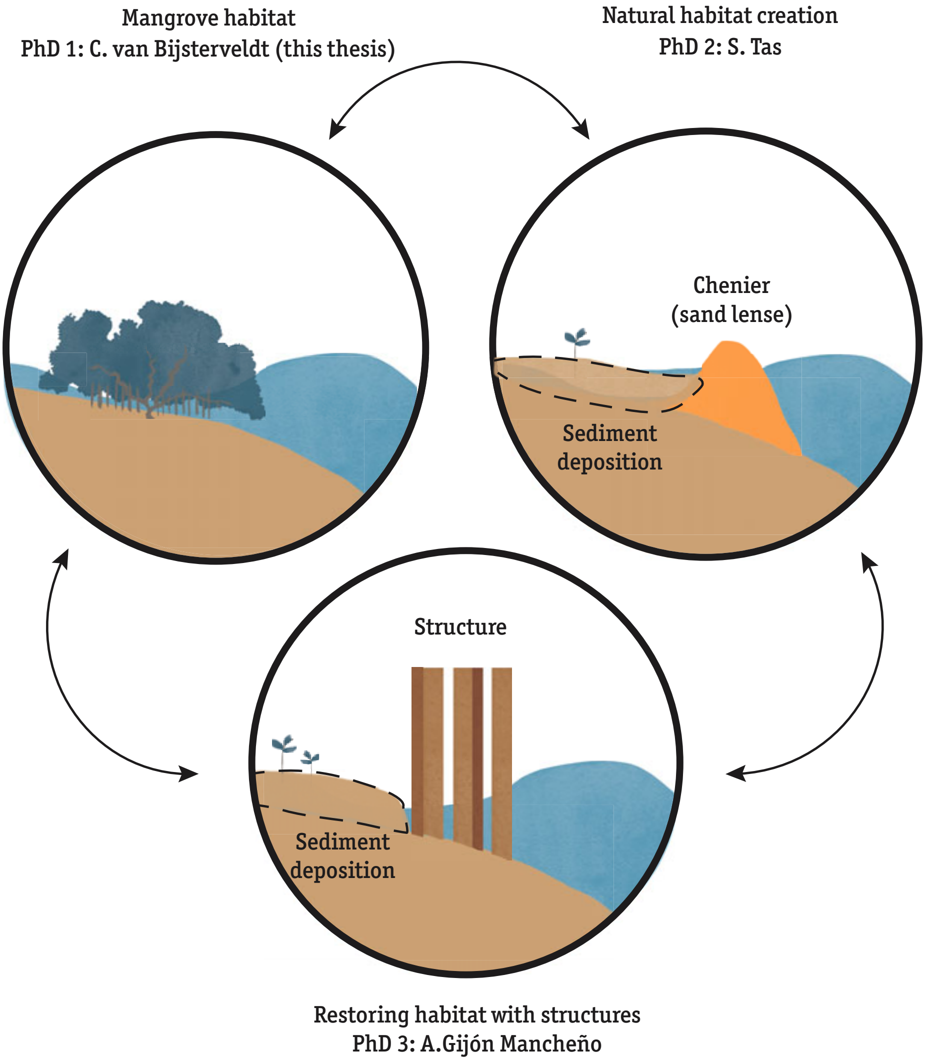

Figure 1-3: *Conceptual figure of the connection between the three PhD studies of the BioManCo project (figure reproduced with permission from (Mancheño, 2022)). This dissertation reports the work of PhD 1.*

Natural promotors of mangrove establishment

CHAPTER 2

Celine E. J. van Bijsterveldt, Bregje K. van Wesenbeeck,
Daphne van der Wal, Norma Afiati,
Rudhi Pribadi, Benjamin Brown & Tjeerd J. Bouma

published in Estuarine, Coastal and Shelf Science (2020) as:
How to restore mangroves for greenbelt creation
along eroding coasts with abandoned aquaculture ponds.

2.1 Abstract

Globally, erosion of muddy tropical coasts that are dominated by aquaculture ponds is an increasing problem. Restoration of mangrove greenbelts may counteract such erosion, by restoring the sediment balance. Hence, we aim to unravel the processes controlling natural mangrove regeneration in both “landward” (i.e., into aquaculture ponds) and seaward direction, using the fast eroding coastline of Demak (Indonesia) as case study. Firstly, we investigated which physical and chemical factors drive landward mangrove expansion by relating them to the presence/absence of mangrove seedlings in abandoned aquaculture ponds. Secondly, we investigated which physical parameters control seaward mangrove expansion by relating them to expansion and retreat at the sea-side of mature mangroves.

Landward mangrove expansion into abandoned aquaculture ponds was positively related to both emergence time (%) and sediment stability (i.e., shear strength), which are in turn both associated to bed level elevation and pond drainage. Surprisingly, there was no effect of soil chemistry. Seaward expansion of existing mangrove stands was strongly associated to foreshore morphology. Mangroves only expanded in the presence of an elevated mudflat, whereas the absence of a mudflat in combination with a concave (hollow) profile was associated with mangrove retreat. Our findings suggest that restoration of a mangrove greenbelt can be stimulated landward by improving drainage of abandoned aquaculture ponds. This enhances sediment stability and allows ponds to accrete. Seaward expansion can be induced by restoring foreshore morphology. Present results are discussed in the context of large-scale applications.

2.2 Introduction

Mangrove ecosystems are widely recognized for their ecosystem services such as carbon sequestration, supporting fisheries, providing timber and coastline protection against erosion and flood protection from storm surges and tsunamis (Alongi, 2008; Barbier *et al.*, 2011). However, over 35% of the world's mangroves worldwide have disappeared since the 1980s, mainly due to clearcutting and conversion to aquaculture ponds (Valiela, Bowen and York, 2001; Richards and Friess, 2016).

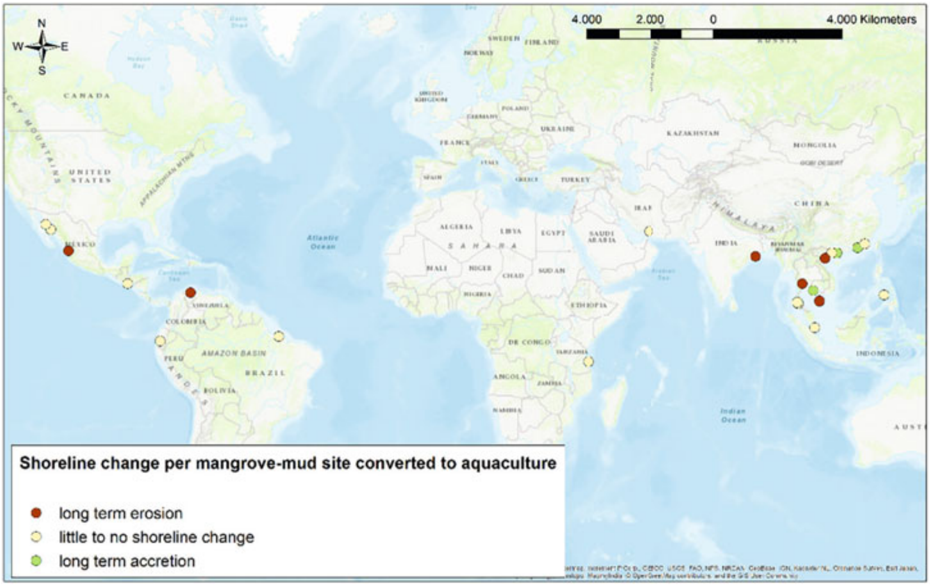

Figure 2-1: *Long term shore line changes from 1984 until 2016 as reported by Luijendijk et al. (2018) averaged per aquaculture coast. Aquaculture coasts displayed are sites designated by RAMSAR as internationally important wetlands, distinguished by mangroves and mudflats that have been (partly) converted to aquaculture (RAMSAR, 2018).*

Tropical coastlines, where mangroves have been converted into aquaculture ponds, consequently lost the aforementioned ecosystem services (Thampanya *et al.*, 2006). Especially the loss of sediment stabilization by mangroves is currently causing problems along multiple aquaculture coasts worldwide in the form of coastal erosion (Figure 2-1). Erosion of aquaculture areas can be as severe as several kilometers per year, putting coastal communities in danger (Van Wesenbeeck *et al.*, 2015). Soft engineering solutions such as restoring mangrove greenbelts seems to be the best way forward to prevent further erosion (Winterwerp *et al.*, 2013, 2016; Van Cuong *et al.*, 2015). Mangroves attenuate both tidal currents (Yoshihiro Mazda *et al.*, 1997) and waves (Winterwerp *et al.*, 2013), thereby diminishing erosion and enhancing sediment accretion (Chen *et al.*, 2018), so that the mangrove greenbelt can keep up with sea level rise and hence offers a climate-proof protection against coastal erosion (Gedan *et al.*, 2011; Lovelock *et al.*, 2015).

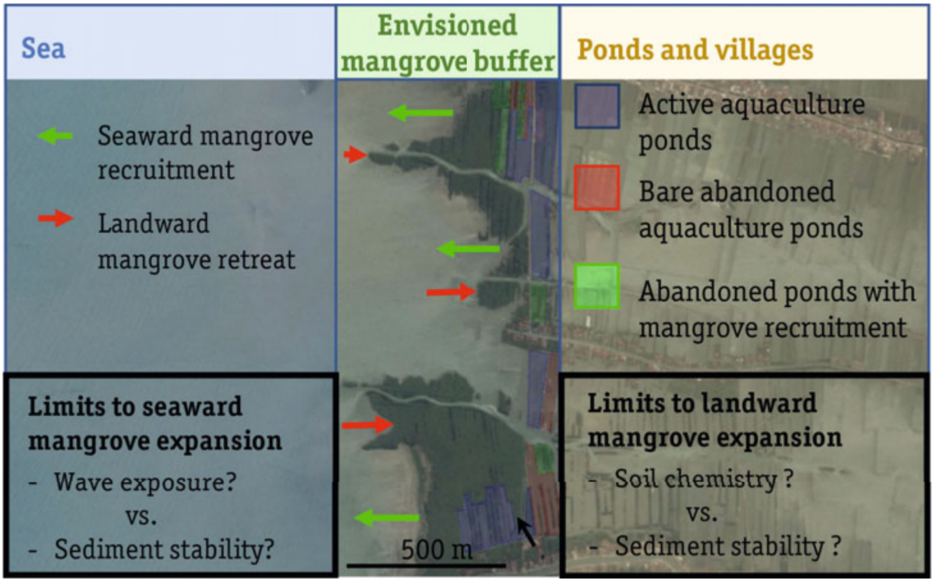

Figure 2-2: *Envisioned mangrove greenbelt buffer along the eroding aquaculture coastline of Demak, Java, Indonesia. Natural expansion of existing mangrove stands in seaward direction is occurring with varying success, possibly due to wave exposure or sediment instability. Natural mangrove expansion of existing mangrove stands landwards into abandoned aquaculture ponds also occurs with varying success, possibly related to unfavourable soil chemistry due to poor drainage or sediment instability. The role of these variables in landward and seaward mangrove expansion is investigated here.*

Creation of a mangrove greenbelt is only possible by seaward extension onto the coastal mudflats, or by landward extension in abandoned aquaculture ponds. Unfortunately, merely planting mangroves without habitat restoration in either seaward or landward direction, has proven to be unsuccessful in many projects (Ellison, 2000; Lewis, 2005; Bayraktarov *et al.*, 2016), with some projects showing mangrove establishment success no higher than 10 per cent (Bayraktarov *et al.*, 2016). Therefore, there is an urgent need to gain a basic understanding of which factors hamper mangrove establishment on both coastal mudflats and abandoned aquaculture ponds, to thereby gain insight in how to overcome thresholds for mangrove greenbelt creation by seaward or landward expansion. The coastline of North Java, Indonesia is an example of a region with extensive aquaculture and massive coastal erosion (Van Wesenbeeck *et al.*, 2015). The recreation of a mangrove greenbelt is envisioned as the best solution to counteract erosion. The co-occurrence of small isolated mangrove stands with various rates of both seaward and landward expansion/retreat in patches along this coastline, make this area an ideal model system to study which factors drive mangrove greenbelt establishment in seaward (i.e., on tidal flats) or landward (i.e., in abandoned aquaculture ponds) direction (Figure 2-2).

Landward mangrove expansion into abandoned aquaculture ponds can be challenging because they can lack many of the favourable environmental conditions needed for establishment (Stevenson, 1997). Poor tidal flushing is often observed in abandoned ponds (Van Loon *et al.*, 2016), and conditions that caused pond abandonment in the first place persist such as low oxygen concentrations due to eutrophication, high acidity, accumulation of toxicants like iron, aluminium (Raven and Scrimgeour, 1997; Stevenson, 1997), and methane (Strangmann, Bashan and Giani, 2008). Although mangroves are generally relatively tolerant to toxins, this type of soil chemistry is known to hamper seedling establishment (Strangmann, Bashan and Giani, 2008; Lewis, Pryor and Wilking, 2011). Hampered pond drainage can also lead to poorly consolidated mud (Fagherazzi *et al.*, 2017), which may be unfavourable for seedling anchorage (Balke *et al.*, 2011). Rehabilitation of mangroves in abandoned aquaculture ponds may thus be affected by various drainage related factors. Given this range of drivers, we aim to identify at the landscape scale the key-drivers restricting natural mangrove colonization in abandoned aquaculture ponds, using the coastline of North Java, Indonesia as case study.

In natural systems, seaward mangrove colonization tends to be episodic, only occurring if a sequence of favourable conditions follow each other in order to offer a Window of Opportunity (WoO) for establishment (Balke *et al.*, 2011, 2015). That is, a *i)* short inundation free period following propagule stranding, should be followed by *ii)* a wave free period to allow propagule anchoring and *iii)* sheet-erosion free period to allow seedling growth, for establishment to occur (Balke *et al.*, 2011, 2015). The last two steps of these windows of opportunity can be affected by two key drivers: sediment stability and wave exposure. Understanding which of these drivers is more important for seaward mangrove expansion in aquaculture areas can help in choosing the right measures for ecological mangrove restoration. Using the coastline of North Java, Indonesia as case study, we aim to identify which drivers determine if a window of opportunity for mangrove establishment occurs versus when mangroves are more likely to retreat.

Summarizing, in the present study we aim to investigate the potential for mangrove greenbelt creation along eroding aquaculture coasts by elucidating (1) what processes are the key-drivers that facilitate or hamper mangrove colonization in abandoned aquaculture ponds (i.e., landward expansion of mangroves) and (2) which factors drive expansion or retreat at the seaward mangrove front (i.e., seaward expansion of mangroves). We hypothesize that the main drivers controlling (1) the landward natural mangrove establishment in abandoned aquaculture ponds are either (1a) unfavourable chemical sediment properties like high acidity or (1b) poor sediment consolidation, reducing seedling stability (Figure 2-2). We hypothesize that the main factors controlling (2) seaward mangrove establishment are either (2a) the presence /absence of wave-sheltering foreshores or (2b) sediment consolidation, with poor consolidation reducing seedling stability and thus survival (Figure 2-2).

2.3 Methods

2.3.1 Site description

Hypotheses were tested along short transects in abandoned aquaculture ponds and mangrove stands along the coastline of Demak, Central Java, Indonesia. The coastline of Demak is a region with a history of extensive aquaculture, with a tidal range of approximately 1 meter and dominated by waves during the north-west monsoon in the wet season from November until February (MMAF, 2012). The coastline of Demak consists of alluvial deposits and used to harbour a mangrove-mud system several centuries ago according to old colonial maps. Since then, mangroves have been converted to rice paddies and later to aquaculture over the last century. The whole coastal plain 5 km inland bordered by the main road from Semarang to Demak and ranging from the city of Semarang, up to the river delta of the Wulan river was covered with aquaculture ponds (50*200 m) which were merely separated by pond bunds. From the year 2000 onward erosion has led to loss of large areas of aquaculture (Marfai, 2011; Van Wesenbeeck *et al.*, 2015). Despite rapid erosion of Demak's coastline, some abandoned aquaculture ponds start showing mangrove recruitment. The vision of the government is to expand these mangrove stands to a mangrove greenbelt along the coastline, to stabilize the coast and to protect the villages and persisting aquaculture from further erosion. However, mangrove recruitment that is currently taking place in abandoned aquaculture ponds seems patchy even though propagule limitation does not seem to be an issue, with some aquaculture ponds showing much better recruitment then others, hampering landward mangrove expansion (Figure 2-3). The few mangrove stands that have appeared along the former coastline are of variable stability in terms of seaward expansion or retreat. Stands show seaward expansion in some places, while they show retreat due to tree mortality at the coastline in other places (Figure 2-3). In this study, we use these contrasting mangrove states to analyse which processes control landward and seaward mangrove expansion along eroding aquaculture coastlines.

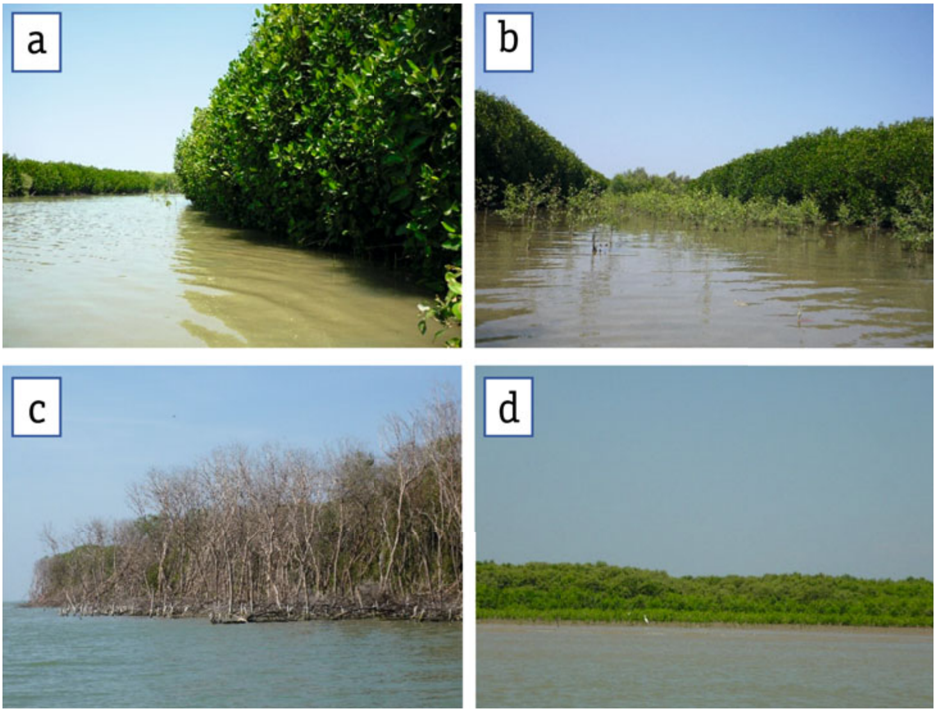

Figure 2-3: *Examples of mangrove retreat and recruitment in Demak (photos by Celine E. J. van Bijsterveldt).* ***a.*** *Abandoned aquaculture pond with little recruitment.* ***b.*** *Sheltered pond land inward of a mangrove stand with Avicennia marina recruitment inside the pond and planted Rhizophora mucronata trees outlining the old pond bunds.* ***c.*** *Mangrove retreat at the sea side of a mangrove stand.* ***d.*** *Mangrove recruitment seaward of a mangrove stand. Three subsequent stages of mangrove recruitment are visible.*

2.3.2 Mangrove recruitment in abandoned aquaculture ponds

Data collection in abandoned aquaculture ponds

The hypothesis that mangrove colonization in aquaculture ponds is driven by processes related to drainage was investigated during a field campaign in which eleven abandoned aquaculture ponds were assessed for their suitability for seedling recruitment (Figure 2-4).

Field study design

The ponds were selected based on abandonment, a comparable surface elevation at first sight and a variable seedling recruitment at first sight. Ponds were considered clearly abandoned when there was no sluice and at least one pond bund was absent. Data was collected in each of these ponds over the course of five weeks in September 2015, before the start of the wet season. Measurement stations were aligned in the pond along a transect starting at the most seaward pond bund in the shallowest area of the pond. The transect ran from the water level at low tide from the north-west pond bund and stations were spaced three meters apart across the pond until the creek (if that was present) to cover the full environmental gradient of the pond (Figure 2-4). Firstly, seedling density of *Avicennia* spp. was recorded in 1 m^2 quadrants at five stations in each pond as a response variable and indicator of suitability for seedling recruitment. Seedlings were defined as all mangrove plants that were below 1 m in height. At each station several explanatory variables were measured: (a) inundation time, (b) soil redox potential, organic carbon, sulphide concentration and pH to assess the soil chemistry and (c) soil water content, bulk density, shear strength and median grainsize to assess soil stability.

Figure 2-4: *Location of the study area Demak (red box) obtained from a Copernicus Sentinel-2 MSI satellite image and its orientation on Java, Indonesia (black box). Yellow points represent stations in abandoned aquaculture ponds, green points represent stations along expanding mangrove transects, and brown points represent stations at mangrove retreat sites. Blue box: a close-up of three abandoned aquaculture pond transects (yellow points represent stations), two expanding mangroves stand transects (green stations) and one mangrove retreat stand transect (brown stations).*

Collection of inundation time data

Inundation time was measured by Sensus Ultra dive data loggers (ReefNet Inc.), which were deployed in each pond for a period of two to three days. A reference water level logger was deployed in the subtidal area to account for the tidal fluctuations during the whole field campaign. Tidal information obtained from each pond logger was then compared to the tidal information logged by the reference logger during that period (*Figure A 2-1*). The vertical position of the pond loggers compared to the subtidal reference logger was determined by overlaying the maximum peak of the tidal curve obtained in each pond to the corresponding peak in the tidal curve of the reference logger. Maximum tidal peaks were chosen as a parameter because average water level could not be determined for sites situated at intertidal level. The vertical position of each pond logger compared to the reference logger was then used to extrapolate the number of times the pond logger was inundated over the whole field campaign. This extrapolation causes some uncertainty in actual inundation times per pond, but enabled us to reuse loggers and thereby include multiple ponds and sites and minimize uncertainties due to local pond and site differences. Submergence time (defined by the width of a peak in the tidal curve) measured by the pond logger was calculated to an average submergence time per day. Average submergence time per day was estimated for each pond-station by relating the relative surface elevation of each station along a transect to the relative surface elevation of the pond-logger. The relative position of each station compared to the pond loggers was measured with a laser (Spectra precision® LL300N). Relative surface elevation per site obtained with the laser could then be standardized to the average submergence time per day during the field campaign.

Collection of soil chemistry and soil stability related parameters

All chemical and physical soil parameters were collected under water logged conditions of the soil, around average water level in the tidal cycle. pH and redox potential were measured simultaneously at 10 cm sediment depth with electrodes (pH-electrode SentixSp, Electrode SenTixORP-T900) and a portable multimeter (multi 3430, WTW). Furthermore, shear-strength was measured with a vane (Geonor H-60 vane borer with tailor made head (h: 100 mm D: 80 mm)) at each station to get an indication of the bed consolidation. All other soil parameters were obtained through sediment sampling. Organic carbon content, bulk density, water content and median grainsize were determined from sediment samples, from the upper 5 cm soil at each station. Sediment samples were collected with a decapitated 100 mL syringe and stored in a fridge until further analysis. Sulphide concentration was determined from 3 mL soil samples that were collected at 5 cm depth in the sediment and were instantly stored in 20 mL Zinc Acetate (ZnAc) 2% solution to prevent oxidation of sulphide. Sulphide concentrations were determined according to a sulphide assay protocol (Trüper and Schlegel, 1964). Wet sediment samples were placed in a container with known volume, weighed and placed in a stove (105 °C) for at least 24 hours and weighed again to determine bulk density and water content within six weeks after collection (samples had been stored at 4 °C). Dry samples were then placed in a 350 °C stove for 24 hours and weighed again to determine organic carbon content. Finally, grainsize distribution of freeze-dried subsamples was determined with a Malvern (Mastersizer 2000, Malvern Instruments).

Statistical analysis of abandoned aquaculture pond data

To test the effect of the different soil chemistry and stability related parameters in aquaculture ponds on mangrove recruitment we used a redundancy analysis (RDA) in R package (R version 3.4.4) vegan. This ordination technique shows how much of the variation in mangrove recruitment (seedling density) is explained by the explanatory parameters measured (inundation time, soil redox potential, organic carbon, sulphide concentration, pH, water content, bulk density, shear strength and median grainsize), and to analyse how the explanatory variables relate to each other.

Data preparation and transformation

Prior to the redundancy analysis, two stations with a missing value in pH were omitted from analysis. Seedling density as a response variable was highly zero-inflated and could therefore not be transformed to normality; instead this variable was categorized into three ecologically meaningful classes: “no seedlings present”, “one seedling present”, “multiple seedlings present” and added to the ordination analysis as an untransformed response variable matrix with three binary variables.

Testing collinearity of explanatory variables

All explanatory variables were tested for collinearity prior to the redundancy analysis, by calculating their variance inflation factors (VIF). VIF values are derived from the R^2 values after regression of each individual variable against all other variables (Legendre and Legendre, 2012). VIF scores of all explanatory variables were below 3, the most conservative threshold value for collinearity mentioned in literature (Zuur, Ieno and Elphick, 2010). None of the explanatory variables was therefore omitted from further analysis based on collinearity.

Redundancy analysis on seedling density classes

A redundancy analysis (RDA) was run to investigate the difference between stations based on classes of seedling density and how the measured variables explain those differences. The analysis was run on the seedling density classes’ matrix and explanatory variable matrix. A permutation test was performed 999 times to test if this relationship between the matrix of explanatory variables and the matrix of response variables was statistically significant (Borcard, Gillet and Legendre, 2018).

2.3.3 Expansion and retreat in sea-side mangrove stands

Data collection in sea-side mangrove stands

To understand what parameters related to sediment stability and wave-exposure drive seaward mangrove expansion and mangrove retreat in existing mangrove stands along Demak's coastline, we used a combination of field-derived data and Geographical Information Systems (GIS) derived data.

Field study design and collection of sediment stability related parameters in the field

Five expanding mangrove sites and five retreating mangrove sites were randomly selected in the study area (Figure 2-4). All mangrove stands within a one hour trip by boat from our field station that showed mangrove expansion or mangrove retreat were included as possible sites for a transect. We defined expanding mangrove stands as stands with seedling recruitment seaward of the stand and progressively older trees towards the land. We defined retreating mangrove stands as mangrove stands characterized by uprooted mangrove trees along the seaward edge. At each site, several sediment stability related parameters were measured during the same field campaign in September 2015. Parameters were collected along transects running from 3 meters inside the mangroves stand, towards the subtidal zone (Figure 2-4). Soil water content, bulk density, shear strength and median grainsize were measured at each station in the field as explanatory variables related to sediment stability. Seedling presence was measured in 1 m^2 quadrats at each station as one of the mangrove response variables.

Collection of wave exposure related parameters

Field derived parameters

At each of the aforementioned transect stations, mudflat elevation relative to a local water level logger was measured, from which submergence time and mudflat-slope were derived as local explanatory variables related to wave exposure. Waypoints of each station in the field were marked in a hand held GPS (garmin etrex) so that GIS parameters could be extracted and added to the dataset.

GIS derived parameters

To further understand drivers of mangrove retreat and expansion along the coast as a result of wave exposure at land-scape level, several additional explanatory variables were obtained by GIS analysis. Mudflat width from each sampling station at low tide, outer boundary shelter width (for instance sand lenses along the coast and sea walls), sand bank width at low tide, and wind fetch at low tide in north-west direction were chosen as proxies for wave exposure.

In order to obtain these wave exposure proxies by GIS analysis, two Copernicus Sentinel-2 MSI satellite images, that were atmospherically corrected using the Sen2cor module in SNAP software, were used to analyse the effect of potentially sheltering geomorphological features such as sand banks and mudflats on the mangrove cover in Demak. The images have a 10 m spatial resolution. Two images with low cloud cover were selected. The first image was acquired at low tide (-24 cm relative to MSL) with an acquisition date close to the period of the field campaign (acquisition date: 7 Oct 2015). These selection requirements ensured that dynamic wave exposure related features such as sand bars and mudflats would be visible and would be located (close to) the position they had during the field campaign. The second image was selected based on a similar tidal level as the first image (-23 cm relative to MSL) and an acquisition date shortly after the stormy wet season (15 January 2016) to detect relevant mangrove cover change related to the explanatory variables as measured during the field campaign before the wet season. Cloud and cloud shadows were flagged in each image using QSC values (produced by Sen2cor). The satellite images were then reprojected to UTM49S. The NDVI, i.e. Normalized Difference Vegetation Index, was calculated based on surface reflectance in the red (R) and in the near-infrared (NIR) spectral bands of the images following (NIR-R)/(NIR+R). These NDVI rasters were then classified into 3 classes for fetch analysis: NDVI > 0.3 = vegetation, NDVI < -0.04 = water, -0.04<NDVI<0.3 = potential geomorphologic shelter (exposed sediment). These cut-off values resulted in classes that corresponded to land cover states as observed in the field, except from seedling covered mud-flats which could not be distinguished from bare mudflat.

The distance to sheltering features in north-west direction from each sampling station was obtained by analysis of lines drawn in north-west direction from each sampling station using the bearing distance to line tool in ArcGIS. Underlying classified raster values (vegetation, water and exposed sediment) were then extracted at points with a 7.07 meter interval (i.e., half the diagonal pixel length) along each bearing line. Resulting lines contained alternating values of mangrove, mud and water depending on the geomorphology obtained from the satellite images (Figure A 2-2). Puddles of water on the mudflat and small boats in the water were excluded from the analysis using a smoothing algorithm (*Figure A 2-3*). This smoothing process was done by assigning the raster value found in the majority of five cells: the cell of interest and two cells before and after the cell of interest along the bearing line (*Figure A 2-3*). Small shifts in mudflat and water border could not be avoided with this method (*Figure A 2-3*). Therefore, the smoothened bearing line values were only used to derive the explanatory variables mudflat width, sand bank width and wind fetch length. The width of each of these parameters was obtained from the location of border cells (e.g. water-mud transition) relative to another border cell of interest. For instance, fetch length was determined by subtracting the distance from the start of the bearing line of the outer water-to-mud transition from the minimum distance of mud-to-water transition along the same bearing line. Fetch length of more than 2 km from the coast was considered infinitive fetch with maximum wave exposure. Mangrove vegetation retreat and expansion between 2015 and 2016 were derived from the unsmoothed NDVI classification raster as response variables.

Statistical analysis of sea-side mangrove stand data

For statistical analysis of the factors that could explain mangrove expansion seaward, we again used a redundancy analysis. Mangrove expansion parameters from both the field (seedling presence) and GIS (mangrove expansion and mangrove retreat) were used as response variables, because seedlings could not be detected in the satellite images. Sediment stability related parameters and wave exposure related parameters from the field and GIS analysis were also all considered for the analysis as explanatory variables.

Data preparation

Prior to analysis, six missing values in the shear strength data collected at retreating mangrove sites were substituted with the maximum measured value of the shear strength values in the total dataset. This substitution was chosen because the missing values were a result of extremely high soil impenetrability due to numerous roots and shell fragments in the sediment that could have damaged the vane head. Using the maximum measured shear strength instead is justified at these sites, as these sites clearly had an extremely high shear strength and no reliable missing value regression could be made. Although the use of a shear strength value that is lower than the actual shear strength likely causes an underestimation of the effect of shear strength on mangrove retreat. Finally, all stations that lacked information on sediment variables were omitted from the analysis resulting in a full dataset with 29 stations, divided over 10 cross-shore transects with one station inside the mangrove forest, one station at the mangrove edge and one subtidal station.

Omission of collinear variables

VIF analysis was run on the entire explanatory data matrix to test for collinearity among the explanatory variables for seaward mangrove expansion. VIF analysis showed that the majority of VIF values were above 3, with relatively high VIF scores for median grain-size (4.5), soil water content (7.6), bulk density (3.4), maximum shelter at low tide (5.9), sand bank width (16.6), wind fetch at low tide (9.0), and mudflat shelter at low tide (20.9). We therefore decided only to include shear strength as most relevant variable for the “stability hypothesis”, and to include slope and mudflat width as most ecologically relevant field and GIS derived variables for the “wave exposure hypothesis”.

Redundancy analysis on mangrove states

The RDA was then performed on a response variable matrix composed of seedling presence in September 2015 (measured in the field), mangrove expansion and mangrove retreat based on NDVI satellite data between October 2015 and January 2016. The response variable matrix with these three response variables was then subjected to a relativity transformation to obtain relative mangrove response values per site. This transformation ensured that sites with both GIS derived mangrove expansion values and seedling presence values from the field, obtained a total score that was equal to sites with just mangrove expansion or seedling presence, as these three response variable combinations are ecologically similar. Finally, a permutation test was performed again to test if the explanatory variables (slope and mudflat shelter) explained the distribution of response variables significantly.

2.4 Results

2.4.1 Mangrove recruitment in abandoned aquaculture ponds

The redundancy analysis showed that all soil stability and soil chemistry parameters together, explained 16% (Ezekiel adjusted R^2) of the variation in seedling density in abandoned aquaculture ponds ($F = 2.1$, p = 0.03, $df = 9$). Furthermore, only environmental variables with their major component along the horizontal axis significantly explained seedling presence in abandoned aquaculture ponds ($F = 20.1$, p = 0.02, $df = 1$) (Figure 2-5). Despite the non-significant relation between the vertical components and seedling density class, we also plotted these environmental variables in order to provide basic insight in the correlations among environmental factors (Figure 2-5). The results show that mainly high soil stability and low submergence time are plotted along the horizontal axis and are therefore related to seedling presence. Especially high soil stability (shear strength) correlates with stations where multiple seedlings were found in abandoned aquaculture ponds (Figure 2-5). Meanwhile, all soil chemistry related parameters such as redox potential, sulphide content and pH are aligned with the vertical axis of the plot and are therefore no significant predictors of seedling presence in abandoned aquaculture ponds. Finally, many of the measured explanatory parameters show expected relations among each other, such as high shear strength with low soil water content and high redox potential with high sulphide concentrations (Figure 2-5).

2.4.2 Mangrove expansion and mangrove retreat along the coast

To understand what processes are most important for natural seaward mangrove expansion in front of abandoned aquaculture coasts, we analysed the effect of parameters related to wave exposure (mudflat width and slope) and of soil stability (shear strength).

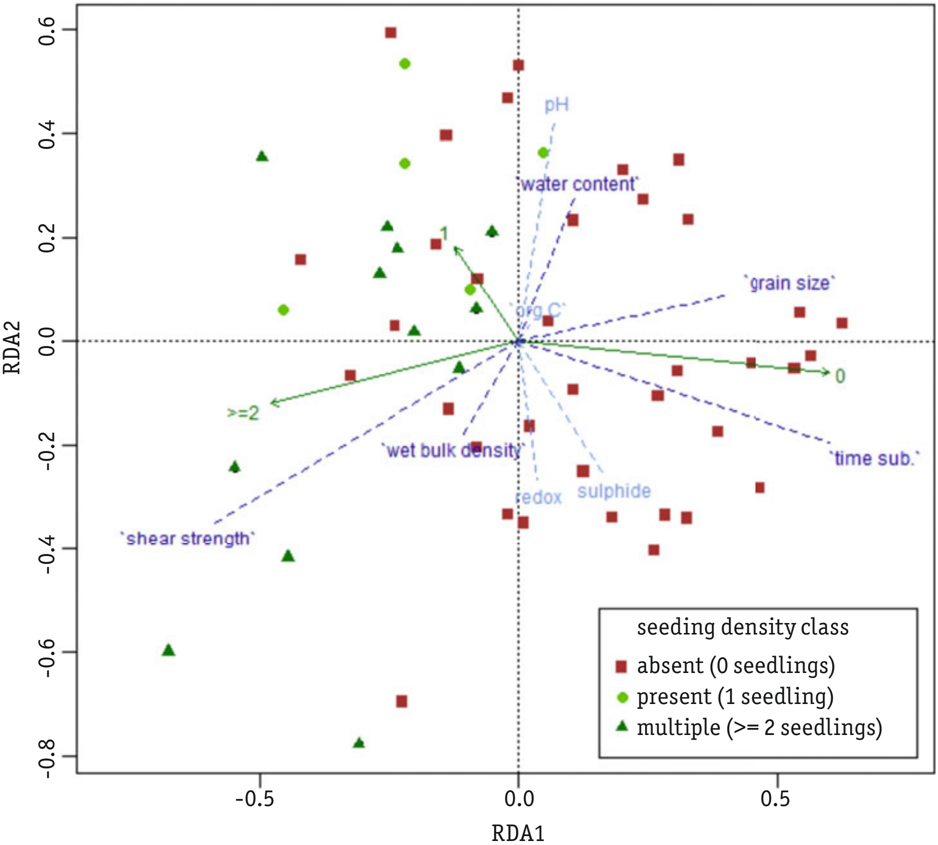

Figure 2-5: *Statistical analysis of parameters controlling mangrove establishment in abandoned aquaculture ponds. The RDA correlation plot relates three seedling density classes (i.e., absent, present, multiple) to the environmental parameters. Our analysis covered 53 stations spread out over 11 abandoned aquaculture ponds. Stations are plotted according to their similarity in number of seedlings, and coloured according to the number of seedlings observed (see legend). The correlation between stations in terms of seedling density is displayed with green solid vectors. The relation between seedling density and every explanatory variable can be interpreted by the direction of the dashed vector of interest to the response category vector. Vectors in the same direction are positively correlated, vectors in the opposite direction are negatively correlated and perpendicular vectors are not correlated. E.g., the shear strength arrow roughly points in the same direction as the arrow for stations with multiple seedlings, indicating that high shear strength often coincides with high seedling density. Environmental parameters that we investigated are depicted as dashed vectors and coloured by group: submergence time (% per day) in blue; soil stability parameters shear strength (kPa), median grainsize (µm), soil bulk density (g cm^{-3}) and soil water content (%) in dark blue; and soil chemistry related parameters soil acidity (pH), sulphide concentration in pore water (µM), redox potential (- mV) and soil organic carbon content (%) in light blue. Only environmental parameters along the horizontal RDA axis explain the variation in seedling density significantly, i.e. 14% (R^2 adj. RDA1= 14%, $p < 0.05$ and R^2 adj. RDA2 =1%, $p = 1$).*

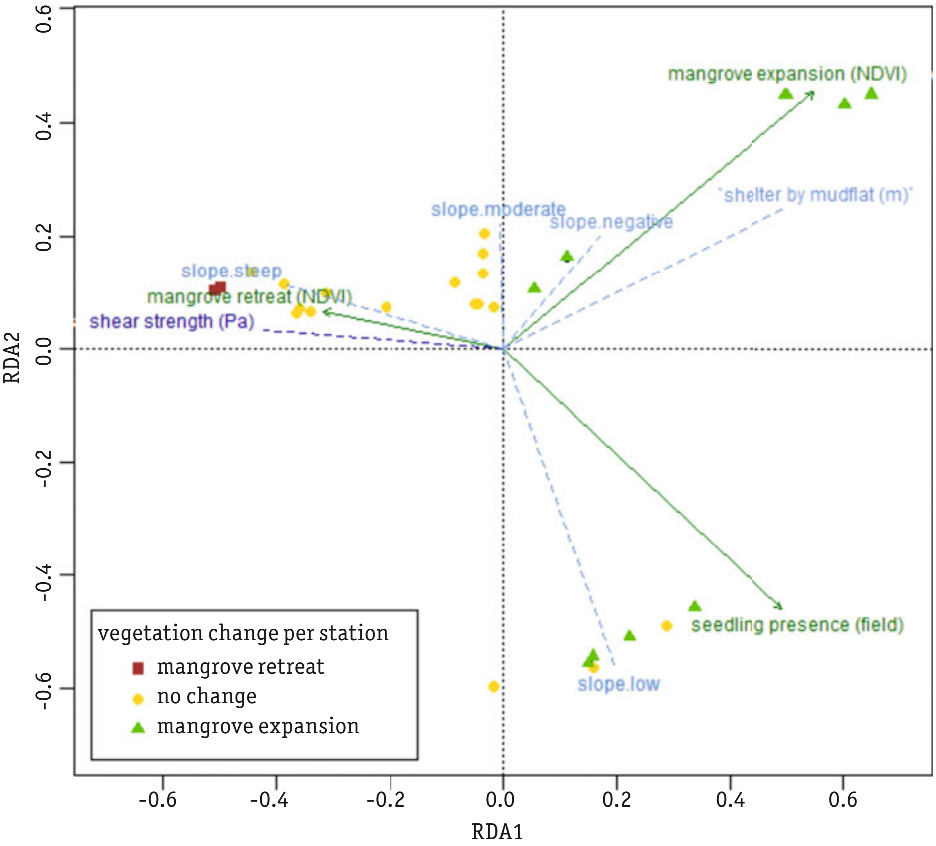

Figure 2-6: *Statistical analysis of parameters that drive seaward mangrove expansion. The RDA correlation plot shows the 29 stations that were investigated, coloured according to the condition as observed in the field and with GIS: seedling presence or mangrove expansion (green dots), stations that did not change state over the wet season (yellow dots) and stations that showed mangrove retreat (brown dots). We related three seaside mangrove conditions (1: mature mangrove retreat landward based on NDVI change over the wet season of 2015-2016, 2: mature mangrove expansion seaward based on positive NDVI change over the same wet-season and 3: seedling presence in the field) with environmental parameters related to soil stability and wave exposure. The relation between the three seaside mangrove conditions (green, solid arrows) and the environmental parameters related to wave exposure (light blue dashed arrow) and soil stability (dark blue dashed arrow) can be deduced from the direction of the arrows similarly as in Figure 2-5. In general, all environmental parameters that work along the horizontal axis explain 17 % of the variance in seaside mangrove dynamics ($p < 0.01$) and along the vertical axis explain 11 % ($p < 0.05$) of the variance.*

The redundancy analysis revealed that 33 % (Ezekiel adjusted R^2) of the variation in mangrove expansion and retreat along the coastline can be explained by the wave exposure and soil stability parameters that we analysed (Figure 2-6, $F = 3.8$, $p < 0.001$, $df = 5$). Sites with mangrove expansion (stations with either seedling presence in the field or NDVI increase over the wet season) generally had low slopes (0-2 $cm\ m^{-1}$) or even negative sloping foreshores, which means that they were protected from waves by a mudflat or sand bank with a higher surface elevation than the bed level at the station itself. Mudflat width also showed a clear positive correlation with mangrove expansion over the wet season of 2015-2016. Conversely, stations where mangrove retreat had taken place or where no vegetation change had occurred had moderate (2-5 cm m^{-1}) to steep (5-18 *cm* m^{-1}) foreshores, which would make those sites more vulnerable to breaking waves. Finally, sites where mangrove retreat occurred generally had a higher shear strength than sites with mangrove expansion (Figure 2-6), indicating that enhanced soil stability is not a main driver of seaward mangrove expansion.

2.5 Discussion

The rapid decline in coastal mangrove areas, caused by clear-cutting, land-use change to cities and aquaculture ponds and eventually coastal erosion, drives an increasing attention for mangrove conservation and restoration. For proper mangrove restoration, a better understanding of factors that drive mangrove recruitment is essential. The current research focusses on main drivers limiting mangrove expansion landward and seaward separately. Present findings are discussed in the context of coastal management aimed at mangrove greenbelt creation to counteract erosion.

2.5.1 Method uncertainty

In general, the redundancy analyses of mangrove restoration state in aquaculture ponds and coastal sites revealed various factors that could significantly explain those states. In hindsight, current correlations could have been stronger due to some uncertainties in our response variables. The field dataset of seedling density contained a large number of zero's. The chances of including a seedling in the quadrat through random sampling per station was low due to the combination of a relatively small quadrat size and a relatively low amount of seedlings. We therefore ended up with a value of zero seedlings in some stations where there were actually some seedlings around, but that just did not happen to fall in our quadrat. Increasing the number of quadrat measurements per station, or increasing the quadrat size could have decreased the zero-inflation of our dataset and could potentially have made the present correlations stronger. In addition, the response variables that were quantified using the Sentinel-2 satellite images had a spatial resolution of 10 meters, which means that lateral changes in mangrove cover smaller than ca 10 meters could not be detected. For mangrove expansion, this uncertainty was minimized by measuring seedling presence in the field. For mangrove retreat, the use of non-freely available satellite images of higher resolution could have detected smaller changes in mangrove retreat, where mangrove state was now classified as stable. This may have made the present correlations stronger, although one may wonder how important it is to detect changes of less than ten meter over a single wet season. Despite this relative course 10 m resolution, the explanatory variables that we chose did explain the variation in mangrove states significantly, yielding meaningful management implications.

Table 2-1: *Descriptive statistics of all parameters measured related to landward expansion in aquaculture ponds; and parameters measured in the field and derived from GIS related to seaward expansion of mangrove stands.*

Parameter	**n**	**Min**	**Mean**	**Median**	**Max**	**SD**
Related to landward mangrove expansion (in aquaculture ponds)						
pH	53	6.6	7.2	7.1	8.0	0.3
Redox potential (mV)	53	-226.3	-146.0	-153.1	-29.5	46.8
Sulphide concentration (µM)	53	395.3	2484.3	1366.4	11181.3	2129.7
Median grain size (µm)	53	5.1	6.0	6.0	7.7	0.4
Soil water content (%)	53	54.4	59.8	59.8	65.5	2.5
Organic carbon content (%)	53	6.8	9.7	9.5	13.8	1.5
Wet bulk density (g cm^{-3})	53	1.2	1.4	1.4	1.7	0.1
Shear Strength (kPa)	53	0.3	1.0	1.0	2.4	0.4
Time submerged per day (%)	53	5.1	36.0	24.9	100.0	28.5
Seedling Density (m^{-2})	53	0.0	1.3	0.0	16.0	3.1
Related to seaward mangrove expansion (from mangrove stands)						
Measured in the field						
Median grain size (µm)	29	5.1	11.6	6.5	144.8	25.7
Soil water content (%)	29	26.8	57.2	59.4	66.0	7.9
Wet bulk density (g cm^{-3})	29	1.2	1.4	1.4	1.7	0.1
Shear Strength (kPa)	29	0.3	1.5	1.0	4.0	1.2
Time submerged per day (%)	29	0.0	49.6	34.2	100.0	40.0
Slope (m m^{-1}), incl. as factor:	29	-0.1	0.03	0.00	0.2	0.1
negative	8	-0.083			-0.003	
low	7	-0.003			0.003	
moderate	7	0.003			0.056	
steep	7	0.056			0.178	
Derived from GIS						
Maximum shelter width (m)	29	0	284.3	141.4	735.4	288.7
Sand bank width (m)	29	0	40.0	0	134.4	50.7
Water fetch at low tide (m)	29	77.8	1143.5	2000	2000	908.6
Shelter by mudflat (m)	29	0	125.1	56.6	537.4	176.3

2.5.2 Mangrove recruitment in abandoned aquaculture ponds

We hypothesized that landward natural mangrove recruitment in abandoned aquaculture ponds would be driven by processes related to either (1a) unfavourable soil chemistry or (1b) poor soil stability. Our results showed that soil chemistry related parameters do not seem to be important predictors of seedling establishment in abandoned aquaculture ponds. Relations among chemical parameters were as expected, e.g. a high pH seems related to a more positive redox potential and highly negative redox values are correlated with sulphidic conditions (Figure 2-5). However, all chemical soil parameters were aligned with the vertical RDA axis and could hence not be clearly related to seedling abundance (Figure 2-5). The fact that the soil chemistry parameters measured in this study did not influence seedling abundance significantly is somewhat surprising, as nutrient rich effluents of active aquaculture have been known to cause eutrophication (Nóbrega *et al.*, 2014) and unfavourable conditions for mangrove establishment (Strangmann, Bashan and Giani, 2008). The redox values and pH values found by Strangmann et al. (2008) (i.e., redox: -276 mV - 294.0 mV, pH: 7.6 - 8.3) and Nóbrega et al. (2014) (i.e., redox: 36 mV – 256 mV, pH: 6.9 - 7.4) in sediment near shrimp farms are variable and cover a large range of values including values found in our study (Table 21). Both studies show that soil chemistry is different in impaired mangroves vs. pristine mangroves, but especially in terms of phosphate (Nóbrega *et al.*, 2014) and methane (Strangmann, Bashan and Giani, 2008). The pH does not actually appear to be much different between such sites in these and our studies. Redox potential does seem to be lower in sites near shrimp farms in both studies and especially redox potential values found in Strangmann et al. (2008) are comparable to values measured in our aquaculture ponds (Table 2-1). However, these values can also be found in their pristine mangrove sites (Strangmann, Bashan and Giani, 2008). The large variation in redox potential found within sites (even pristine sites) in these studies and the small explanatory power of these parameters on seedling presence found in our study, suggest that pH and redox potential are possibly poor indicators for mangrove establishment.

Our results do show a much clearer picture with regard to the effect of soil stability on seedling density in abandoned aquaculture ponds. Shear strength showed a clear positive correlation with high seedling abundance (Figure 2-5) and seems to be related to low water content and high median grain size from the redundancy analysis. The grainsize analysis showed that abandoned aquaculture ponds are generally filled with muddy sediment of a low grain size and that there was little variation in grainsize overall (Table 21). Sediment of such a low grainsize is usually slow in consolidation, as the particles are very light and are therefore easily resuspended. Slow consolidation can result in a liquid mud layer which has a high water content and can easily be mobilized (Winterwerp *et al.*, 2012).

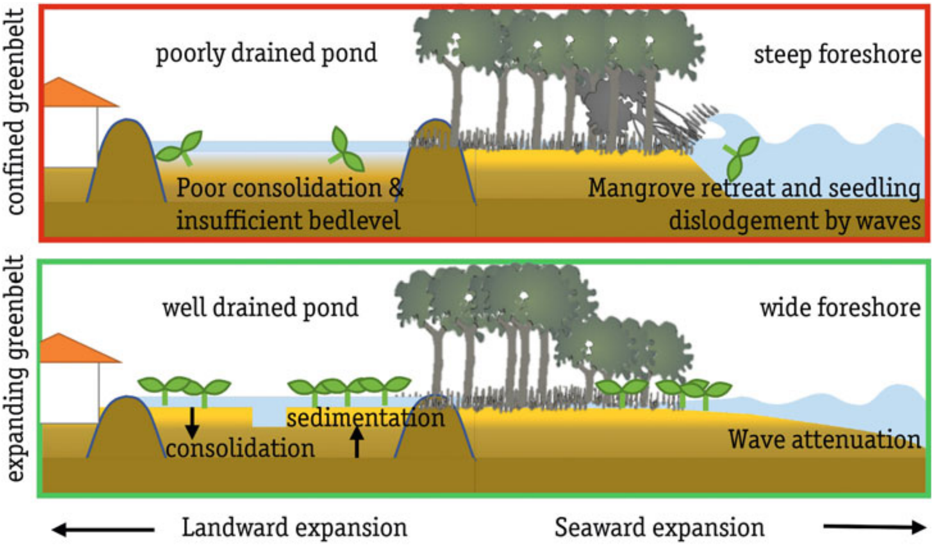

Figure 2-7: *Conceptual figure showing factors that negatively (red box) and positively (green box) influence mangrove greenbelt expansion. Landward expansion: poorly drained aquaculture ponds may have insufficient sedimentation to reach a good surface elevation (red box), or contain poorly consolidated sediment too unstable for seedling anchorage (red box). Conversely, ponds with improved hydrological connectivity would have better sediment consolidation due to improved drainage and more sediment input resulting in accretion (green box). Seaward expansion: existing mangroves are at risk for collapse when there is a deep foreshore with a steep profile (red box) and are more likely to expand seaward when there is a foreshore with a gentle slope like a mudflat (green box).*

The low critical bed-shear stress of fine sediment (Mitchener and Torfs, 1996) is known to offer very little stability to mangrove seedling establishment (Balke *et al.*, 2011; Hu *et al.*, 2015). Low shear strengths found at the stations where seedlings were absent versus high seedling abundances at locations with high sediment shear strengths, seem to be in accordance with these studies. The fact that shear strength of mud is closely related to water content complies with Fagherazzi et al. (Fagherazzi *et al.*, 2017), who found that the erodability of mud-flats significantly decreases with increasing evaporation.

Our results furthermore suggest that decreasing submergence time in aquaculture ponds also favours seedling presence in abandoned aquaculture ponds, as seedlings were never found at low-lying stations and only occurred at locations with an inundation period of less than 40% of the day. These findings are in accordance with the vegetation classification per hydrological class from Watson (1928), and modified to inundation time per day by Van Loon et al. (2016), in which *Avicennia* species occur at inundation durations of 400-800 minutes per day (or 28-56% per day). Although mature mangroves are well adapted to flooding, seedlings are vulnerable to prolonged inundation times and have shown increased mortality due to flooding in various species (Krauss *et al.*, 2008). Increasing bed level elevations in abandoned aquaculture ponds could therefore both increase low inundation habitat surface area for seedlings and improve soil stability.

Overall, present findings suggest that seedling establishment in abandoned aquaculture ponds could be enhanced both by enhancing soil stability and decreasing inundation stress through bed level accretion. Providing there is enough sediment available in the water column, both aspects may be increased through improvement of hydrological connectivity, as tidal flooding is needed to deliver sediment while good drainage is needed to obtain bed-level stability. Hydrological connectivity can for instance be improved through breaching of the sea-ward pond bund, or digging drainage canals after a pond has been abandoned (Figure 2-7). These measures have indeed been proven effective for mangrove restoration in abandoned aquaculture ponds, and have as additional advantage that they also improve propagule distribution in areas where that is a limiting factor (Lewis, 2005; Van Loon *et al.*, 2016; Proisy *et al.*, 2018).

2.5.3 Mangrove expansion and mangrove retreat along the coast

We hypothesized that once a mangrove stand has developed, e.g. in an abandoned aquaculture pond, further seaward expansion would be mainly driven by either (2a) the presence of sheltering features on the foreshore that attenuate waves or (2b) by sediment consolidation, with poor consolidation reducing the expansion. In line with hypothesis 2a, our results show that mangrove retreat along the coast is indeed found at locations with minimal shelter from wind induced waves. That is, areas with mangrove retreat had a minimal mud-flat width, as quantified from Sentinel-2 satellite images, and field observations showed that transects with mangrove retreat generally had steep slopes (Figure 2-6). In contrast, mangrove expansion by young mangrove trees that could be observed from Sentinel-2 images was found behind wide mud-flats and in areas with a level to negative slope (Figure 2-6). A negative slope measured along expanding mangrove transects, indicates that the young mangrove trees were literally sheltered by mud-flats with a higher surface elevation seaward of the mangrove border. Stations that did not show mangrove expansion based on NDVI values, but did show seedling recruitment in the field were found at places with low to level slopes from sea to land. This could be explained by the fact that natural seedling settlement can only take place if a window of opportunity occurs (Balke et al. 2011, 2015). A Window of Opportunity requires both a high area of the mudflat, where an inundation-free period may occur following propagule stranding, and a sheltered area so that a wave-free period enables emerging seedlings not to be dislodged (Balke *et al.*, 2011, 2015).

The importance of foreshore morphology as a driver of vegetation dynamics is in accordance with findings of Winterwerp et al. (2013), Hu et al. (2015) and Bouma et al. (2016). Winterwerp et al. (2013) observed that in the mangroves of Surinam, concave (hollow) profiles concentrate wave energy, thereby causing a positive erosional feedback loop. In contrast, convex profiles, created by migrating mud banks from the Amazon (Anthony *et al.*, 2010), slowly dampen the waves allowing for sediment deposition inside the mangroves and thus for a positive accretion loop (Winterwerp *et al.*, 2013). Hu et al. (2015) modelled these processes to predict the window of opportunity for salt marsh expansion in the Netherlands with a model in which hydrodynamic forcing was incorporated on top of inundation time. Both slope and bathymetry profile had significant influence on salt marsh width (Hu *et al.*, 2015). Slope itself is an important driver of salt marsh width, as different slopes inherently create different amounts of settlement space at a certain tide level. However, mudflat profile along transects with the same slope proved to impose large differences on salt marsh width as well, predicting an increase of 300% salt marsh width along a transect sheltered by a convex mudflat compared to a salt marsh sheltered by a concave mudflat with the same 1/200 slope (Hu *et al.*, 2015).

In contrast to the expected outcome of hypothesis 2a, our second hypothesis (2b) “sediment stability causes seaward mangrove expansion through enhanced seedling stability” was not confirmed by our shear strength measurements. Sediment stability has been reported as an important limit to sea-ward mangrove expansion (Balke et al. 2013a, 2013b, 2015). This usually goes hand in hand with wave exposure: the more hydrodynamic forcing, the lower the sediment stability, and therefore the lower the seedling stability. Sediment stability was observed to be directly related to foreshore width (Bouma et al. (2016). In all these studies, the sediment stability was however measured as bed-level changes over time. In contrast, we measured sediment strength using a shear vane. This method is widely used (Serota and Jangle, 1972; Hauton and Paterson, 2003; Khan and Kostaschuk, 2011; Chen, Thompson and Collins, 2012; Winterwerp *et al.*, 2012), as shear-vane observations typically correlate well with physically measured critical erosion threshold (Grabowski, 2014). However, this indirect measurement, makes that our observations cannot be directly compared to studies where the sediment stability was measured in terms of bed-level change over time. Nevertheless, it is surprising that we found high shear strengths to be positively correlated with retreating mangroves (Figure 2-6). Shear strength along the coast seems to have been driven by grainsize rather than consolidation, as the variation in grainsize along the coast was much more pronounced than was the case in the aquaculture ponds (Table 21). In our retreating mangrove stands, the steep foreshore can have resulted in breaking waves on the mangrove border, which can cause resuspension of small sediment particles and leaving sandier sediment (with a higher shear strength) behind. Alternatively, previous erosion may have exposed a deeper stronger soil layer. Most importantly however, although experimental studies have shown that the absence of bed-level dynamics (i.e. soil stability) is good for seedling establishment (both for mangroves and salt marshes (Balke, Bouma, *et al.*, 2013; Bouma *et al.*, 2016)); at our field sites foreshore morphology appears to be the most important driver of seaward expansion.

2.5.4 Landward versus seaward mangrove expansion for green-belt creation

The difference between abandoned aquaculture ponds and coastal mangrove restoration sites in importance of shear strength for seedling stability has not been demonstrated before. In aquaculture ponds, we found that stability related soil-parameters like shear strength, besides submergence time, mostly seem to determine where seedlings grow (Figure 2-5). Contrastingly, in more exposed conditions along the seaside, high shear strength was correlated with mangrove retreat (Figure 2-6). These results show that increasing soil stability itself through bed level increase and improved drainage (as the results from the aquaculture ponds alone suggested), does not necessarily result in more seedling establishment. Along the coastline, management efforts should instead focus on creating and conserving wave sheltering features, such as sand banks, mudflats or artificial wave attenuators (Figure 2-7). Understanding such main driving processes of the ecosystem is important because it has implications for management. The patchy mangrove fringes and states that we found in our research area are common for many former mangrove coasts that have been converted to aquaculture ponds, such as e.g. in Vietnam (Van Cuong et al., 2015) and in Thailand (Winterwerp et al., 2013). In addition, the aquaculture ponds in our study area are very common all throughout Asia in terms of hydrological connectivity and size. Of course it cannot be excluded that in some regions soil chemistry might be a more important stressor for mangrove recruitment, e.g., when aquaculture practices have been industrialized (Stevenson, 1997) or herbicides like agent orange have been used in large quantities (Hong and San, 1993). However, when such very specific historical local land use effects are absent, present findings offer insight in the main drivers that may be manipulated for mangrove restoration projects along many Asian coastlines.

2.6 Conclusions and management implications

Our findings suggest that mangrove recruitment can potentially be initiated in old aquaculture ponds if: (1) The bed level of an old pond exceeds mean water level so that inundation times are below 40% per day and (2), sediment stability is sufficient.

Both beneficial conditions for mangrove establishment in aquaculture ponds could be improved through promotion of pond drainage after abandonment, so that sediment accretion and consolidation can take place (Figure 2-7). For management purposes and greenbelt restoration in aquaculture areas, it is therefore important to keep monitoring if the bed level is increasing after breaching of the pond bunds until the desired bed level is reached, and investigate the cause of insufficient sedimentation if the bed-level does not increase after breaching of pond bunds.

With regard to maintenance of newly established mangrove stands for greenbelt restoration, our findings suggest that expansion and retreat is primarily related to the shelter provided by the foreshore. Convex mudflat profiles seem to promote mangrove expansion and concave profiles without sheltering mudflats seem to induce mangrove retreat (Figure 2-7). To maintain restored mangrove stands along former aquaculture coasts, it is therefore important to monitor the state of the foreshore and understand the processes that cause convex and concave shore profiles.

2.7 Acknowledgements

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2.8 Appendices

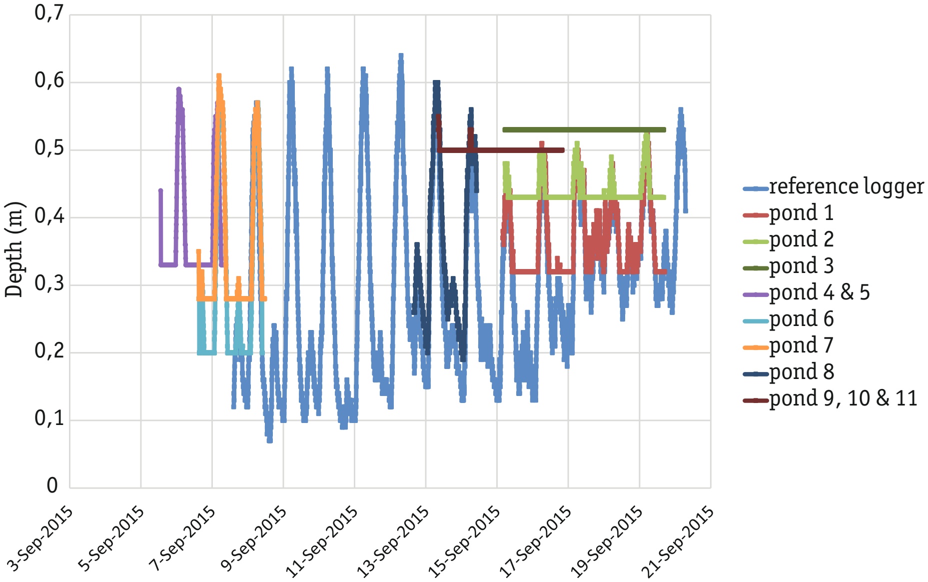

Figure A2-1: *Tidal inundation measured with pressure sensors at 11 aquaculture ponds compared to a subtidal reference station in open water (reference logger). Aquaculture ponds with ID number 4 and 5 and ponds 9, 10 and 11 had the same water level logger.*

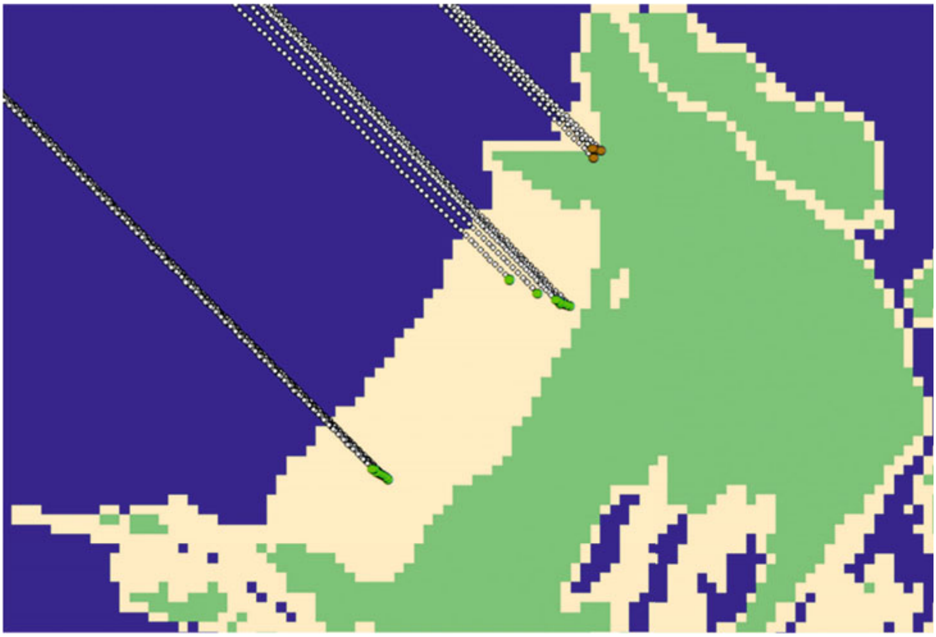

Figure A2-2: *Example detail of classified raster image based on Sentinel 2 NDVI data, with vegetation (green), exposed sediment (beige) and water (blue) (see main text for classification). Bearing lines (white dots) are indicated for all field-stations along two field transects with mangrove expansion as observed in the field (seedling recruitment on the mudflat) as green dots, and one transect with mangrove retreat as observed in the field (mature mangrove retreat) as brown dots.*

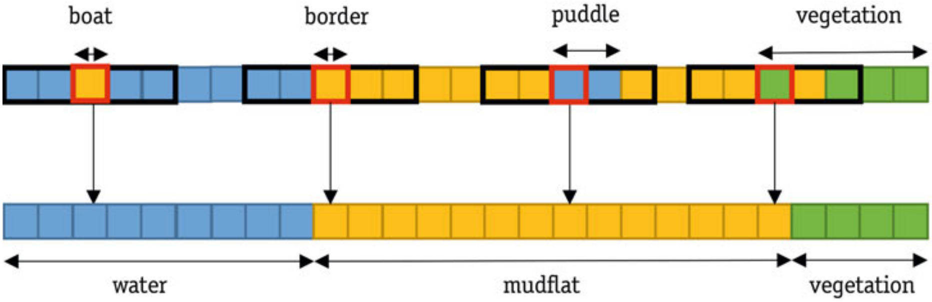

Figure A2-3: *Smoothening process of mudflat and water bodies in cells of bearing lines to exclude false positives in the raster classification such as boats on the water and puddles on a mudflat. Classification of single and double cells are transformed to the value found in the majority of a group of five cells: the cell of interest and two neighbouring cells in both directions.*

Natural foreshore types and their effect on mangrove edge dynamics

CHAPTER 3

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Can cheniers protect mangroves along eroding coastlines? –
The effect of contrasting foreshore types on mangrove stability.

3.1 Abstract

Mangrove forests are increasingly valued as wave-attenuating buffers in coastal flood defence strategies. However, as mangroves are vulnerable to wave-induced erosion, this raises the question, how to maintain these protectors? To address this question, we investigate how mangrove dynamics in a microtidal system can be related to different types of foreshores. We used remote sensing to investigate mangrove fringe stability over multiple years in relation to intertidal mudflat width (i.e., emerged at low tide) and the presence and stability of cheniers, which are sand bodies on top of muddy foreshores that are characteristic for eroding coastlines. In addition, we investigated local and short-term foreshore effects by measuring wave propagation across two cross-shore transects, one with a mudflat and chenier and one with a deeper tidal flat foreshore. The satellite images (Sentinel-2) revealed that mangrove dynamics over multiple years and seasons were related to chenier presence and stability. Without a chenier, a mudflat width of 110 m (95%CI : 76 – 183 m) was required to make mangrove expansion more likely than mangrove retreat. When a stable chenier was present offshore for two years or more, a mudflat width of only 16 m (95%CI: 0 - 43 m) was enough to flip chances in favour of mangrove expansion. However, mangrove expansion remained heavily influenced by seasonal changes, and was highly event driven, succeeding only once in several years. Finally, although mudflat width was a direct driver of mangrove expansion, and could be targeted as such in coastal management, our field measurements demonstrated that cheniers also have an indirect effect on mangrove expansion. These sand banks significantly reduce wave height offshore, thereby likely creating favourable conditions for mudflat accretion landward, and thus mangrove habitat expansion. This makes stabilization - and possibly also the temporary creation - of cheniers an interesting target for mangrove conservation and restoration.

3.2 Introduction

Mangrove ecosystems have been increasingly valued for their ecosystem services in the past few decades (Barbier *et al.*, 2011). Besides traditionally valued services such as viable fisheries, nurseries and water filtering capacity, the use of mangroves for coastal protection has also received attention (Yoshishiro Mazda *et al.*, 1997; Temmerman *et al.*, 2013). Mangroves attenuate waves with their dense tree tissues, such as extensive aerial root networks and canopy (Quartel *et al.*, 2007; Bao, 2011; Horstman *et al.*, 2014). Their complex root and branch structures reduce the wave velocity and can decrease wave height by 50% with every additional 100 meters of forest (Mazda *et al.*, 2006). The use of mangrove greenbelts for wave impact reduction is therefore often discussed in literature and implemented in coastal zone management (Othman, 1994; Duarte *et al.*, 2013; Spalding *et al.*, 2014; Narayan *et al.*, 2016). However, mangrove vegetation itself is also vulnerable to high wave energy and does not typically occur along exposed coastlines (Chapman, 1976). This "vulnerable protectors" paradox can be easily overlooked in management discussions debating the ideal width needed to obtain the desired amount of wave reduction on the landward edge of the mangrove forest. As a result, there may be clear specifications on the width of a protective mangrove greenbelt in various countries, but it is not always clear if and how such mangrove width can be sustained. For instance, Indonesia prescribes a mangrove green belt for coastal protection to be 130 m times the annual average tidal range (Presidential Decree (Kepres) No. 32/1990), whereas the Philippines uses a minimum width between 50 m and 100 m as guideline for coastal mangroves (R.A. 8550, P.D. 705, P.D. 953). Having such clear restrictions on the required wave-attenuating width of a protective greenbelt, requires tools to manage the mangrove forest width, especially under physically hostile conditions. Only by having an in-depth understanding of the environmental conditions that spark mangrove forest retreat and forest regeneration can we develop the means by which to achieve sustainable and effective forest widths.

Mangroves need an episodically occurring period of calm conditions to establish, a so-called Window of Opportunity (Balke et al. 2011, 2014). On a small scale, favorable conditions for natural mangrove establishment are nowadays well understood: mangrove seedlings need a window of opportunity in the form of an inundation-, wave- and erosion-free period to strand, root and anchor themselves to survive the first life stages (Balke *et al.*, 2011). On a larger scale, such calm-and wave free conditions can frequently be found in relatively sheltered areas such as lagoons and estuaries. However, at seaward facing sites, such calm conditions will only occur rarely. That is, the presence of such episodic calm conditions may be expected to be driven by the seasons in combination with the foreshore morphology (van Bijsterveldt *et al.*, 2020). As such, dynamic foreshore structures such as mud-banks, intertidal mudflats with or without chenier-sand-banks may be expected to play an important role in creating windows of opportunity for mangrove establishment. This is exemplified along the coastline of the Guianas, where the mangrove dynamics are dominated by fluid mud banks that originate from the Amazon river, and migrate west-ward along the coast (Augustinus, 1978). Mangroves extend seaward when sheltered by wave-dissipating intertidal mud banks and erode during exposed inter-bank stages (Anthony *et al.*, 2010). Along the mangrove-mud coast of north Java, similar patterns of mangrove recruitment and mangrove die-back can be found in a relatively patchy and young mangrove forest (van Bijsterveldt *et al.*, 2020). Here however, mud-banks are absent and the foreshore seems to be more dominated by intertidal mudflats with, in some cases, cheniers.

Cheniers, bodies of sand sitting on top of intertidal mudflats, can potentially create shelter for mangrove recruitment along coastlines. However, cheniers are typically described as features of eroding mangrove coasts (Anthony *et al.*, 2010). Sand is supplied in small amounts by rivers (Prost, 1989), but is only formed into cheniers when enough wave energy is present to rework the sediment (Augustinus, 1978). Chenier formation has been observed at locations such as the Red River Delta, Vietnam (Van Maren, 2005), the Mississippi Delta (Russell and Howe, 1935; McBride, Taylor and Byrnes, 2007), China (Liu Cangzi and Walker, 1989), West-Africa (Anthony, 1989), North-Java (van Bijsterveldt, 2015; Tas, van Maren and Reniers, 2020), Australia (Woodroffe and Grime, 1999),and at the Suriname-Guyana coastline (Augustinus, 1989; Anthony *et al.*, 2019). Along the Suriname coast, the wave-conditions driving chenier formation are typically found during the erosive stages of mud-bank migration, when mangroves are also eroded by the waves (Anthony *et al.*, 2010). Satellite images of the mangrove-mud coasts of Java suggest that cheniers might also be present during periods where parts of the mangroves expand, suggesting that the cheniers may take on a sheltering role, enabling mangrove expansion. However, intertidal mudflats also reduce waves at the coastline, and if they are of sufficient width, they may also provide the physical requirements for mangrove establishment (van Bijsterveldt *et al.*, 2020). In this study, we aim to investigate in depth how foreshore characteristics such as intertidal mudflat width and the presence of cheniers relate to mangrove dynamics. We investigate this along the coastline of Demak, Central Java, Indonesia on two temporal and spatial scales (Figure 3-1): (1) At the scale of the coastal system (i.e. in the order of 10s of kilometers) and yearly timescales, we used satellite-derived data in a Geographical Information System (GIS). (2) On a local and short-term scale (i.e. the order of 100s of meters and a period of days), we used field derived data.

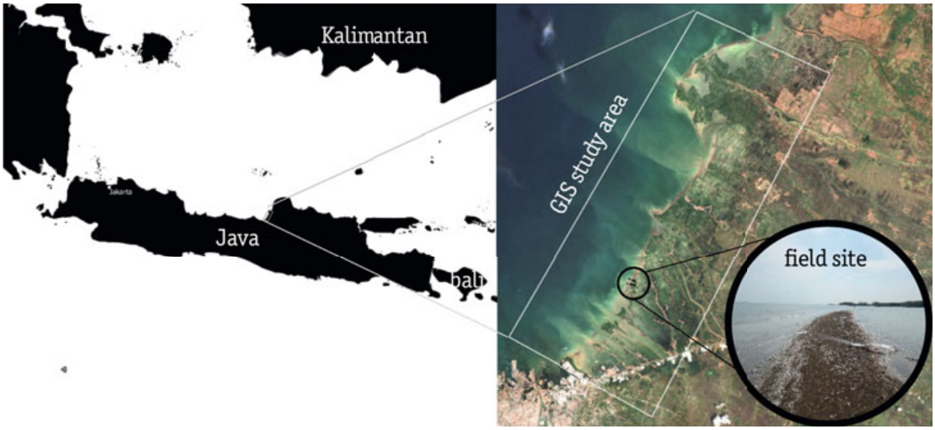


Figure 3-1: *The coastline of Demak district, on the Indonesian island of Java. The two focus areas of this study are indicated by (1) the white box, indicating the focus area of the GIS study in which we studied large scale and long-term chenier effects, and (2) the black circle, pinpointing the location of the two cross-shore transects in which we studied the small scale and short-term effects of cheniers. The circle shaped picture features a chenier in one of the transects, where small waves arrive on the seaward side (left) and distant mangroves are visible on the landward side (right).*

3.3 Methods

3.3.1 Field study: short-term cross-shore wave transformation with and without chenier

Site description

The coastline of Demak, North Java (Indonesia), is bordered by the city of Semarang in the South, and the Wulan River in the North. Demak experiences a microtidal range of 1 m and a mixed, mainly diurnal, tide (Tas et al., 2020). The local wave conditions are mild during most of the year, except during the NW monsoon between November and March, where significant offshore wave heights reach up to 2 m (*Figure A 3-6* (Van Domburg, 2018)). The coastal area is mostly formed by fine muddy sediment, except for the presence of cheniers along the coast. To investigate the short-term sheltering effect of cheniers on a local scale, we measured wave transformation and erosion in two cross-shore transects that were installed in Demak, Java, Indonesia: one transect with a chenier, and one transect without a chenier (Figure 3-2a). The location for the transects was selected in such a way that the hydrodynamic boundary conditions of the two locations was as similar as possible except for the presence of a chenier. Therefore, both transects started at a water depth of approximately 1 m with respect to MSL and were spaced 400 meters apart along the coastline. The first transect started 260 meters offshore from a chenier (chenier transect) and the second transect started at a similar depth and distance from the shoreline, but without a chenier (exposed transect) (Figure 3-2b).

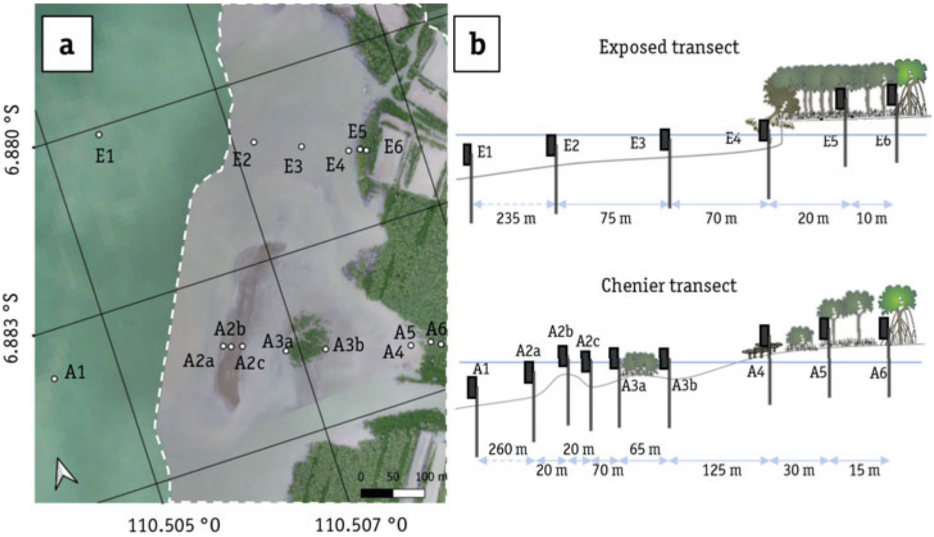


Figure 3-2: *Two cross-shore transects in the field with and without a chenier.* ***a.*** *Wave logger deployment locations are indicated in a drone image (indicated with white dashed outline, a Sentinel-2 image from two weeks later is used in the background) of the transect area in November 2017. The exposed transect did not feature a chenier (E1-E4) and showed mangrove die-back E4-E5. The chenier transect contained a chenier (A2a-A2c), a mangrove stand on an old chenier (A3a-A3b) and a mudflat with seaward expanding mangroves (A4-A6).* ***b.*** *Schematized bathymetry and instrument deployment along the exposed and chenier transects.*

The chenier transect featured two cheniers; one bare sand bank that consisted mostly of fine sand, ranging from 63.5 to 500 µm in grainsize (as measured from sediment samples of the top 3 cm, freeze-dried and analyzed using a Malvern Mastersizer 2000), and one vegetated chenier that consisted of a thin layer of sand on top of mud (*Figure A 3-1*). In the field, both cheniers were easy to walk on, although the layer of sand appeared thin. Jumping on top of the sediment caused the sand body to quiver, and when walking towards the landward side of each chenier, the sand became so thin that one could sink through the sand into the underlying mud. The exposed transect did not have an emerged chenier, although the foreshore stations of the exposed transect showed grainsize distributions with much more sand mixed through the sediment than the stations seaward of the chenier in the chenier transect. This could indicate that the sediment at these stations were the remnants of an old chenier or the start of a new chenier forming in the exposed transect.

The most landward station of the two transects was situated inside the mangrove forest. The sediment composition inside the mangrove stands of the two transects was very similar with a high silt content (>88%) at all sites (*Figure A 3-1*), although the forest stations in the exposed transect also contained fine sand (125-250 µm, 2 ± 0.2 %) and very fine sand (62.5-125 µm, 7.6 ± 1.2 %), indicating that the mangroves at the fringe of the exposed transect were subjected to more wave energy than the mangroves in the chenier transect.

Data collection

Wave loggers (OSSI Wave loggers and NIOZ, MARK III SED pressure sensors) were deployed along the two transects at equal distances from the mangrove border in both transects perpendicular to the coast in a north-west direction (Figure 3-2b). Additional wave loggers were placed across both the bare and the vegetated chenier in the chenier transect, in order to measure wave propagation across these sheltering landscape features as well. Wave data were collected continuously with a frequency of 10 Hz during 8 days in the wet season of 2017-2018 (November-March). This relatively short timeframe did, however, include a storm event that caused extensive flooding of the whole area (Afifah and Hizbaron, 2020). The average significant wave height was therefore determined for both the entire 8-day period, as well as for the storm on the 1st of December, between 00:00 and 05:00, giving insight into the impact of cheniers on average wet-season waves and on extreme storm waves.

Forest parameters were recorded at the landward edge of each transect to typify the forest. We counted the number of seedlings (height < 0.5 m) per species and recorded the diameter at breast height (DBH) of individuals that were larger than 1 m. Individuals between 0.5 m and 1 m in height were recorded as saplings. These species counts and DBH measurements were conducted in circular plots at the most landward station in the exposed and chenier transects after the wet season of 2017-2018. Plot size differed between the exposed and chenier transect (78.5 m^2 and 38.5 m^2 respectively) due to difficulty to move around in the muddy sections of the chenier transect and the risk of trampling seedlings. Forest parameters were therefore corrected to counts per hectare (ha) to compare the two transects.

Data analysis

Processing hydrodynamic data

The pressure measurements from the wave loggers were corrected for the atmospheric pressure, using the air pressure data collected by a wave logger installed in a nearby tree. The offset of each instrument was determined by in-situ calibration: instruments were placed at one location, and water depth was measured manually at different moments of the tidal cycle for validation. After offset correction, the pressure measurements were transformed into water depth assuming a water density of ρ=1024 $kg^{-1} m^3$ and a gravitational acceleration of g=9.8 $m^{-1} s^2$. The mean water levels were derived from the pressure signal, and the detrended pressure signal was then used to calculate the wave density spectra over 19.5-minute intervals. The significant wave height (H_m0), and peak period (T_p) were derived from the spectra of each interval. In order to compare the wave conditions at the different stations, intervals where all sensors were submerged were selected. The wave heights during these submergence periods were then averaged over the duration of the storm (1st of December) and over the full 8-day measurement period.

3.3.2 GIS study: the relation between intertidal foreshore features and mangrove dynamics

Data collection

To study the effects of cheniers and mudflats on mangroves over multiple years, we performed a GIS study on the coastline of Demak. Sentinel-2 satellite images were selected to study the effect of cheniers because 10 meter is the highest resolution of freely available satellite images and a sufficient resolution for the detection of cheniers and changes in mangrove cover. All available Sentinel-2 satellite images during a 4-year period were therefore assessed for cloud cover in the research area and tidal level. Ultimately, only eight images could be selected based on cloud cover (<10%), low tide conditions and season (one post-dry-season and one post-wet season for each year). The exact tidal level at the moment of satellite image acquisition was obtained from a tidal harmonic analysis of the tide station of Semarang. To detect seasonal changes in mangrove and mudflat dynamics, images were selected based on acquisition dates before the stormy wet season (Dec-Feb) and before the relatively calm season of the year: the dry season (Jun-Aug) and transitional seasons (Mar-May and Sep-Nov).

Image classification

Satellite images were atmospherically corrected using Sen2Cor software. Clouds and cloud shadows were removed from the images by masking QSC values produced by the Sen2Cor software. Then, a normalized difference vegetation index (NDVI) band was computed for all selected Sentinel-2 images, and pixels for all bands outside the study area and the zone of interest were masked. The study area was restricted to the region of the tidal flat beginning from the coastal mangrove forest as it appeared in October 2015, reaching 2 km out to sea in a northwest direction from the mangrove edge (Figure 3-3a). Satellite images were then subjected to two steps of unsupervised classification, to cluster cells into four relevant classes in the study area: water, mud, sand and vegetation.

These four classes are easy to distinguish manually from the satellite imagery (Figure 3-3b), but the differences in background reflection between the different dates made it difficult to use fixed thresholds of a certain band to distinguish these classes consistently between the dates. For instance, one spot in the middle of a mangrove stand can have an NDVI of 0.3 in the satellite image of one date and 0.7 on a different date, with mudflats having an NDVI of up to 0.5. Using a threshold of NDVI = 0.3 would therefore overestimate the mangrove cover on the second date, misclassifying sections of the mudflat as mangroves. To avoid this problem, we used an unsupervised classification tool (ArcGIS pro, Iso Cluster Unsupervised Classification Tool), which made it possible to automate the classification process for multiple images.

The tool uses a combination of an iterative self-organising (iso) algorithm (migrating means clustering) and a multivariate analysis of the input satellite bands to classify the raster cells based on their statistical similarity (maximum likelihood classification)(ESRI, 2020). When only one band was fed into the tool, the statistical method clustered the image in a way that was very similar to a clustering based on “natural jenks” in the frequency distribution. These tool-properties were used to classify the images in two steps (*Figure A 3-2*). For the first step, the masked Sentinel-2 images were clustered into 5 groups using only the NDVI band as input for the unsupervised classification tool. One cluster was then classified as vegetation, two clusters were classified as exposed sediment (either wet or dry bed) and two clusters as water (either with moderate or high levels of suspended sediment) (*Figure A 3-3*). The exposed sediment group was subsequently used in the second step of the classification. The exposed sediment layer per Sentinel-2 image was used as a mask for all 10 m and 20 m resolution bands per satellite image. These masked bands were then fed into the unsupervised classification tool, and subsequently clustered into a predefined number of sediment classes. Trial and error runs with various numbers of classes revealed that clustering the masked images into 8 groups consistently grouped “sand” into one class for all Sentinel-2 images (*Figure A 3-4*). The signal of the “sand” cluster consistently showed a relatively high surface reflectance in the short wave infra-red (SWIR) bands in combination with a low to medium reflectance in the visible and near infrared bands, whereas the 7 mud clusters all showed a strong drop in surface reflectance between the near infrared and SWIR bands. This difference in SWIR reflectance between sand and mud is probably caused by the efficient drainage of sand in comparison to mud (Small *et al.*, 2009). However, because we could not validate the different drainage levels of mud in the field, we added only the classes “sand” and “mud” to the first image classification resulting in a raster with classes: water, mud, sand and vegetation for every Sentinel image of interest (*Figure A 3-5*).

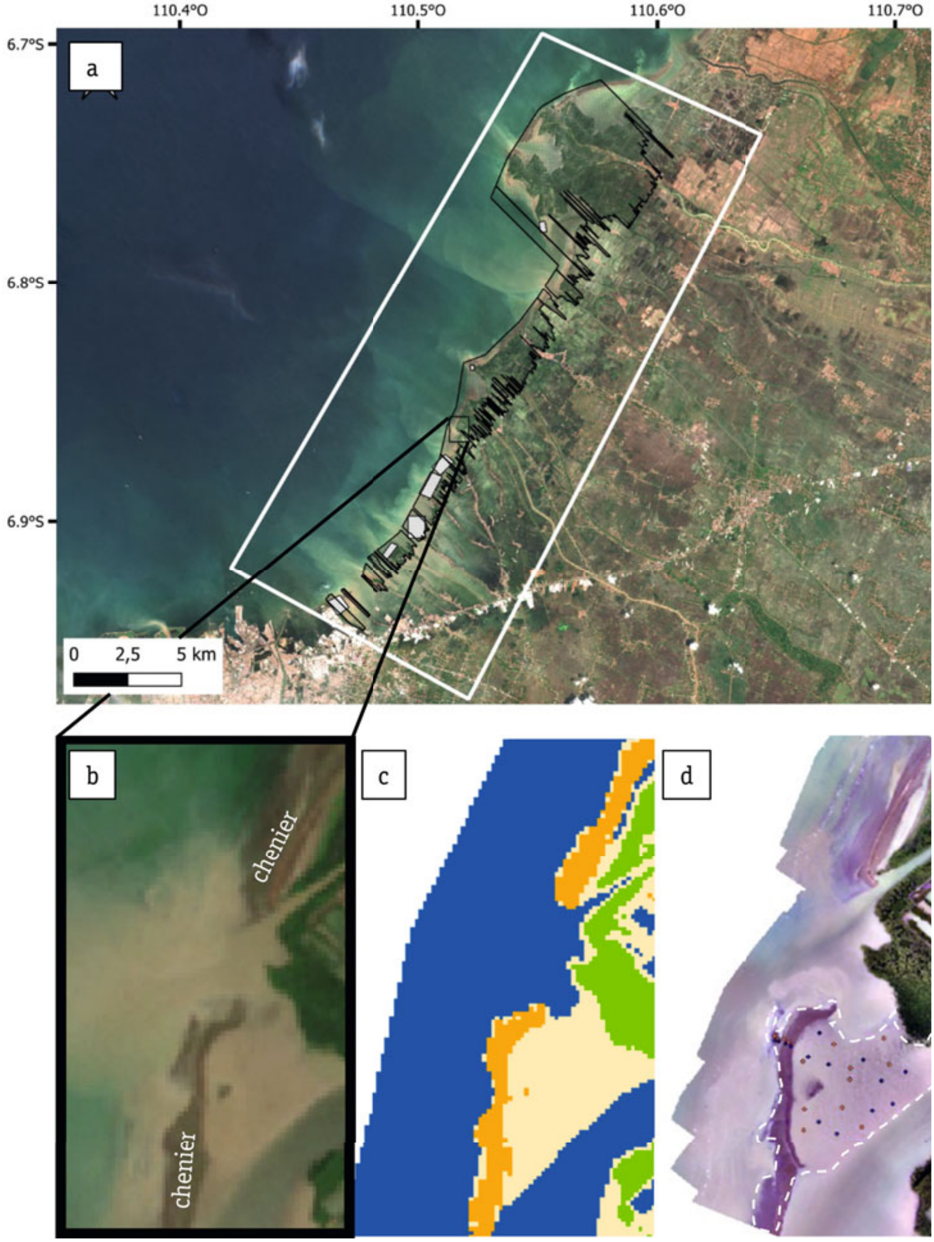


Figure 3-3: *Satellite image classification and validation to study mangrove response to chenier presence and mudflat width.* ***a.*** *The area of interest of the large scale and multi-temporal GIS study consisted of the area starting at every mangrove pixel at baseline (07-10-2015) until 2 km offshore (indicated with the black lined polygons).* ***b.*** *Close-up of a mangrove site featuring a mudflat and chenier on the RGB composite of one of the Sentinel-2 images of interest (15-11-2018, 02:59 UTC, water level: -30 cm rel. to MSL).* ***c.*** *Classified Sentinel-2 image showing the same close-up area with mangrove cover (green), mudflat (beige), sand (yellow) and water (blue).* ***d.*** *Drone image (6-11-*

2018, 08:47 UTC, water level: -15 cm rel. to MSL) of the field validation at this close-up site, with groundtruthing points on the chenier and mudflat and the outline of the emerged area during the drone flight. Another 7 validation sites were visited with the drone (indicated with the white filled rectangles in a).

Validation of GIS classification

To validate the unsupervised classification we visited 8 sites within the study area in October and November 2018 with a drone to collect a total of 171 ground control points with a dGPS and high resolution imagery of the cheniers, mudflats and vegetation at low tide (Figure 3-3d). Ground control points were classified as mud or sand in the field. These points were then used to validate the sediment type as classified in the Sentinel image of November 2018. We determined the percentage of field stations that were classified correctly as sand and mud in GIS (the producer's accuracy), and we determined the percentage of test pixels from Sentinel-2 that were classified correctly based on the sediment type in the field (the user's accuracy *Table 31*). The user's accuracy showed that 94% of the ground control points that were classified as mud from the Sentinel-2 image were indeed muddy in the field, and that 83% of the locations that were classified as sand were indeed sandy in the field. Similarly, 94% of the sandy field locations were also classified as sandy based on the Sentinel-2 image and 85% of the muddy sites were classified as mud (Producer's Accuracy). Overall, the accuracy of the classification was 91% (kappa=0.78, Lower 95%CI=0.67, Upper 95%CI=0.88).

Table 3-1: *Error matrix resulting from sediment classification of exposed intertidal sediment in a Sentinel-2 image (November 15, 2018) and the sediment type observed in the field at 171 ground control points.*

		Ground control data (known sediment types)			
		Mud	Sand	**Row total**	
Classification data (from GIS)	Mud	117	7	124	**User's accuracy** Mud = 117/124 = 94 %
	Sand	8	39	47	**User's accuracy** Sand = 39/47 = 83 %
	Column Total	125	46	171	
		Producer's Accuracy Mud = 117/125 = 94 %	**Producer's Accuracy** Sand = 39/46 = 85%		**Overall accuracy** = (117+39)/171 = 91%

Definition of explanatory and response variables from GIS

The classified Sentinel-2 images were used to quantify the effect of the absence or stability of cheniers on changes in mudflat cover and mangrove border along the dominant wind-direction during the monsoon season, which is north-west (MMAF, 2012). In order to obtain information along this direction, a total of 3,255 north-west bearing lines were drawn from every cell that contained mangroves at baseline in October 2015 (the first selected Sentinel-2 image available) in the whole project area from Semarang to the Wulan Delta (Figure 3-3a). Each bearing-line contained sampling points every 14.14 m, based on the diagonal width of the Seninel-2 raster cells. The feature classification was subsequently extracted at each sampling point from each date's classified raster with mangrove, mudflat, sand and water pixels. Bearing lines that contained clouds seaward of the mangrove border were excluded from further analysis.

Mangrove cover change between the acquisition dates was extracted from the classified images and used as the response variable. Mangrove cover change was categorized into one of three relevant response classes between two consecutive points in time: the classes being: "expanding", "stable", or "retreating" if the number of vegetation cells between the mangrove-sea border at t_n and t_{n+1} was respectively larger than -, equal to - , or less than zero.

To obtain chenier presence-absence data and mudflat data, the classified images were subjected to a smoothing algorithm according to van Bijsterveldt et al. (2020), which excluded small patches (10-20 m wide) of a certain category, such as ships (classified as sediment) in the water, or puddles of water on the mudflat. The smoothed classified Sentinel images were then used to extract shelter-related variables from the images such as, chenier presence and mud-flat width per bearing line for each of the selected acquisition dates. These characteristics were obtained by quantifying the number of cells of that class from the mangrove border in seaward direction along each bearing line and multiplying that number by the cell length 14.14 m.

Data analysis

Hypothesis testing for the effect of cheniers and mudflats on mangroves

To test the hypothesis that the presence of cheniers and mudflats drive mangrove border dynamics, we performed a linear regression separately on each of the three possible mangrove states (retreat, stable or expanding). For these models, we decided only to include wet-season data because the largest changes related to mangrove cover were expected during this season; the propagule dispersal peak inducing mangrove expansion occurs at the start of the wet season (van Bijsterveldt *et al.*, 2022), and the most impactful storms that could induce mangrove retreat occur during this season.

The response variable for each of the three models (mangrove expansion, stability and retreat) was the proportion of transects with that mangrove response occurring (e.g. mangrove retreat) for each unique combination of chenier stability and mean mudflat width. For example: mangrove retreat occurred in 33 out of 47 bearing lines without a chenier (0 years), and a mudflat width of 40 meters wide. Mean mudflat width and chenier stability during the study period were thus added to each of the three linear regressions as explanatory variables. Chenier stability was defined as the number of years that a chenier had been present in a bearing line (0, 1 or ≥2 years). Mean mudflat width over the 4 wet seasons was log transformed and binned, to obtain groups of transects of similar size (a similar number of transects per unique mudflat width), to account for the log-normal distribution of mudflat widths (there were many transects with a small mudflat width and fewer transects with large mudflats).

3.4 Results

3.4.1 Field study: short-term cross-shore wave transformation with and without chenier

Wave attenuation by cheniers

An offshore wavebuoy (12 km offshore in NW direction) revealed that waves during the field campaign arrived from a north-west direction (Figure A6), in line with each transect. The significant wave height (Hs) at the most seaward station of both transects was 0.5 ± 0.2 m on average during the field campaign, indicating that the boundary conditions during average wet season conditions of the two transects were comparable. The 8-day-averaged significant wave height (dashed line in Figure 3-4) then dropped below 0.26 ± 0.06 m in both transects between the first two stations, where the foreshore of both transects was still comparable in terms of sediment composition (*Figure A 3-1*) and profile (Figure 3-4). From there on, the wave height remained stable in the exposed transect, only showing a strong drop at the mangrove edge between station E4 (Hs: 0.24 ± 0.06 m) and station E5 (Hs: 0.15 ± 0.08 m), indicating that the waves break on the edge of the mangroves forest. This is further supported by *Figure A 3-8* (a), which shows a linear relationship between wave height and water depth at E5, characteristic of depth-limited wave breaking. In contrast, the waves in the chenier transect already showed a strong drop at the chenier stations 2a, 2b, and 2c (Hs: 0.26 ± 0.06, Hs: 0.26 ± 0.06, and Hs: 0.14 ± 0.06 resp.), indicating that the waves break on the chenier. All stations landwards from A2a thus display a linear relation between water levels and wave heights (*Figure A 3-8* b). This resulted in a significant wave height of 0.13 ± 0.06 m at the mangrove edge of the vegetated chenier (A3a) and waves of 0.11 ± 0.04 m at the edge of the main mangrove forest (A4) of the chenier transect under average wet season conditions. The significant wave height at both of these mangrove stations was significantly lower (*F* = 229.9 , df = 2, $p < 0.0001$) than the waves at the mangrove border in the exposed transect (E4, Hs: 0.24 ± 0.06). The full time series of wave heights and water levels at all stations can be seen in *Figure A 3-7.*

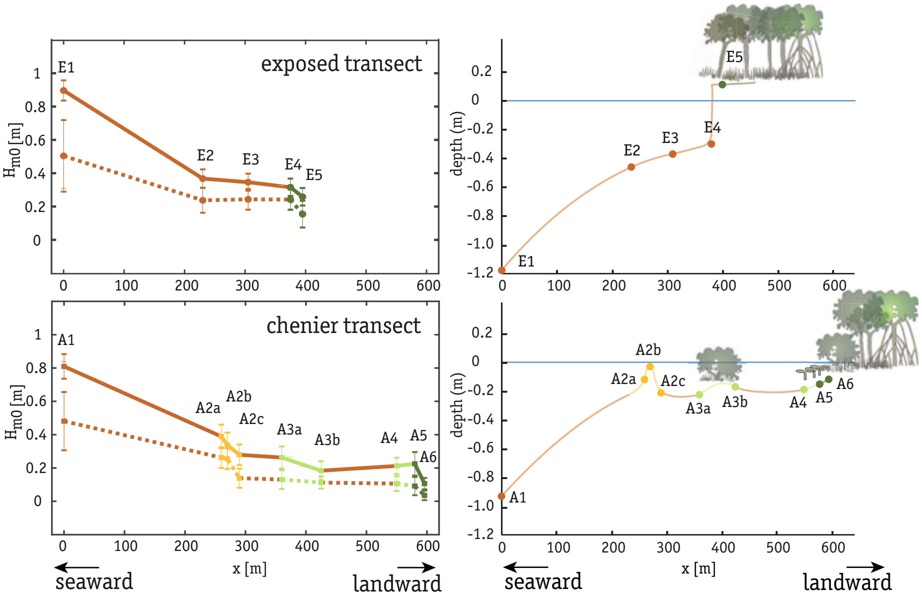


Figure 3-4: Average significant wave height at each of the stations across the exposed transect and the chenier transect during storm conditions on the 1st of December (solid lines) and on average during the 8 days measured (dashed lines). The location of each of the stations is indicated with colors relative to the colors of the schematized bathymetry profiles on the right. Depth was only measured at the stations and is displayed relative to mean water level during the campaign. The lines in between the stations are estimates of the profile contour.

Wave attenuation by cheniers under storm conditions

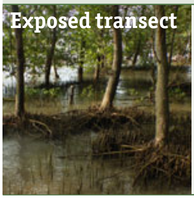
In addition to the average wet season conditions, we plotted the wave conditions measured on the first of December separately in Figure 3-4. On this day, the instruments detected a significant increase in water level and significant wave height. This signal was caused by cyclone Dahlia passing nearby, along the south coast of Java. The cyclone increased the water level 60 cm above mean sea level (Alferink, 2022), and the flooding that followed was reported by villagers to be the worst flooding in the last 30 years. During this storm, the chenier had a similar effect on the waves, decreasing the significant wave height by more than 10 cm (from 0.39 ± 0.07 m to 0.28 ± 0.06 m). As a consequence, the significant wave heights at the mangrove edge of the vegetated chenier (Hs A3a: 0.26 ± 0.07) and the edge of the main forest of the chenier transect (Hs A4: 0.21 ± 0.04) were both significantly lower (*F* = 12.94, df = 2, $p < 0.0001$)

than at the edge of the mangrove forest of the exposed transect (Hs E4: 0.32 ± 0.05). Further landward from the chenier, the wave height also decreased over the vegetated chenier (from 0.26 ± 0.07 m to 0.18 ± 0.06 m) and between the two stations inside the mangrove forest (from 0.23 ± 0.07 m to 0.11 ± 0.03 m). This illustrates how the canopy of the young, shrub-like, trees are able to cause further wave attenuation during the high water levels of a storm.

Forest characteristics behind cheniers

The forest characteristics at the most landward stations were very different for both transects (*Table 3-2*); the chenier transect had a high seedling density of the two common *Avicennia* species in the area (*Avicennia alba* and *Avicennia marina*), whereas no seedlings of these species were found in the exposed transect. In the exposed transect, 20% of the mature trees were dead, mostly occurring at the edge, indicating mangrove retreat as a result of erosion. Saplings of *Avicennia* were completely absent from both forest plots, indicating that seedlings of those species that had established before the wet season of 2017-2018 had not survived.

Table 3-2: *Forest parameters at the most landward stations of the exposed and chenier transects after the wet season in 2017-2018*

	Species	**Seedling Density (ha^{-1})**	**Sapling Density (ha^{-1})**	**Tree Density (ha^{-1})**	**Tree DBH (±SD) (cm)**	**Tree Mortality (% of dead trees)**
Chenier transect 	*A. marina*	42116	0	260	0.3 (0)	0
	R. mucronata	0	0	260	1.0 (0)	0
	A. alba	3120	0	1040	2.4 (2.1)	0
	Total	45236	0	1560		
Exposed transect 	*A. marina*	0	0	1274	10.2 (6.6)	20
	R. mucronata	0	382	1529	1.7 (0.8)	0
	A. alba	0	0	0		
	Total	0	382	2803		

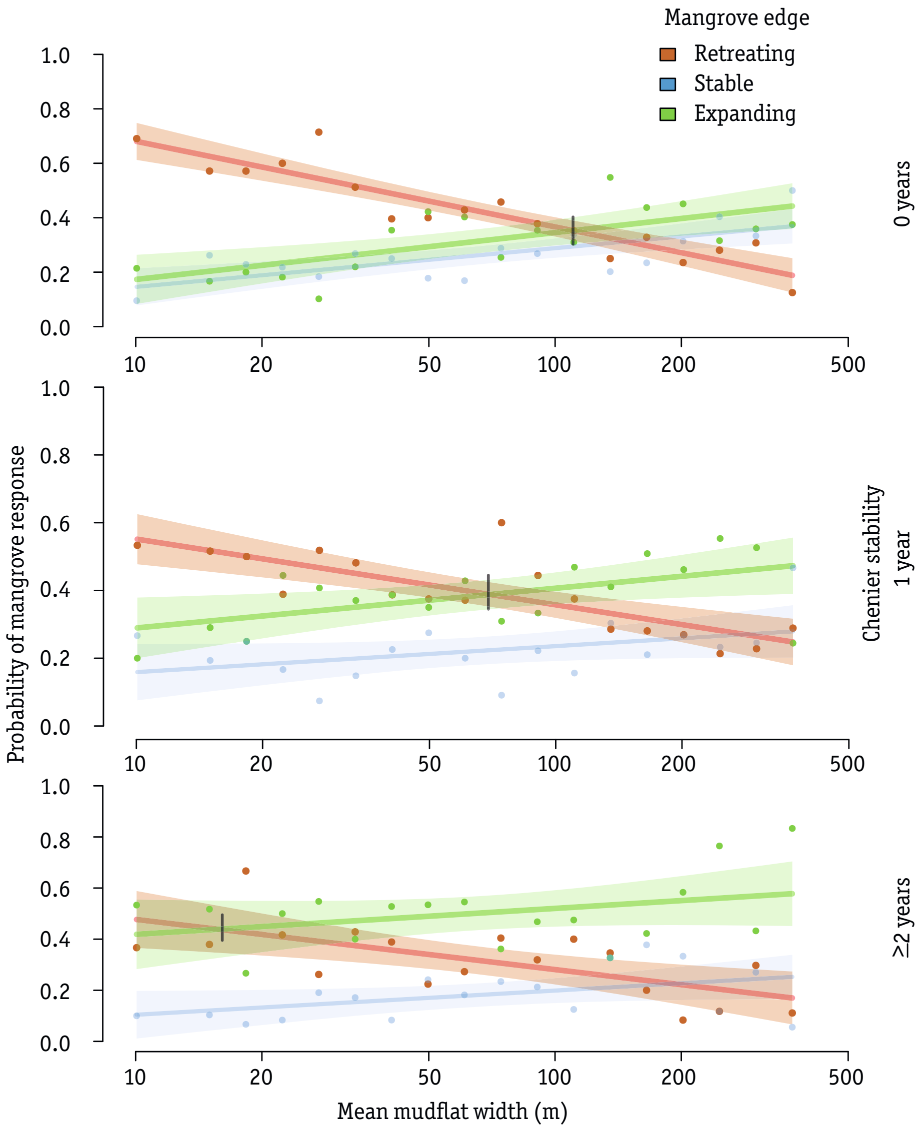


Figure 3-5: *Probability and 95% confidence intervals of mangrove response (retreat, stable or expansion) in relation to mean mudflat width (m, in bins) for various degrees of chenier stability (the number of years a chenier was present per transect) during the 4 year time frame of the study. The mean mudflat width required to make mangrove expansion more likely than mangrove retreat is indicated with a black vertical line in each panel.*

3.4.2 GIS study: the relation between intertidal foreshore features and mangrove dynamics

The multi-year net effect of cheniers and mudflats on mangrove dynamics

The probability of mangrove expansion on a larger scale and over multiple years in relation to the presence of cheniers and the width of mudflats was investigated using linear regression. Both mean mudflat width and chenier stability proved to have a significantly positive impact on mangrove expansion ($F = 33.1$, df = 2 & 58, $R^2 = 0.52$, $p < 0.001$). The probability of mangrove retreat decreased significantly with larger mudflat widths and more stable cheniers ($F = 77.6$, df = 2 & 58, $R^2 = 0.72$, $p < 0.001$). The proportion of stable mangrove fringes was small under all foreshore conditions, indicating that mangroves tend to be dynamic, switching between a state of expansion or retreat, although mudflat width did have a positive effect on mangrove stability ($F = 22.56$, df = 2 & 58, $R^2 = 0.42$, $p < 0.001$). Plotting the observed probabilities of each mangrove state shows that without a chenier and the smallest observed mudflat, the probability that mangroves retreated was much higher (70%) than the chance that they were stable (13%) or expanding (17%) (Figure 3-5). However, mangrove forest retreat clearly decreased with an increase in mudflat width, even in the absence of a chenier (chenier stability = 0 years). Without a chenier, mangroves were more likely to expand than retreat from a mudflat width of 110 meters onward (95% CI: 76 – 183 m). This tipping point between mangrove retreat and mangrove expansion occurred at smaller mudflat widths when a chenier was present offshore. When a chenier was stable for one year in front of the mangrove fringe, a mudflat width of only 70 m (95% CI: 35 – 145 m) was needed to flip the odds in favor of mangrove expansion. When a chenier had been present for two years or longer, this tipping point occurred at a mudflat width of 16 m (95% CI: 0 – 43 m). Therefore, the more stable a chenier, the larger the chances of mangrove expansion (Figure 3-5).

3.5 Discussion

In this study, we investigated the effect of both *i)* cheniers and *ii)* intertidal mudflat width on mangrove dynamics, using *i)* wave measurements at cross-shore transects in the field and *ii)* multi-year satellite data on mangrove dynamics. Our field data show that existing cheniers reduce the height of the waves arriving at the mangrove fringe, thereby creating a shelter for mangroves as long as the chenier is present. Our GIS data confirm that the temporary shelter created by cheniers increases the chances of net mangrove expansion and reduces the occurrence of mangrove retreat. In the absence cheniers, a much wider intertidal mudflat is required to facilitate mangrove expansion.

3.5.1 Local chenier effects: wave reduction and habitat creation

The fact that offshore cheniers reduce the wave height at the mangrove fringe is in itself not unexpected. In sandy systems, sand banks and barrier islands are well known to cause wave height reduction at the shoreline (Short, 2001). However, the mechanism of wave height reduction over sandy offshore features is different from wave height reduction over muddy foreshores. Muddy foreshores cause wave attenuation due to bottom friction caused by sediment resuspension and the absorbing effect of the liquid mud top layer (Sheremet and Stone, 2003), whereas the relatively steep and hard surface of sandy foreshores cause waves to break (Short, 2001; Wolf *et al.*, 2011). These two wave reducing processes are seemingly combined in the case of sandy cheniers atop of a muddy foreshore, where wave height is reduced over the muddy foreshore (from A1 to A2a and from E1 to E2 in Figure 3-4), before breaking on top of the sandy chenier (A2a-A2c, Figure 3-4). The difference between the storm conditions and average wet season furthermore show how the effect of the chenier is influenced by the water depth. When a chenier is fully emerged during low tide it acts as a barrier, and the water surface on the landward side of the chenier is completely still (e.g. Figure 3-1 Field site picture). When a chenier is submerged, for instance during average wet-season conditions (dashed lines in Figure 3-4), the chenier reduces the wave height by around 10 cm. The amount of wave reduction by the chenier (±10 cm) remained the same when the water level peaked during the storm of the 1st of December. However, due to the larger size of the incoming waves, the wave height reduction at the mangrove edge was relatively smaller, causing significantly larger waves to reach the mangrove edge during the storm than under average conditions. Nevertheless, these data show that even submerged cheniers have a clear sheltering effect on existing mangroves.

The relatively calm backwater that is created by cheniers affects both mudflats and mangroves. The low wave height that was measured directly landward of the chenier in this field study is favorable for deposition of small sediment particles. The high silt content (*Figure A 3-1*) and the soft quality of the mud that were observed landward of the chenier (Table 2 picture) indeed suggest that cheniers facilitate mudflat formation in the area that they shelter. Mudflats in their own right are known to have a protective (Bouma *et al.*, 2016; van Bijsterveldt *et al.*, 2020) and nursing role (Swales *et al.*, 2007) towards mangroves. Unfortunately, the size of intertidal areas has been declining on a global scale over the last 30 years as a result of, among others, coastal development, decreased sediment input, and increased drainage and compaction (Murray *et al.*, 2019). The few tropical sites that show an expansion of intertidal area, also display a seaward migration of mangroves (Murray *et al.*, 2019), illustrating the importance of a sizable intertidal area for mangrove development. Our GIS results showed that the likelihood of mangrove expansion indeed increased significantly with increasing size of intertidal mudflats, with mangrove expansion becoming more likely than mangrove die-back from an intertidal mudflat size of 110 m (95% CI: 76 – 183 m) onward in this microtidal system. In macrotidal systems the necessary intertidal mudflat width to support mangrove expansion might be larger, as deeper water at high tide allows for higher waves to reach the shoreline, though intertidal areas tend to be wider in such systems as well (Murray *et al.*, 2019). Nevertheless, one third of the world's tropical mangroves can be categorized as micro-tidal and sedimentary (Balke and Friess, 2016), like the coastline of Demak. Therefore, the tipping points in mangrove expansion in relation to mudflat width found in this study could potentially be helpful in management of other micro-tidal mangrove forests around the globe as well.

3.5.2 Large-scale and long-term chenier effects: the importance of a calm wet season

The data in this study revealed that the presence of a wide mudflat or the presence of a chenier can support net mangrove expansion over multiple years. However, this does not mean that the presence of a chenier or mudflat during a wet season necessarily results in mangrove expansion during that fruiting season. The latter requires also a Window of Opportunity to occur, consisting of an inundation-, wave-and erosion-free period to strand, root and anchor themselves, and thereby survive the first life stages (Balke *et al.*, 2011). In the field, the forest characteristics at the most landward edge of the chenier transect showed that there had been no window of opportunity for mangrove growth during the previous season, because saplings of pioneers (were completely absent) from the site. The absence of *Avicennia* saplings, while seedlings were abundant, suggests that the seedlings that had established at the start of the wet season (and would have grown into saplings before this field campaign) did not survive this particular storm season. This observation indicates that, while cheniers and mudflats reduce wave height and promote mudflat accretion, successful mangrove establishment in the seaward direction remains an event-driven process.

Satellite analyses support that the colonization of mangrove habitat occurs episodically and is therefore non-linear through time. Net mangrove expansion occurred primarily during only one wet season (2016-2017: *Figure A 3-5*). During this wet season, a window of opportunity probably occurred due to the remarkably low maximum daily wind speeds of ±10 m/s (as retrieved from the Ahmand Yani airport station in Semarang (http://dataonline.bmkg.go.id/data_iklim)). Thereby the dry-season-like wind speeds, though in onshore direction, coincided with the fruiting season of the common *Avicennia* species in the area, blowing the propagules towards the shore. The combination of available propagules and the presence of wide mudflats for establishment, followed by months of calm conditions were likely the cause of the positive mangrove cover change during that same season. This combination of favorable conditions resulted in a staircase-like appearance of the forest-canopy (e.g. Fig. 3d in van Bijsterveldt et al., 2020), caused by separated events of mangrove expansion interspaced by years of non-expansion. Non-linearity in seaward mangrove expansion is not uncommon, and has also described in the coastal system of the Guianas in South America, where migrating mud banks offer temporary shelter and habitat for mangrove expansion, interspaced with decades of non-shelter and non-expansion (e.g. Fig. 11 in Anthony et al., 2010). Similar periodic mangrove expansion, though by a different mechanism, has been observed in the Firth of Thames, New Zealand, where reduced wind and wave energy during the El Niño events of 1978-1981 and 1991-1995 resulted in two major seaward forest expansion events (Lovelock *et al.*, 2010). Our findings therefore illustrate how seaward mangrove expansion can be induced by a combination of temporal shelter and temporal calm conditions.

3.5.3 Management implications

The observation that cheniers create temporary shelter for mangroves from waves, especially when they are stable over longer periods of time, has implications for mangrove conservation and restoration. For example, despite their role in mangrove establishment, cheniers have been mined to use their sand for construction in Demak, which deprives the coastline from their erosion mitigation function. Sand mining should thus be strictly regulated to maximize mangrove colonization and mitigate retreat. Conversely, mangrove persistence and expansion could be favored if existing cheniers are stabilized or supplied with sand from a sustainable source. Although artificial sand nourishments have been used as wave breakers before (e.g. Hwung, Huang, & Hwang, 2010), little is known about sand nourishments on muddy substrate, presumably because sand is relatively rare along muddy coasts. The tipping points for mangrove retreat at specific mudflat widths also have useful, and perhaps more feasible, implications for management. Existing mudflats might be nourished to obtain a desirable width, but the erection of brushwood structures or fences, placed parallel to the coastline to trap sediment, have proven to be highly effective as well, even along eroding coastlines (Winterwerp *et al.*, 2020). So far, these structures have been intended to elevate mudflats high enough (> MSL) to restore mangrove habitat (Mancheño *et al.*, 2022), but the insights gained in this study show that restoration of lower elevation mudflats could already be worthwhile to reduce the chances of mangrove retreat. Foreshore modifications that create wide intertidal foreshores may thus be useful measures to ascertain that wave-attenuating ecosystems such as salt marshes and mangroves become stable enough to be utilized in coastal protection schemes.

3.6 Acknowledgements

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3.7 Appendices

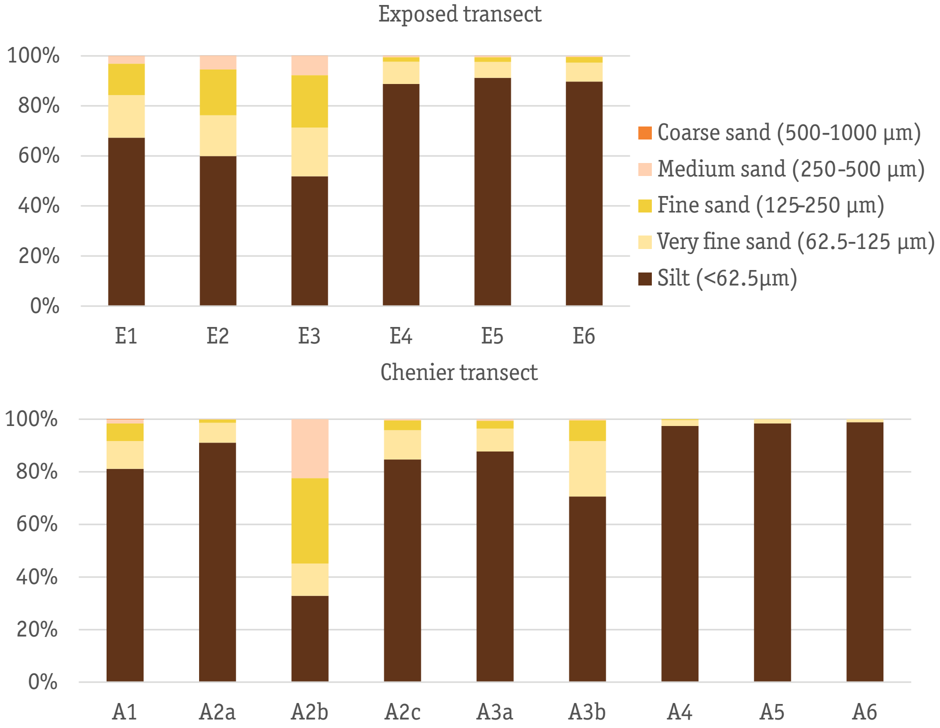

Figure A3-1: *Sediment grainsize distribution at each of the stations in the exposed and chenier transects.*

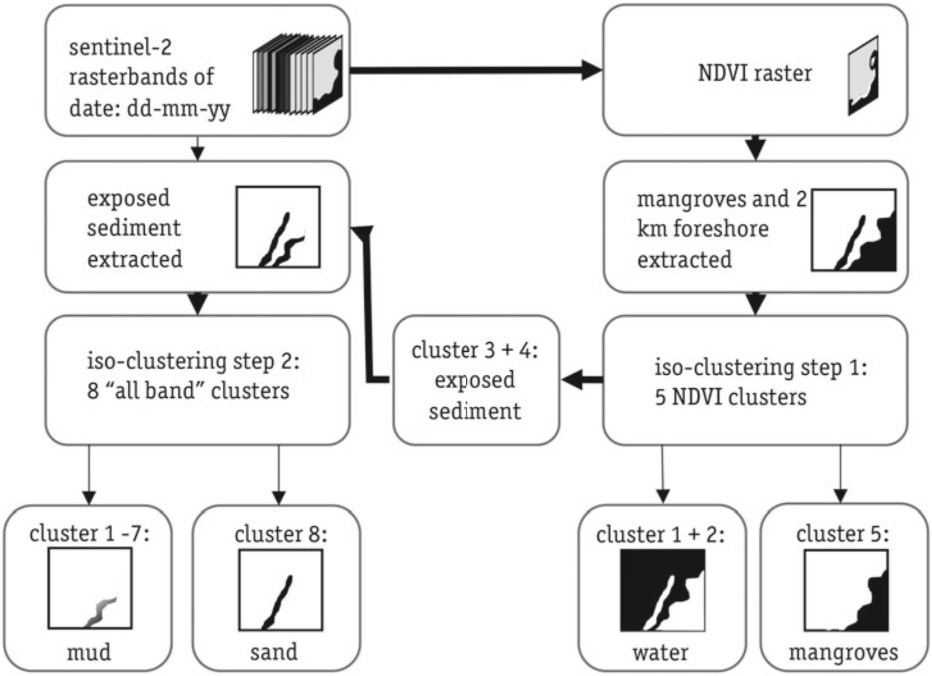

Figure A3-2: *Flow chart of the steps for unsupervised classification of all Sentinel-2 images of dates of interest between 2015 and 2019 in 4 relevant classes (mud, sand, water and mangroves).*

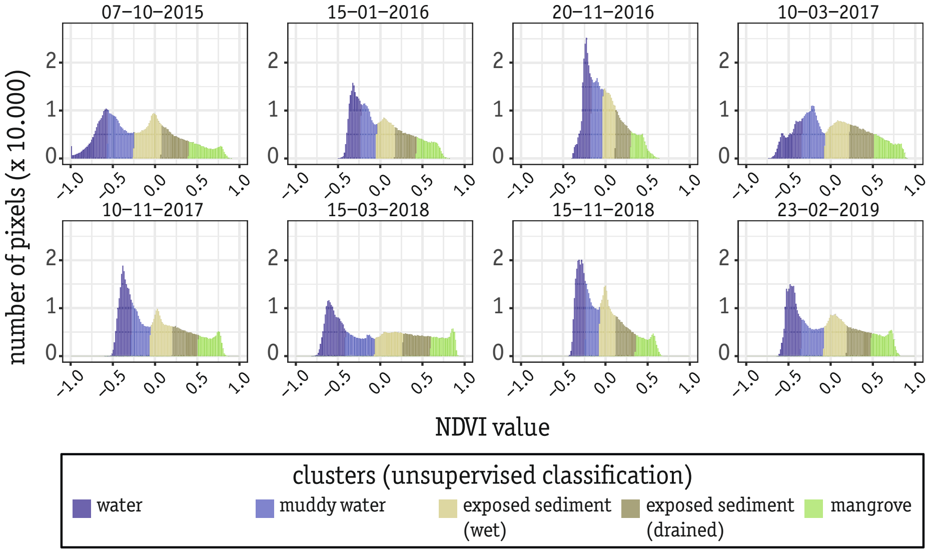

Figure A3-3: *Histograms showing the number of pixels that were clustered into one of the 5 classes by the iso-cluster unsupervised classification tool based on maximum likelihood clustering of the NDVI rasters computed from each of the Sentinel-2 satellite images.*

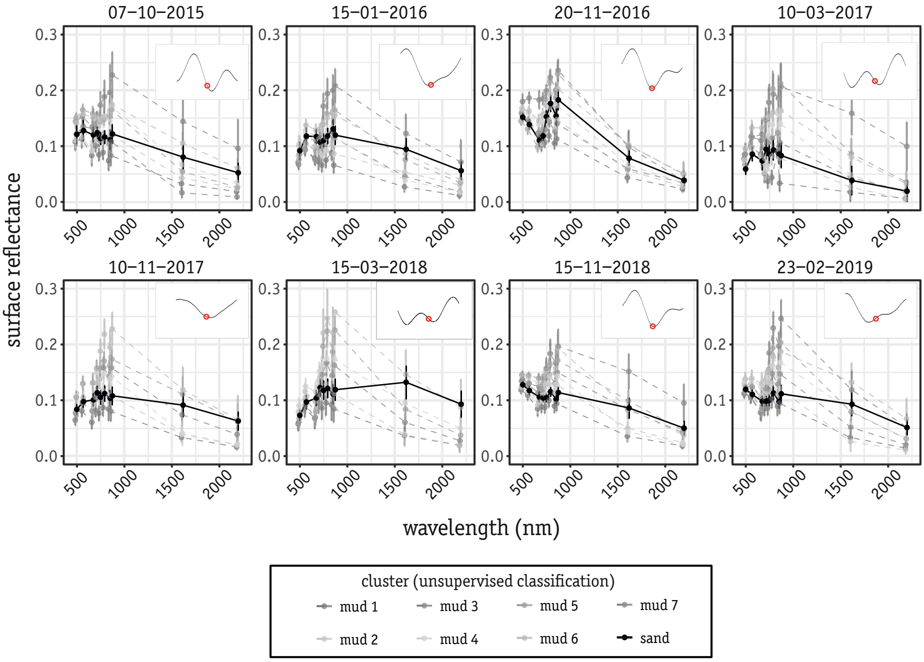

Figure A3-4: *Spectral signals of each of the 8 clusters produced by the unsupervised classification step. The sand cluster distinguishes itself from the "mud" clusters by relatively high surface reflectance values in the SWIR bands (wavelength > 900 nm) and low to average reflectance in the visible (wavelength < 700 nm) and the near infrared spectrum (wavelength: 700-900 nm). The tidal stage at the time and date of image acquisition is indicated with a red dot within the tidal cycle of that date in the upper right corner of each panel.*

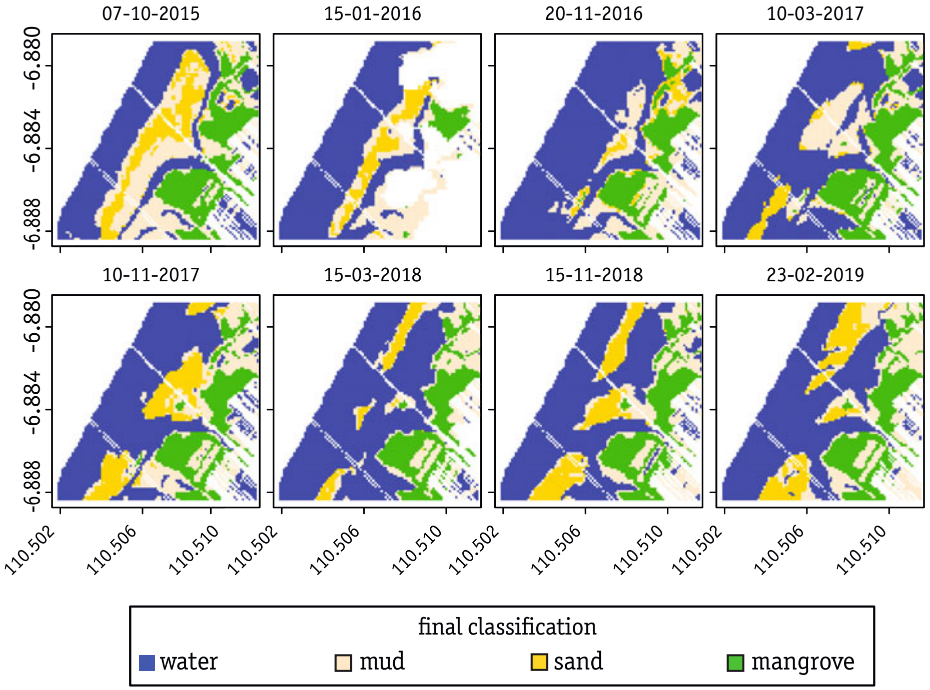

Figure A3-5: *NW bearing lines along which the information of the classified Sentinel-2 images is displayed, with mangroves (green), water (blue), mud (beige) and sand (yellow). These classified images are zoomed in on the site where the two field transects were deployed in November 2017.*

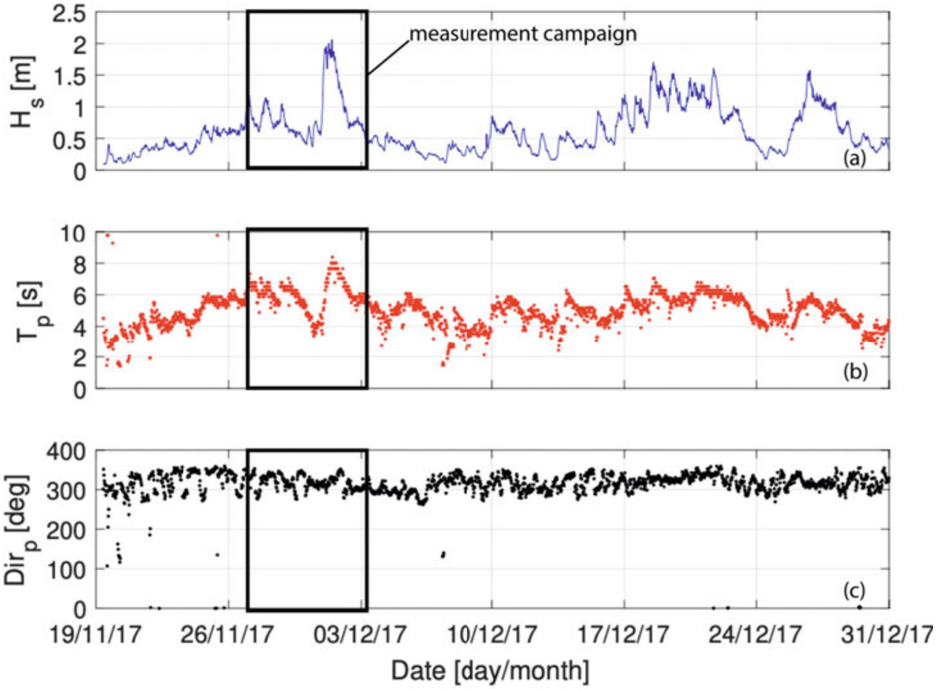

Figure A3-6: *Time series of significant wave height (a), peak wave period (b), and wave direction (c) from an offshore buoy (Wave Droid) during the measurement campaign (shown with a black rectangle). Source: (Van Domburg, 2018).*

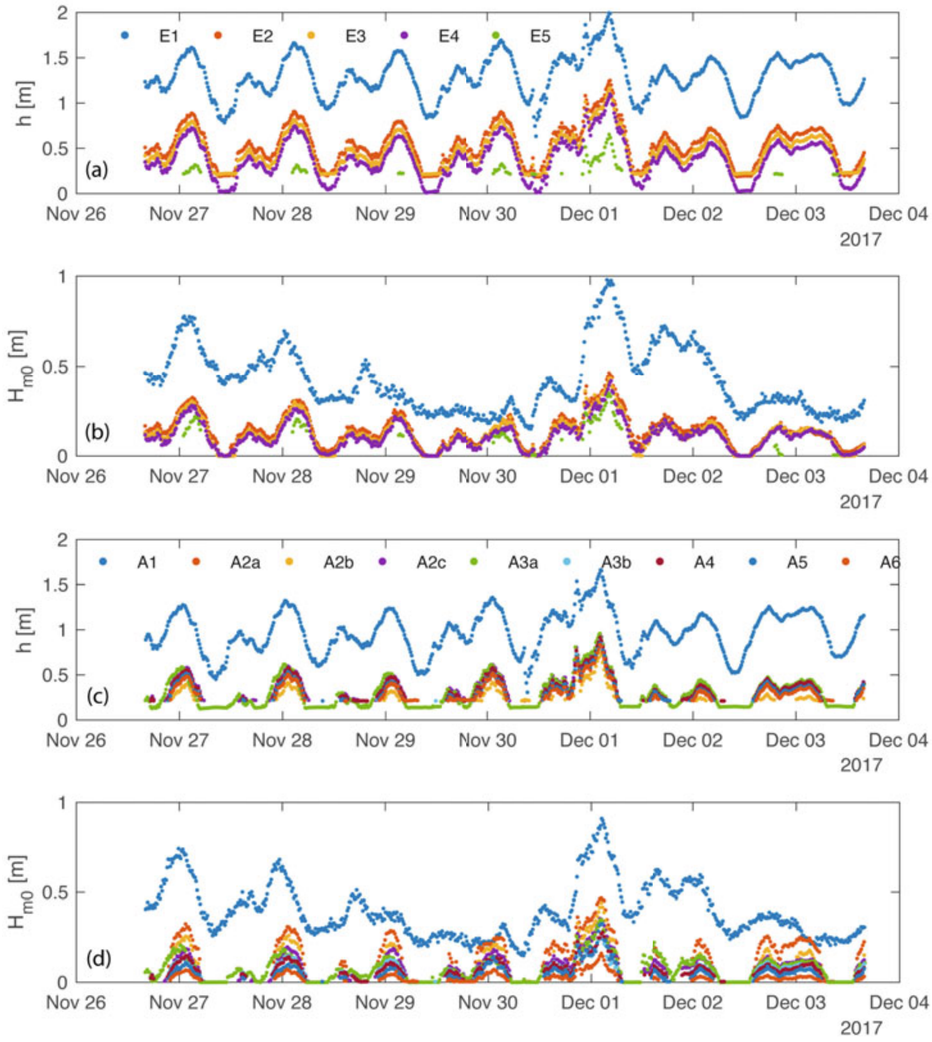

Figure A3-7: *Time series of the water depth in (a) the exposed transect, and (c) the chenier transect. Time series of the wave height in (b) the exposed transect, and (d) the chenier transect.*

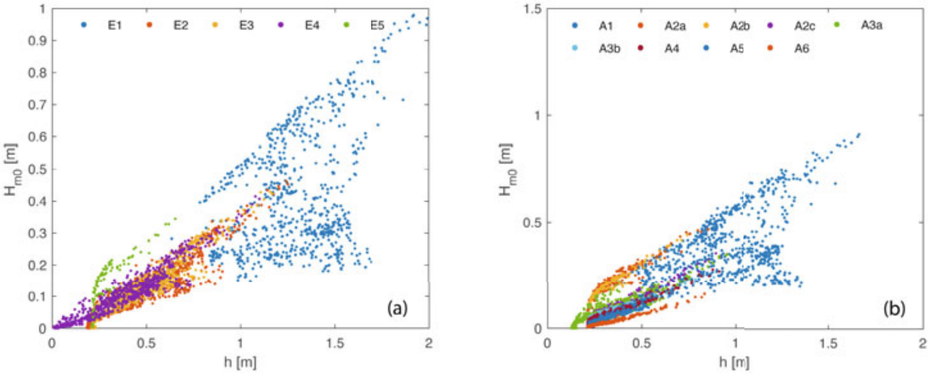

Figure A3-8: *Ratio of significant wave height to water depth at (a) exposed transect and (b) chenier transect.*

Assisted mangrove establishment along vulnerable coastlines

CHAPTER 4

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To plant or not to plant: when can planting facilitate mangrove restoration?

4.1 Abstract

Global change processes such as sea level rise and the increasing frequency of severe storms threaten many coastlines around the world, and trigger the need for interventions to make these often densely-populated areas safer. Mangroves could be implemented in nature-based flood defence, provided that we know how to conserve and restore these ecosystems at those locations where they are most needed. In this study, we investigate how best to restore mangroves along an aquaculture coast that is subject to land-subsidence, comparing two common mangrove restoration methods: (1) mangrove restoration by planting and (2) Ecological Mangrove Restoration (EMR), the assistance of natural mangrove regeneration through mangrove habitat restoration.

Satellite data revealed that historically, landward mangrove expansion into the active pond zone has mainly occurred through mangrove planting on pond bunds. However, there is potential to create greenbelts along waterways by means of EMR measures, as propagule trap data from the field revealed that propagules of pioneer species were up to 21 times more abundant in creeks of the pond zone than near their source in the coastal zone. This was especially true during the prevailing onshore winds of the wet-season, suggesting that smart seasonal sluice gate management could help to efficiently trap seeds in target ponds. In the coastal zone, field experiments showed that permeable brushwood structures, aimed at expanding mangrove habitat, could not sufficiently overcome subsidence rates to increase natural mangrove expansion in the seaward direction, but did significantly increase the survival of already established (planted) seedlings compared to more wave-exposed sites. The survival and growth rate of EMR-supported plantings greatly varied between species. Out of the 4 planted species, *Rhizophora mucronata* had the highest survival (67%) but the lowest growth rate. Whereas the pioneer species *Avicennia alba* and *Avicennia marina* had lower survival rates (resp. 35 and 21%), but significantly higher growth rates, even resulting in fruiting young trees within a 16-month timeframe. Overall, we conclude that i) EMR has potential in the pond zone, given that propagules were observed to reach well into the backwaters; and ii) that mangrove recovery in the coastal zone may be facilitated even at very challenging coastal sites by combining EMR with the planting of pioneer species.

4.2 Introduction

Climate change effects such as sea level rise and an increased frequency and severity of storms threaten many coastlines around the world (IPCC, 2019). The impact of global climate change is often magnified by regional problems such as land subsidence (Hallegatte *et al.*, 2013; Nicholls *et al.*, 2021), and land-use change (Zhang *et al.*, 2021), leading to coastal encroachment. Hence, there is a pressing need for intervention strategies with which to adapt to climate change and reduce flood risks along populated coastlines. Coastal wetlands such as mangroves and salt marshes have gained interest as important ecosystems to reduce vulnerability of coastal communities (e.g. Temmerman et al., 2013; Zhang et al., 2021; Zhu et al., 2020). Coastal wetlands are able to attenuate waves even during storms (Möller *et al.*, 2014; Willemsen *et al.*, 2020), dampen storm surges (Stark *et al.*, 2015; Montgomery *et al.*, 2019), trap sediment and grow with rising sea level (Kirwan *et al.*, 2016). This makes conservation and restoration of such wetlands interesting for coastal protection, and consequently economically interesting. Not only in areas where construction of conventional coastal defence structures, such as sea walls and levees, is not feasible (Winterwerp *et al.*, 2013), but also to reduce costs of such conventional structures at locations where they are feasible (Sutton-Grier, Wowk and Bamford, 2015; Narayan *et al.*, 2016; van Wesenbeeck *et al.*, 2016; Schoonees *et al.*, 2019).

In tropical regions, mangrove restoration is regarded as a widely applicable strategy to enhance coastal safety (e.g. Alongi, 2008; McIvor et al., 2012). In this, planting has long been a favored approach (Ellison, 2000), especially in the lower intertidal zone where land rights are often not an issue (Erftemeijer and Lewis III, 1999). However, this type of planting projects have often given low success rates due to a combination of inappropriate species selection (non-pioneer species) (Ellison, 2000; Primavera and Esteban, 2008) and/or unfortunate site selection (mudflat below mean sea level where mangroves would not naturally occur). When planting is reported to be successful in terms of survival, there is the risk of hampered natural succession, as seedlings are often planted at such high density that the resulting stand leaves little sunlight for potential natural recruitment (Barnuevo *et al.*, 2017; Pranchai *et al.*, 2018; Proisy *et al.*, 2018).

To overcome plantation failure, interests have been shifting increasingly towards restoration of mangrove habitat to promote natural recruitment, rather than using active planting of mangrove propagules or seedlings (Lewis, 2005; Winterwerp *et al.*, 2013, 2020; Balke and Friess, 2016). The focus of mangrove habitat restoration (i.e., EMR), lies in mitigating the establishment thresholds that limit natural propagule settlement and survival (Lewis III and Brown, 2014). Lewis (2005) demonstrated the importance of addressing the first threshold, (i.e. limited propagule availability) (*Figure 4-1*, panel 0), by digging a creek, thus reestablishing aquatic connectivity at a propagule-deprived site, thereby initiating mangrove regeneration over the course of 6.5 years. At more exposed sites, propagules need to overcome additional thresholds to establish successfully for which they need to encounter three size-dependent windows of opportunity: i) a flooding-free phase, ii) followed by a wave-free period and iii) an erosion-free period (*Figure 4-1*, panel 1-3) (Balke *et al.*, 2011, 2015; Balke, Bouma, *et al.*, 2013). Van Cuong et al. (2015) demonstrated that these windows of opportunity can be extended along eroding shores by placing permeable fences parallel to the coastline. These fences both increased the bed level through sediment trapping (i.e. creating 1st window) and mitigated the wave stress (i.e. creating 2nd window), allowing natural seedling recruitment to increase from zero to 24,000 seedlings ha^{-1} over the course of three years. The examples provided by Lewis and Van Cuong et al. clearly show that informed mitigation of stressors restricting natural mangrove recruitment can be sufficient for successful restoration. It is noted that this approach requires proper understanding of the natural system, as the optimal method may differ depending on the specific abiotic and biotic constraints at the selected sites.

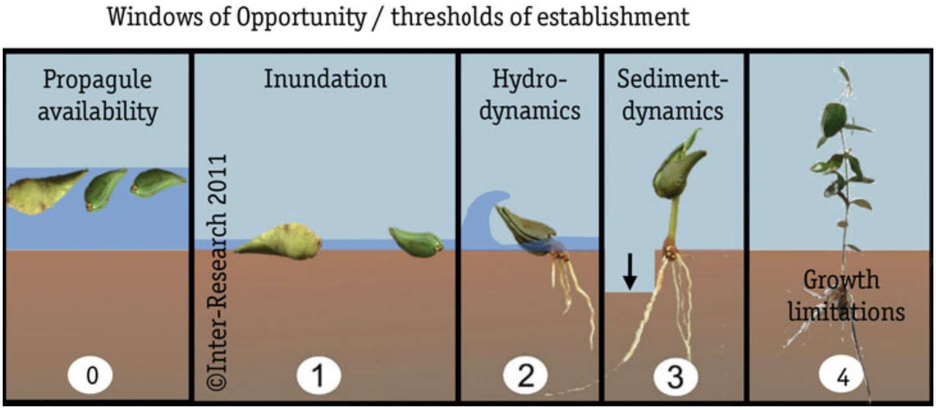


Figure 4-1: *Windows of opportunity that propagules should encounter (or threshold of establishment that propagules should overcome) before they can colonize a site. EMR makes use of measures that extend the window of opportunity for natural establishment (0-3). Mangrove planting is either aimed at overcoming propagule limitation (0) or at skipping the thresholds of propagule establishment (1-3) by using larger seedlings or saplings that would, to a certain extent, be able to withstand conditions that are too harsh for smaller seedlings. Finally, saplings need to overcome limitations of growth (4) such as predation, disease or other stressors to grow into reproductive young trees (Note, panel 0 and 4 are additions to Balke et al. (2011)'s Windows of Opportunity figure).*

Despite the shift in interest towards EMR instead of mangrove planting, the latter remains a popular practice. From a scientific perspective, planting of larger seedlings may be expected to be effective, as it can either overcome propagule limitations (*Figure 4-1*, panel 0) or skip many of the size-dependent establishment thresholds that propagules need to surpass (*Figure 4-1*, panel 4). After all, if propagules have established and grown taller, they can persist at locations where newly established seedlings would be uprooted (Balke *et al.*, 2011, 2015; Balke, Bouma, *et al.*, 2013). In that sense, there may be merit to mangrove planting when the purpose is to accelerate mangrove recovery by skipping the most sensitive propagule life-stages at sites where EMR is not entirely sufficient to improve all conditions required for natural colonization. In other words, could mangrove planting help to accelerate the recovery process when it is combined with EMR, especially at challenging locations?

The purpose of this study was to advance current insights into the best mangrove restoration practices, using a 20 km coastline stretch of Demak district, Java, Indonesia as example of a challenging restoration site. The area has a history of aquaculture and is marked by the associated impaired hydrological connectivity. The ongoing subsidence and subsequent erosion of the shoreline further complicate mangrove restoration. Various restoration projects, initiated by the government, NGO’s and local communities, have however attempted to expand the district’s existing mangroves into a greenbelt using either planting or EMR measures. In seaward direction, restoration has mainly been attempted through planting of *Rhizophora* species. Although more recently, sediment-trapping brushwood structures have also been implemented as an EMR measure to facilitate natural mangrove expansion (Tonneijck *et al.*, 2015). In landward direction, mangrove restoration has mainly focused on compensating the loss of terrestrial vegetation, brought on by salinization after the conversion of rice paddies to aquaculture in the past. Local communities and NGO’s therefore planted *Rhizophora* spp. in the pond zone, both to re-enforce the pond bunds and to create a local source for firewood. More recently, EMR in the active pond zone has also been initiated in the form of the creation of mangrove habitat in active ponds lining the waterway. To this end, pond owners have partitioned their pond, and sacrificed the parts lining the creek for mangrove rehabilitation, motivated by higher yields from mangrove-associated aquaculture (R. Bosma *et al.*, 2020). The observed natural expansion in the coastal zone, the various attempts to plant mangroves, and the EMR measures in both the coastal zone and aquaculture pond zone have all had their positive effect on the mangrove cover in Demak in the last decade.

We here evaluated both the effectiveness of the already installed EMR-measures and mangrove plantings, and the effectiveness of combined EMR-measures and plantings in an experimental setting. The aim was to investigate how mangrove regeneration can best achieved in landward and seaward direction from the subsiding coastline.

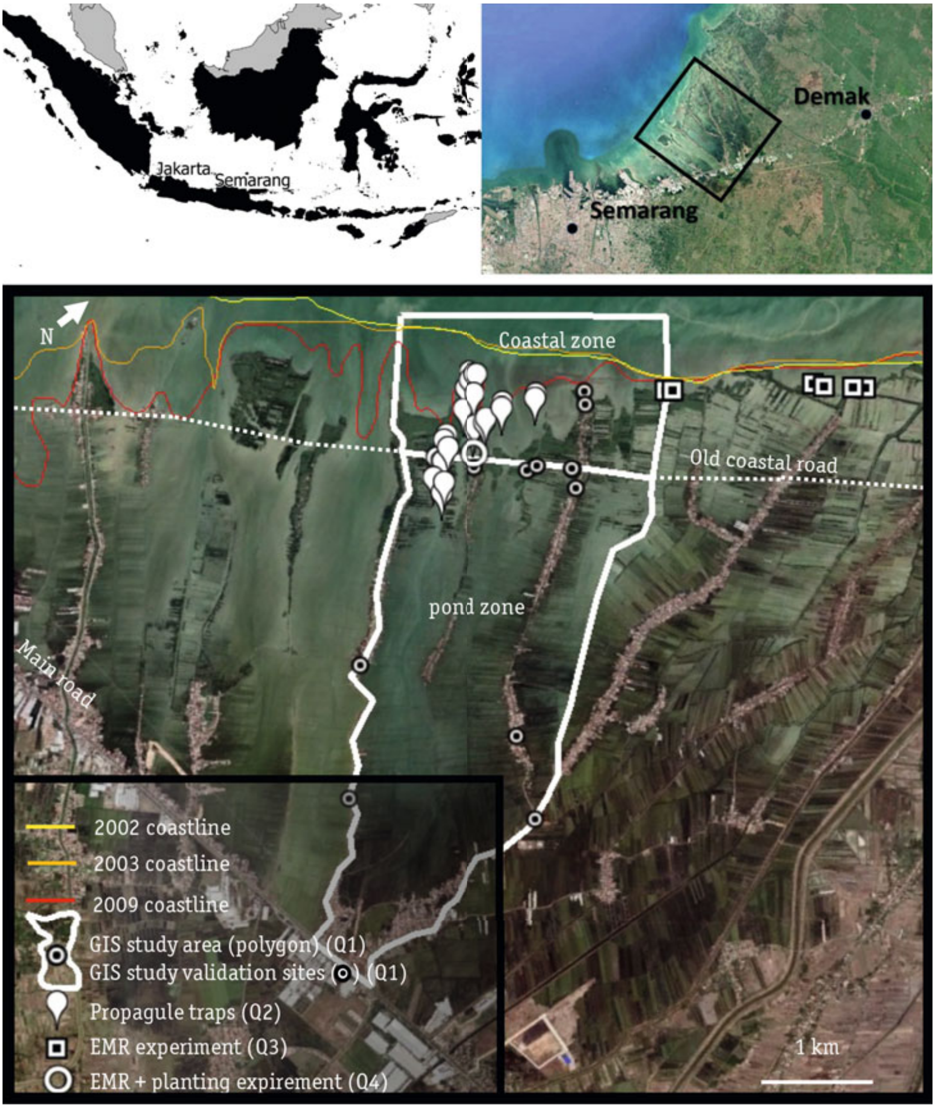


Figure 4-2: *The focus area of this paper, the coastline of Sayung subdistrict (black rectangle) in Demak, Indonesia. Historic coastlines are indicated (before, in between and after two major erosion events), as well as the old coastal road which became the last standing structure resisting coastal erosion, dividing the active aquaculture zone from the dynamic coastal zone. The locations of the four experiments linked to the four research questions are indicated with the icons displayed in the legend. Detailed location descriptions and overviews per experiment are shown in the method section.*

We explored this question with the following sub-questions: 1) how has mangrove expansion occurred in seaward and landward direction in the past, mainly through planting or mainly through natural expansion?; 2) Can hydrological EMR measures (e.g. sluice gate management) induce natural mangrove recovery in the pond zone (i.e. would enough propagules be available at landward sites if hydrological connectivity to target ponds was increased)?; 3) Can wave-reducing and sediment-trapping measures (EMR-structures) induce natural mangrove recovery at challenging sites in the coastal zone by increasing the chances of a) new seedling establishment or b) survival of established seedlings? and; 4) Can mangrove planting in combination with EMR-structures accelerate mangrove recovery at challenging coastal sites?

4.3 Methods

4.3.1 Site description

The study site is located along the coast of Demak district at 6.53 S, 110.30 E on Java, Indonesia. The shore is characterized by a mixed, mainly diurnal tide, with a tidal range of 1.1 m (MMAF, 2012). The region has an average annual rainfall of 2200 mm (Suryadi, Sugianto and Hadiyanto, 2018), with a dry season dominated by an offshore wind (SE) from June to August, and a wet season with onshore wind (NW) from December to February (MMAF, 2012). During the wet season, the maximum significant wave height 4 km offshore is reported to be 1.5 m, with a period of 5.5 seconds (Tonneijck *et al.*, 2015). The onshore waves during this season leave the coastline of Demak prone to erosion and flooding, which are further exacerbated by land subsidence. These processes have led to two major erosion events of Demak's shoreline: One between 2002 and 2003 causing the coastline directly east of Semarang city to retreat with 3 km, and one between 2007 and 2009 affecting our study area, 5 km east of Semarang (*Figure 4-2*). This erosion event removed the majority of the aquaculture ponds 1 km seaward of the coastal road, making the coastal road an unintended flood defence structure.

4.3.2 Rationale

We studied the natural and assisted processes of mangrove expansion in landward direction (i.e. the pond zone) and seaward direction (i.e. the coastal zone) from the current coastline in the project area (the old coastal road). The pond zone in the study area is characterized by active traditional aquaculture ponds, and drowning abandoned aquaculture ponds towards the coast. The coastal zone in the study is defined as the area ranging from MHW to MLW (the maximum depth at which the EMR-structures are placed), which ranges roughly 0 to 600 meters from the shoreline (Tonneijck *et al.*, 2015). In order to shed light on best practices for mangrove recovery, we addressed research questions 1-4 in the introduction using the following correspondingly numbered methods. 1) GIS monitoring of natural and planted mangrove vegetation in the coastal zone and pond zone using time series of satellite images (GIS study in Figure 4-2). 2) A year-round propagule monitoring campaign seawards and landwards of the old coastal road (propagule traps in Figure 4-2) to assess the potential for natural mangrove colonization in the active pond zone if maximal EMR-hydrology would be applied. 3) A field experiment at multiple coastal sites with different wave exposure conditions to understand the effect of EMR-structures on mangrove recruitment and seedling survival (EMR-experiment in Figure 4-2). Finally, 4) a field experiment to study the effect of mangrove planting in combination with EMR-structures on seedling growth and survival (EMR + planting in Figure 4-2). These four experiments are further elaborated on below.

4.3.3 Q1. GIS study: Which mangrove recovery process is dominant in each zone?

To quantify the importance of the two contrasting processes of mangrove recovery in Demak, we quantified changes in vegetation types (i.e. natural mangroves, planted mangroves and terrestrial vegetation) in the coastal zone and the pond zone of the study area (Figure 4-2) over multiple years. We analysed high resolution (<1 m^2) satellite images in Google Earth for vegetation cover, starting in 2005 (i.e. before the 2007 erosion event), in 2010, and then on a yearly basis from 2013 until 2018 (*Table A 4-1*). The timeline tool of Google Earth displays mosaics of multiple high-resolution satellite images (i.e. acquired at different dates or from different sources) for the area of interest. We therefore attempted to select the most complete image (no mosaic) for each year of interest (*Table A 4-1*). Due to limited coverage, we were obligated to use a mosaic of two different years for the baseline year 2005, where the majority of the image was comprised of a 22-4-2005 image, and a small corner of the area of interest was dated 31-5-2003. Vegetation cover in both 2003 and 2005 was very low however, so the mosaic was deemed appropriate to use as a pre-erosion baseline.

Images from each year were exported from Google earth as kml files and imported to ArcGIS with the “kml to layer tool”. Vegetation was manually digitized per image in the two areas of interest, the coastal zone seawards of the old coastal road, and the pond zone landwards of that road (e.g. *Figure A 4-1*). The sparse and fragmented mangroves in the pond zone, interspaced with muddy and algae rich aquaculture ponds, made semi-automated recognition of vegetation cover challenging for our study area. Especially since the imagery was limited to wavelengths in the visible range of the light spectrum (Red, Green and Blue). However, the high-resolution imagery did allow for precise “manual” photointerpretation. A method that is recognized as reliable for both coastal object identification (Chinnasamy and Parikh, 2020) and fine-scale forest-cover mapping in general (Castilla, Hay and Ruiz-Gallardo, 2008). In our study area, we identified and categorized vegetation “objects” based on location and pattern. For instance, vegetation along the southern main road (Figure 4-2) or near active rice paddies (i.e. fresh water available) was classified as “terrestrial”, whereas vegetation lining the coast or aquaculture ponds was classified as "mangroves". Planted mangroves were distinguished from natural stands based on their darker green colour and structured pattern of evenly spaced straight lines, as opposed to the grey-green colour and more diffuse pattern displayed by natural mangrove stands. Ultimately, we classified all vegetation into one of three categories: natural mangrove cover (predominantly *Avicennia* species), planted mangrove cover (predominantly *Rhizophora mucronata*), or terrestrial tree cover, (e.g. garden plants, fruit trees, and palm trees).

Validation of these three vegetation types was conducted during a field-truthing visit in November 2018 in which the vegetation in randomly selected plots of differing sizes (40-2400 m^2) was identified to species level (*Figure A 4-1*). Within each plot, we also collected forest structure data, both by counting the number of individuals per species and by recording each tree's diameter at breast height (DBH). From these parameters, we calculated forest parameters such as tree density (n ha^{-1}) and basal area (m^2 ha^{-1}). In addition to validation plots inside the study area, 13 additional validation plots were selected in the pond area 10 km to the north-east, which was still unaffected by erosion and salinization. These served as a reference for terrestrial vegetation diversity inside the area back in 2005, before the onset of coastal erosion. Field validation of the 2018 GIS vegetation categories showed that the visual characterization of vegetation from the satellite images was accurate, with an overall accuracy of 89 % and a kappa-coefficient of 0.81.

Trends in vegetation cover changes over time for planted and natural mangroves were investigated per zone and species with a generalized regression model, using the surface area of the vegetation category of interest as a response variable, assuming a Gaussian distribution for the two mangrove types (planted vs natural). A logarithmic regression model was used to analyse the decline in terrestrial vegetation. Data from the field campaign in terms of DBH, basal area and stem density were compared between planted and natural mangrove stands with Kruskal-Wallis tests, and the tree density differences between species and stand types was tested with a two-way ANOVA after log-transformation of tree density.

4.3.4 Q2. Propagule traps: Can EMR-hydrology support natural mangrove recovery in the pond zone?

To investigate if natural mangrove recovery in the pond zone could be supported through EMR-hydrology, we studied landward propagule dispersal throughout the main creek of the pond zone and compared that to propagule abundance in the coastal zone (the propagule source). We did this by trapping floating propagules in modified crab-traps, to compare weekly propagule abundance in both zones. The propagule traps

(*Figure A 4-2* a) were designed as netted bamboo cylinders 120 cm long, with a circular 55 cm diameter opening at one end. During deployment, 50% of this opening was always above the water surface, as the traps were maintained horizontally in the water using floats. The floating traps were loosely anchored to an anchoring pole in such a way that the trap could rotate freely around the pole, so that the circular opening was always facing the current.

The traps were placed across the creek and throughout the coastal bay. In total, 40 traps were placed throughout the basin, 9 traps in the creek of the pond zone and 31 traps in the coastal zone (*Figure 4-2*, close-up in *Figure A 4-2* b) and they were emptied weekly throughout the year in 2017. Data from the wet season are partially missing as the coastal traps were continuously destroyed by waves, despite repeated efforts to replace them. In addition, the rough seas made it hard to reach and empty the remaining traps regularly.

The variability of propagule abundance over time and in space was compared between the pond zone and coastal zone (source zone) using a generalized linear regression model, assuming a negative binomial distribution (package glmmTMB in RStudio version 1.0.143). Propagule counts of the three most abundant species observed in the traps (*Avicennia marina*, *Avicennia alba* and *Rhizophora mucronata*) were used as response variables, using week number and zone of interest (near the source or in the pond area) as explanatory variables. Two other species were sporadically found in the traps, but their numbers were too low for meaningful comparison between the two zones. The effect of week number on the residuals of the model was furthermore tested by fitting a smoothing term using a general additive model (GAM) from R package mgcv to investigate the effects and significance of seasonality (ie. the non-linear effect of “week number”) on propagule counts.

4.3.5 Q3. EMR-structures: Can EMR-structures induce natural mangrove recovery at coastal sites?

We quantified the effect of EMR-structures (i.e. sediment trapping brushwood structures) on natural seedling establishment and survival of established seedlings. To do so, we set up a seedling monitoring study using 2 * 2 m plots at coastal sites above mean sea level (MSL) (*Figure 4-2*). To evaluate the wave sheltering effect of the structures, we set up seven monitoring blocks: three blocks that were artificially sheltered by permeable structures constructed the previous season (EMR-structure sites) (*Figure A 4-3* c), three blocks at a naturally sheltered site behind a vegetated chenier (i.e. elevated sand bank with vegetation on top; positive controls)(*Figure A 4-3* b), and one block at an exposed site above MSL (negative control) (*Figure A 4-3* c). We were only able to include one block at an exposed site, due to the limited availability of wave-exposed sites above MSL. Each block consisted of three 2 * 2 m plots which were used to monitor natural mangrove recruitment in terms of number of seedlings per species, seedling survival and new seedling recruitment over one year in 2017 (*Figure A 4-3* d).

To investigate the survivability of established mangrove seedlings at these locations, we designed a small-scale planting experiment in which seedlings of two mangrove species were planted in two of the three 2 * 2 m plots per block in the following year (i.e. 2018; *Figure A 4-3* e). One of the 2 * 2 m plots per block was used to plant *R. mucronata* seedlings (the species most commonly used for mangrove planting in this area) and the other was used to plant *A. alba* seedlings (the most abundant, naturally expanding mangrove species in the area). The seedlings were planted in a matrix of 5 * 5 individuals per plot, resulting in a planted seedling density of 6.5 m^{-2}. Seedling height and viability was recorded at (t_0), and again after half a year and a full year. Differences in survival between the species and between the shelter-types was tested with a Kaplan-Meijer survival test, and differences in average growth rate between species were tested with a t-test.

4.3.6 Q4. EMR-structures + planting: Can mangrove planting combined with EMR-structures accelerate recovery?

To further investigate if and how mangrove planting could accelerate mangrove recovery at sites with EMR-structures, we designed a larger mangrove planting experiment with which we closely monitored the survival and growth of the multiple mangrove species planted at the oldest EMR site in the area (constructed and maintained since 2013, open circle in *Figure 4-2*). The planting experiment was set up in 20 plots of 2 * 2 m in which four mangrove species were randomly planted in a matrix of 5 * 4 individuals (*Figure A 4-4*). The 4 species that were planted all occurred naturally in the vicinity: *A. alba*, *A. marina*, *R. mucronata* and *Rhizophora apiculata*. Prior to planting, the seedlings were raised in a nursery. *A. marina* and *A. alba* were collected as propagules from the water, and *R. mucronata* and *R. apiculata* were collected as young wildlings from underneath a variety of mature trees in the region to ensure some genetic heterogeneity. Propagules and wildlings were then raised in a nursery for two months, of which six weeks in the shade and two weeks in full sunlight to acclimatize the seedlings to life in the field.

4.4 Results

4.4.1 Q1. GIS study: Which mangrove recovery process is dominant per zone?

Satellite data

The GIS time series of different types of vegetation cover showed a decline of terrestrial vegetation and an expansion of mangrove vegetation throughout both the coastal and pond zone from 2005 until 2018 (*Figure 4-3*). The 64.9 ha of terrestrial vegetation in the pond zone in 2005 decreased logarithmically with $e^{-0.24}$ ha y^{-1} ($R^2 = 0.9$, $p < 0.001$) between 2005 and 2018, leaving only 2.8 ha of terrestrial vegetation in 2018. Meanwhile, natural mangrove recovery was especially dominant in the coastal zone and showed an average increase of 3.9 ha per year ($R^2 = 0.85$, $p < 0.001$), increasing from 6.3 ha in 2005 to 46.8 ha in 2018. Mangrove expansion through planting was significantly less rapid in the coastal zone ($p < 0.01$), with an average expansion rate of 1.3 ha per year ($R^2 = 0.56$, $p = 0.03$).

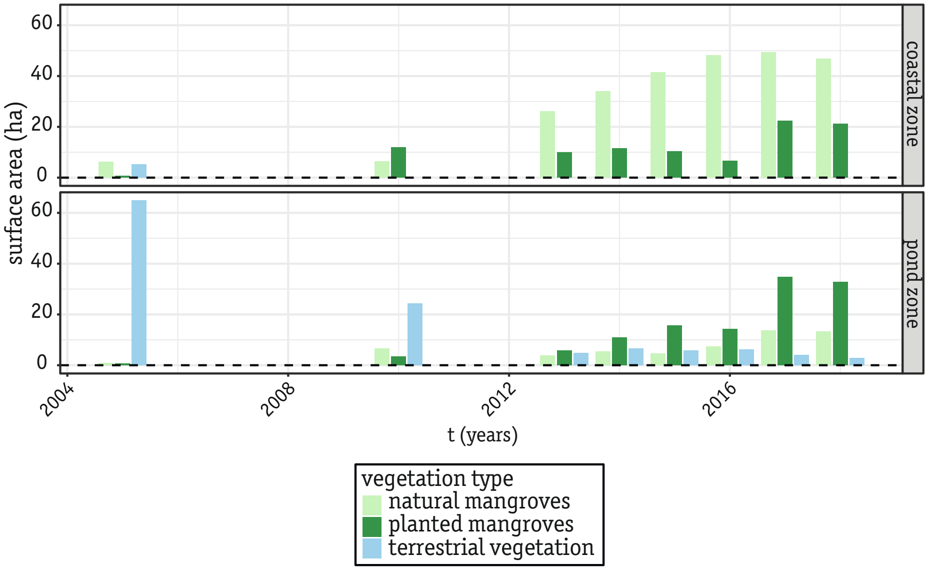


Figure 4-3: *Trends in total surface area (ha), digitized from high resolution images available in Google Earth, of the three vegetation types (terrestrial vegetation, planted mangroves (R. mucronata) and naturally-recovered mangroves (Avicennia spp.) in the coastal zone and the pond zone from 2005 until 2018.*

In contrast, mangrove expansion in the pond zone could be mostly attributed to planting efforts, with an average increase of 2.5 ha per year ($R^2 = 0.62$, $p = 0.01$) of planted mangroves as opposed to an average expansion rate of 0.8 ha per year in natural mangrove stands ($R^2 = 0.53$, $p = 0.02$). This resulted in 32.9 ha of planted mangroves in the pond zone in 2018 (almost exclusively on pond bunds), while natural mangrove cover (inside abandoned ponds) amounted to only 13.3 ha. The relative success of planting in the pond zone can most likely be attributed to the much slower natural mangrove expansion in this area ($p < 0.01$), as the absolute expansion rates through planting were not significantly faster in the pond zone compared to the coastal zone ($p = 0.17$).

Field data

In general, the composition and structure of the mangrove stands classified as “planted” in the GIS study were significantly different from the stands classified as “natural”. Planted mangrove stands showed a significantly different species composition than natural mangrove stands ($F = 19.6$, df = 2, $p < 0.001$), with 95% of the planted stands consisting of *R. mucronata*. In contrast, natural mangrove stands consisted of 92% *A. marina*, and only 4% *R. mucronata* (*Figure 4-4*). There were no significant differences found between planted stands and natural stands in terms of basal area ($m^2\ ha^{-1}$) and tree density ($n\ ha^{-1}$) (*Table A 42*), although the trees in the natural stands had a significantly larger DBH than the trees in the planted stands ($X^2 = 10.5$, df =1, $p < 0.01$). Most trees in the planted stands had a DBH of 3 to 7 cm, whereas the trees in the natural stands were much more diverse in terms of stem diameter, with many young trees, and progressively fewer older and thicker trees (*Figure 4-4*).

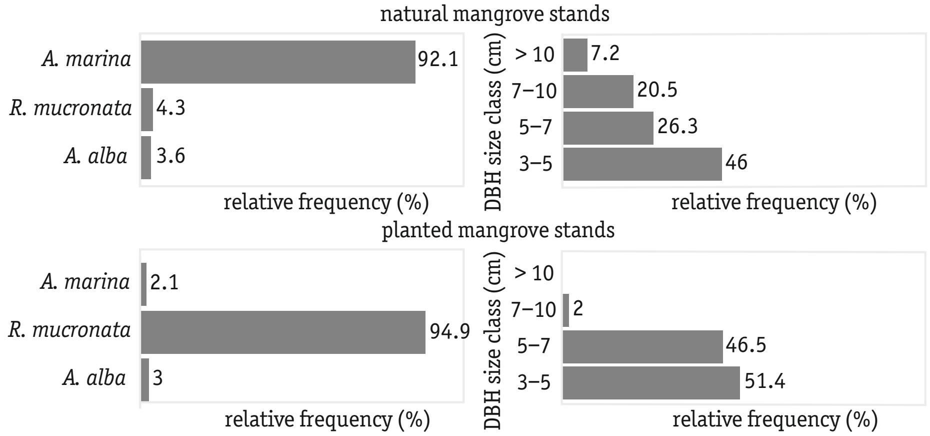


Figure 4-4: *Forest structure of natural and planted mangrove stands in field validation plots of the GIS study in terms of relative tree density per species (left) and the relative distribution of trees per DBH size class (right).*

The field observations further revealed that the class of declining terrestrial vegetation was composed of fresh to brackish-water village “forest gardens”, used by the villagers for basic fruit and materials provision. Based on a tally of 3.5 thousand trees in remaining forest garden fragments (also including the nearby reference area which was not severely affected by increased salinization), these forests were found to be composed of more than 50 different species of trees, almost all of which had an important utilitarian role in terms of food, forage or materials provision. More details on the composition and uses of these forest gardens as well as their degradation and loss caused by coastal erosion and salinization will be presented elsewhere.

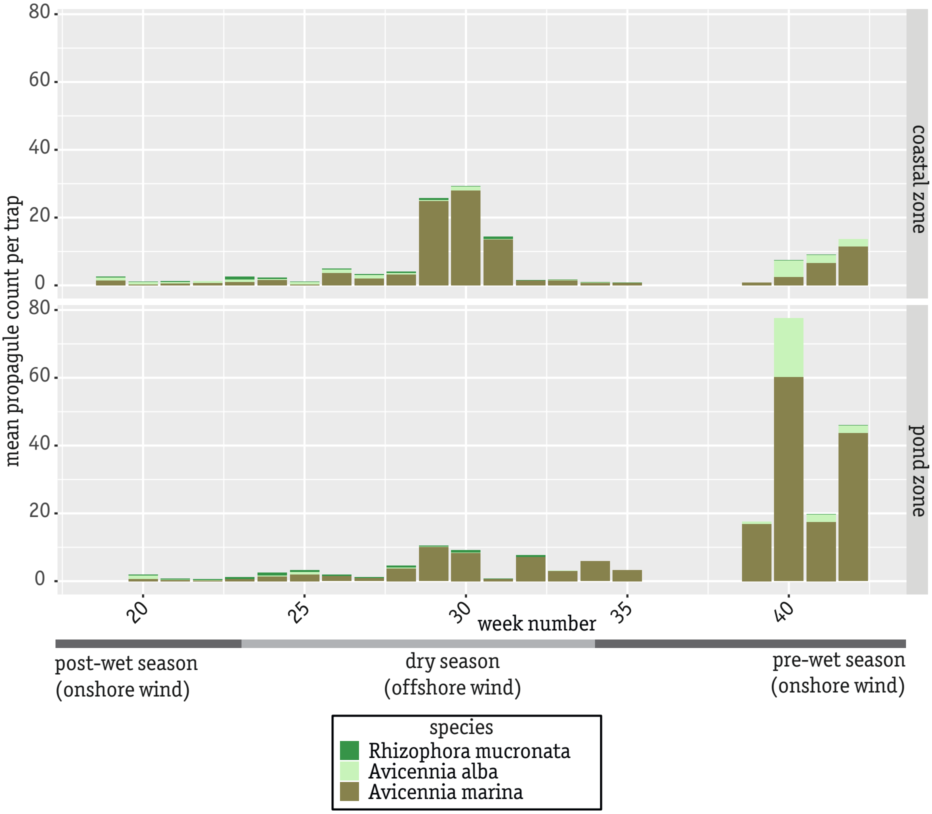


Figure 4-5: *Average weekly propagule counts of the three most abundant mangrove species per trap in the coastal zone (propagule source) and in the pond zone. Data from the mid-wet season are missing due to storm damage to the traps.*

4.4.2 Q2. Propagule traps: Can EMR-hydrology induce natural mangrove recovery in the pond zone?

The propagule traps that were deployed throughout the area proved effective in trapping propagules of multiple mangrove species. Propagules of *A. marina*, *A. alba* and *R. mucronata* were the most abundant (*Figure 4-5*), although few propagules of *Rhizophora stylosa* and *Avicennia officinalis* were sporadically found as well between week 32 and 35 (not visible in *Figure 4-5*). The relative abundance of propagules in the coastal zone versus the pond zone depended significantly on the time of year ($p = 0.01$), with propagule counts in the pre-wet season (week 39-42) being 2 to 21 times higher in the pond zone than in the coastal zone (*Figure 4-5*). While in the mid-dry season (week 28-31), propagule counts in the pond zone were lower than in the coastal zone. This temporal variation in propagule abundance was caused by the timing of the propagule rains of the two most abundant pioneer species in the area: *A. alba* and *A. marina*.

Both *Avicennia* species showed a clear temporal signal that explained 9.6% and 3% of the variance in seed abundance of, respectively, *A. marina* (gam: $F = 5.6$, $edf = 8.2$, $p < 0.0001$) and *A. alba* (gam: $F = 3.9$, $edf = 2.7$, $p = 0.007$). *A. marina* showed a clear peak around week 30 (in the middle of the dry season) and an increase in propagule counts from week 40 onward at the start of the wet season. The pre-wet season peak was visible in both the coastal zone and the pond zone, although more pronounced in the pond zone (*Figure 4-5*). The mid-dry season peak was only observed in the coastal zone. *A. alba* only showed a peak before the wet season around week 40 and at the end of the wet season around week 20. This could be caused by one big propagule rain over the whole wet season, but due to missing data over the wet season we cannot be sure of this. *R. mucronata* did not show a significant seasonal pattern in propagule availability, with propagules present year round, though in low abundance.

4.4.3 Q3. EMR-structures: Can EMR-structures induce natural mangrove recovery at coastal sites?

Monitoring of plots at coastal sites revealed that most sites behind EMR-structures and the exposed sites could not (yet) support natural mangrove recruitment in 2017. Only the naturally sheltered site harboured an abundance of natural seedlings at the start of the experiment, and showed significant new recruitment of natural seedlings over the course of the monitoring campaign (*Figure 4-6*a). One of the EMR-sites did have a few seedlings of *A. marina* that survived over the subsequent wet season (*Figure 4-6*a), and it showed the establishment of one new *A. marina* seedling, suggesting that this particular EMR site had favourable establishment conditions.

The small-scale experimental planting of *R. mucronata* and *A. alba* seedlings in the following year, to quantify the effect of EMR-structures on seedling survival and growth of established seedlings, showed that survival of both species decreased over time (*Figure 4-6*b). This is an expected natural process in such a dynamic environment, but there were some clear differences between species. Overall, the relatively large *R. mucronata* seedlings showed significantly higher survival rates than the smaller *A. alba* seedlings (Kaplan-Meier, $X^2 = 109$, $p < 0.0001$). However, *A. alba* grew 61 mm $month^{-1}$ faster on average than *R. mucronata* ($p < 0.001$).

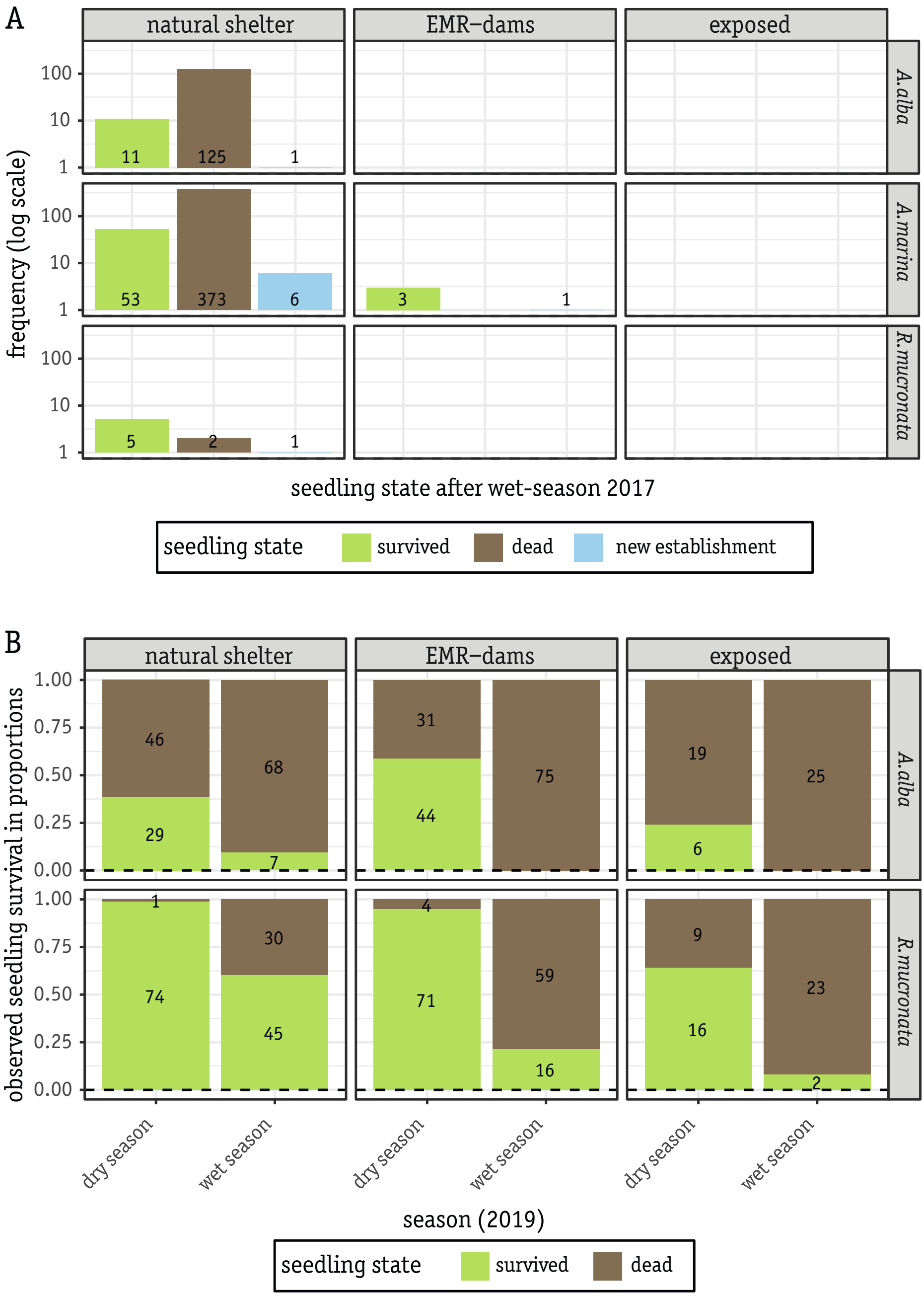


Figure 4-6:* *a. *Existing and new natural establishment of mangrove seedlings at the three differently sheltered coastal sites above MSL (natural shelter, artificial shelter (EMR), and wave-exposed), and survival of existing natural recruits over the course of the experiment.* ***b.*** *Survival of A. alba and R. mucronata seedlings that were planted at naturally sheltered sites, at habitat restoration sites behind an EMR-structure, and at an exposed site.*

Survival rates at the exposed sites and EMR-structure sites were significantly lower than at the naturally-sheltered sites (Kaplan-Meier, $X^2 = 27.8$, $p < 0.0001$). However, the EMR-structures did increase the survival of *R. mucronata* compared to the wave-exposed site (binom.test, $p < 0.001$). Ultimately, the best surviving seedlings were those at the naturally-sheltered site behind a vegetated chenier (Kaplan-Meier, $X^2 = 148$, $p < 0.0001$).

4.4.4 Q4. EMR-structures + planting: Can mangrove planting combined with EMR-structures accelerate recovery?

The four species that were planted above mean sea level at an EMR-structure site, but just below the threshold of natural mangrove recruitment, showed clear differences in survival and growth (*Figure 4-7*a). Over the course of 16 months, the survival rate of the *R. mucronata* seedlings was significantly higher than of the *A. alba*, *A. marina* and *R. apiculata* seedlings (Kaplan Meier, $X^2 = 131$, $p < 0.0001$). At the end of the experiment, only 6% of *R. apiculata* seedlings had survived, as opposed to 21% and 35% of the *A. marina* and *A. alba* seedlings, and 67% of the *R. mucronata* seedlings (6% > 21% > 35% > 67%, $p < 0.001$). However, a comparison of the growth rates of the surviving seedlings (*Figure 4-7*b) showed that *A. alba* and *A. marina* had significantly higher growth rates than *R. mucronata* and *R. apiculata* survivors (GLM, $p < 0.001$), which even resulted in two fruiting *A. marina* individuals by the end of the experiment.

4.5 Discussion

In this study, we aimed to evaluate the potential for- and the effectiveness of EMR and mangrove planting in the pond zone and coastal zone of a site in Northern Java. Our aim was to understand under what conditions planting, EMR or a combination of both can best be applied.

4.5.1 Mangrove restoration in the pond zone

4.5.2 Q1.a Which mangrove recovery process is dominant in the pond zone?

In the pond zone, GIS information showed that disappearing terrestrial trees were partly substituted by salt tolerant mangroves over time, mostly as a result of planting on pond bunds. The spatial continuity of these planted stands was therefore fragmented, with a very limited patch size (max 5 m wide cross-shore and 200 m long longshore) of only a single species. Small forest patches are known to be vulnerable to disturbance and likely have little value in terms of coastal protection (Koch *et al.*, 2009) or biodiversity (Hanski, 2015). However, when viewed as fragments to the larger forest, these smaller patches may provide a propagule source for further forest expansion, although natural expansion of *Rhizophora* species was neither observed on satellite images nor in the field, suggesting that there was little suitable natural habitat for this particular species. Nevertheless, the planted stands may account for some ecological connectivity and structural heterogeneity of the larger system (Fahrig, 2017).

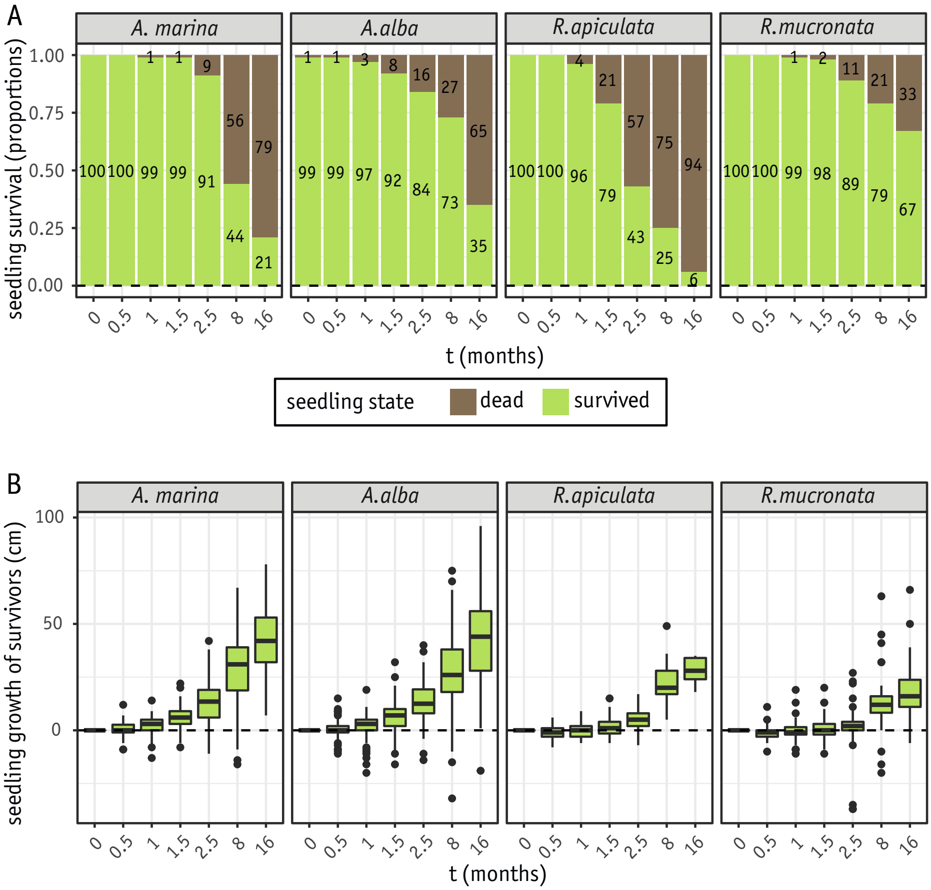


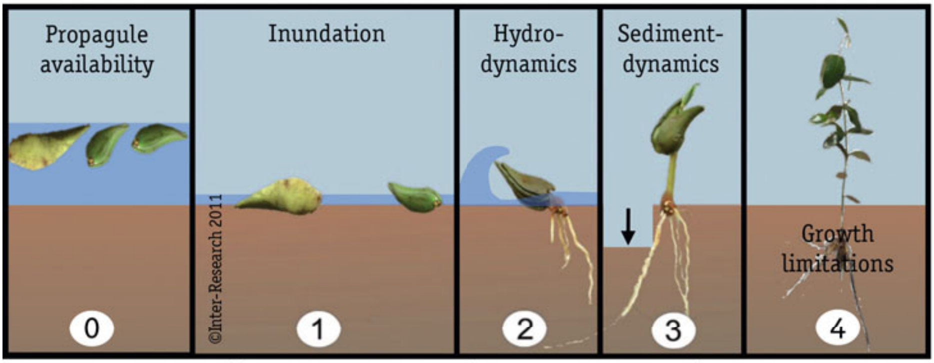
Figure 4-7: ***a.*** *Observed survival over time of four different mangrove species that were planted at a sheltered site behind an EMR-structure around mean sea level.* ***b.*** *Average seedling growth of surviving seedlings from the four planted mangrove species.*

In addition, the planted patches might still be useful for shade, timber harvest (Van Oudenhoven *et al.*, 2015), mitigation of some of the CO_2 release from the pond bunds (Sidik and Lovelock, 2013), and some limited fish pond water-quality regulation (Rönnbäck and Primavera, 2000). However, decomposing leaves may also worsen the water quality (R. H Bosma *et al.*, 2020). Finally, planted mangrove stands are valued by the local communities for their aesthetic value in a lowland previously devoid of tree-like vegetation (pers. comm. local communities). While mangrove planting on banks of active ponds may have some advantages, planting mangroves inside disused ponds without EMR measures is reportedly a poor practice, either resulting in slow growth (Proisy *et al.*, 2018) or mortality of planted seedlings (Brown *et al.*, 2014). Also, in the case that natural colonizers can reach the site, additional planting often hampers the growth and succession of natural colonizers (Matsui *et al.*, 2010; Proisy *et al.*, 2018). Therefore, we conclude that planting may be a valuable measure in the pond zone but only when applied on pond bunds when aquaculture ponds are still active, bearing in mind that this will not result in a fully functional mangrove forest (*Figure 4-8*).

4.5.3 Q2. Can EMR-hydrology support natural mangrove recovery in the pond zone?

Besides planting on pond bunds, there is also potential for natural mangrove recovery inside aquaculture ponds. That hydrological EMR measures can be effective in ponds has been demonstrated in several studies on the effect of strategic pond breaching in disused or degraded ponds (e.g. Brown et al., 2014; Matsui et al., 2010). In such areas, natural recruitment often fails, either because propagules do not reach the targeted aquaculture pond (Di Nitto *et al.*, 2013), or because the sediment condition of the pond floor is not yet favourable for seedling survival. This was for instance the case in degraded ponds in North Sulawesi, where six planting attempts over the course of nine years all resulted in total seedling mortality. It was not until man-made drainage channels were filled-in and pond bunds were breached in strategic directions that the sediment condition improved (Brown *et al.*, 2014). These measures subsequently resulted in the recruitment of 32 mangrove species, with overall seedling densities up to 20.000 ha^{-1}. In Demak, disused aquaculture ponds are generally abandoned because pond bunds can no longer be maintained under the rising water levels. Along the coast, these ponds typically fill up with sediment quickly and mangrove recruitment follows not long after (van Bijsterveldt *et al.*, 2020). Recruitment in abandoned ponds therefore does not seem to be an issue in this area. However, our findings show that propagules can travel further inland than where the current natural recruitment occurs based on the satellite data. In addition, despite the lack of mid-wet season data, it is clear that the major propagule release peak of the two pioneer species (i.e. *A. marina* and *A. alba*) co-occurs (at least partly) with the NW monsoon. Wind is known to be an important factor in propagule transport (Di Nitto *et al.*, 2013; Van der Stocken *et al.*, 2013; Van der Stocken, Vanschoenwinkel and De Ryck, 2015). Therefore, the onshore winds during the wet season most likely propelled large quantities of propagules further inland, explaining the relative larger abundance of propagules in the pond zone compared to the coastal zone during that season. These observations suggest that there is additional unutilized potential for EMR measures in the active aquaculture zone (*Figure 4-8*).

Windows of Opportunity / thresholds of establishment



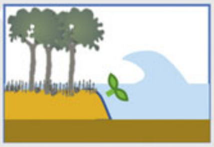
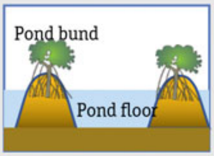
Baseline situation	EMR through:	Planting on:	EMR + planting
Coastal zone	Permeable dams	Foreshores (>MSL)	Planting behind dams
 0 1 2 3 4	0 1[a]* 2[b]* 3[c] 4[c] ~ Marginally successful in this study, but works in non-subsiding areas based on literature[c]	0[d] 1* 2* 3* 4* X Unsuccessful seaward of naturally colonized sites, unless propagules (of pioneers!) are the limiting factor[d]	0 1* 2* 3* 4* ✓ Successful if pioneer species are used for planting
Pond zone	Hydrological connectivity	Pond floor	Planting pond floor + Hydro-connectivity
 0 1 2 3 4	0* 1[e] 2 3 4 ✓ Successful if natural stands are present upstream/upwind from the pond zone	0 1 2 3 4 X Pond bunds 0* 1 2 3 4 ✓	0 1 2 3 4 ~ Not investigated in this study, but planting is unnecessary when hydrology is restored, unless propagules are limited[e]

Figure 4-8: *Summary of the effectiveness of different restoration methods (EMR, planting, and a combination of both) to overcome thresholds of establishment in the coastal zone and the pond zone (red: ineffective, green: effective, yellow: condition already met in baseline situation), based on the findings of this study (*) and literature (**a:** (Cado et al., 2021), **b:** (Winterwerp et al., 2020), **c:** (Van Cuong et al., 2015), **d:** (Saenger and Siddiqi, 1993), **e:** e.g. (Lewis III, 2005; Matsui et al., 2010; Proisy et al., 2018; van Bijsterveldt et al., 2020))*

For instance, season optimized sluice gate management could ensure that sufficient propagules enter target sites in the pond zone, if seasonality of propagule release and wind direction are taken into account. Additional measures, such as pond partitioning and localized pond-floor raising (R. H Bosma *et al.*, 2020), would of course be necessary to create habitat for the propagules to land, and eventually realize mixed-mangrove aquaculture or greenbelt-rimmed ponds along rivers and creeks.

4.5.4 Mangrove restoration in the coastal zone

4.5.5 Q1.b Which mangrove recovery process is dominant in the coastal zone?

Our GIS study revealed that mangrove expansion in the coastal zone could be attributed mostly to dispersion of pioneer species, and only for a small extent to planting efforts (*Figure 4-3*). We know from frequent field visits that planting efforts in the coastal zone of Demak have been initiated by various organizations on a yearly or two-yearly basis. In most cases, this involved large-scale planting on exposed, partly low-lying coastal mudflats (e.g. *Figure 4-9*, white polygon), or at sites that were already colonized by pioneers (e.g. *Figure 4-9:* 2013, 2015, behind breakwater). The use of natural recruits as an indicator for a sites' suitability for mangrove planting in the coastal zone is a common practice in community-based restoration efforts (Wodehouse and Rayment, 2019), but has also been reported to be unnecessary 70% of the time, as mangroves are already colonizing those sites naturally (Wodehouse and Rayment, 2019). In the case of Demak's breakwater, the practice has led to a mangrove stand of extremely dense and stunted *R. mucronata* trees, with a few surviving *A. alba* trees that rise head and shoulders above the rest (*Figure 4-9*, top, 2017 and 2018). This illustrates that planting (especially of non-pioneer species) at already newly-colonized sites, is a poor practice.

Most of the large-scale plantations that were done at bed levels below the threshold of natural recruitment in Demak disappeared before our field visit in a following year. In literature, there have been examples of successful planting on unvegetated foreshores in the coastal zone (Saenger and Siddiqi, 1993; Uddin and Hossain, 2013). For example, a series of large-scale afforestation projects on newly accreted foreshores in Bangladesh successfully planted an area of 120 000 ha with several mangrove species in the 1980s, including pioneer species such as *Sonneratia apetela* and *Avicennia officinalis* (Saenger and Siddiqi, 1993). The elevated foreshores on which the mangroves were planted were subject to complete desiccation during dry seasons and would normally have been colonized by salt tolerant grasses. One reason for the reported high survival rates of 52% might therefore be that the supply of propagules from pioneer species was the limiting factor for natural mangrove expansion in these projects. These examples suggest that planting, as the sole restoration method, in the coastal zone is only useful if it is done to reintroduce species that are moreover suitable to colonize the site (*Figure 4-8*). In Demak, natural mangrove cover showed a linear increase on the satellite images, indicating that propagule limitation of pioneer species was not an issue.

Figure 4-9: *Top: Bird's-eye view of the breakwater seaward of Wonerejo, Demak in 2017 (drone image courtesy of CoREM-UNDIP). The white arrow indicates the direction of the picture time series. The time series show the development of a mudflat after construction of the breakwater in 2012, followed by natural colonization in 2013 and subsequent planting of R. mucronata in 2014. Note the relative height difference in all pictures of the naturally colonized A. alba trees (black arrows) versus the surrounding planted R. mucronata trees (2015-2018).*

However, the cover of natural mangroves seemed to stabilize in recent years (*Figure 4-3*), suggesting that most of the suitable mangrove habitat has now been occupied.

4.5.6 Q3. EMR-structures: Can EMR-structures induce natural mangrove recovery at coastal sites?

The aimed-for effect of the EMR-structures in our study area was to expand the existing mangrove habitat by means of wave attenuation, so that the resulting sediment deposition would raise the bed level above the threshold for natural colonization. This method has been effective along several eroding mangrove mud-coasts around the globe (Winterwerp et al., 2020; *Figure 4-8*). However, along the subsiding coastline of Demak, EMR with the use of permeable structures did not result in large-scale mangrove establishment over the years that the plots were monitored, even though a few seedlings were able to colonize the EMR-sites. It is possible that seedling establishment did occur at a larger scale at the EMR sites, but that the seedlings had disappeared by the time we assessed the plots. This would be in accordance with the findings of Cado et al. (2021), who found that there was abundant seedling establishment behind the structures, but that these seedlings did not persist longer than a few months and did not grow into mature mangroves. This rapid mortality of seedlings could reportedly be explained by a sequence of two processes: a) the permeable structures do elevate the bed level of the targeted coastal sites, especially during the wet season (Winterwerp *et al.*, 2020), but b) the whole area is subsequently subject to rapid land subsidence (Kuehn *et al.*, 2009). Hence, the freshly accumulated sediment might initially surpass the elevation threshold for seedling establishment during the wet season, but later sink below the threshold of survival, thus killing the young mangrove seedlings. EMR-structures as a restoration method alone therefore appear not to be enough to support mangrove recovery in this severely subsiding coastal area (*Figure 4-8*). For mangrove recovery to be successful in Demak, subsidence urgently needs to be addressed, especially by halting ground water extraction.

4.5.7 Q4. EMR-structures + planting: Can mangrove planting combined with EMR-structures accelerate recovery?

Despite the low success in natural recruitment at the EMR sites, EMR did increase the survival of established (planted) seedlings (*Figure 4-6*b), even at sites where natural recruitment behind the structures was not yet taking place (*Figure 4-7*a). This suggests that a combination of planting and EMR might be useful to overcome (temporary) thresholds of establishment and accelerate revegetation of challenging sites in the coastal zone. However, species choice appears an important factor in the development of the revegetated site after initial recruitment. *R. mucronata*, the species that is most often used in large-scale planting (Wodehouse and Rayment, 2019), proved to have the highest initial survival rate (*Figure 4-7*a), which might explain the popularity of these species in many voluntary planting projects (Primavera and Esteban, 2008; Wodehouse and Rayment, 2019). However, the slow growth rate of *R. mucronata,* such as observed at this muddy coast (*Figure 4-7*b), might lead to a mangrove stand that is more stunted and/or less regenerative than when a pioneer species appropriate to this site would have been used. This outcome has indeed been observed in various studies (e.g. Barnuevo et al., 2017; Fickert, 2020; Proisy et al., 2018), as well as behind the breakwater in Demak (*Figure 4-9*). It should be noted that in carbonate systems, *Rhizophora* species have been observed as pioneers of the coastal zone (e.g. (McKee, 1993; Piou *et al.*, 2006; Prabakaran *et al.*, 2021)). This illustrates that it is important to look at natural example sites with a similar biophysical typology as the restoration site (e.g. minerogenic versus organogenic soil (Worthington *et al.*, 2020)), when selecting an appropriate pioneer species. The presence of a few fruiting pioneers can reportedly revegetate a site more efficiently than a planting effort with better surviving, but slower growing species (Fickert, 2020). Although we were not able to include formerly native pioneers such as *Sonneratia alba* and *Aegialitis annulata* (Balun, 2011; Ilman *et al.*, 2016), our Q4 experiment indeed demonstrated that planting with pioneers can quickly lead to the presence of a few reproductive young trees. This illustrates that the trade-off between a species' survival and growth rate should be taken into consideration when choosing species to plant in combination with EMR-measures, preferably using pioneer species at newly accreted sites (*Figure 4-8*).

4.6 Conclusions and management implications

Although the coastline of Demak is a rather extreme case of erosion and land-subsidence, this study does give some useful insights regarding when and how planting and EMR are likely (un)successful. Through a combination of GIS, propagule monitoring, and planting experiments, our results show that: (1) Even in areas with mainly active aquaculture, there is potential for ecological mangrove restoration in ponds lining creeks and rivers, as long as there is a propagule source upstream / upwind. EMR measures in active ponds along rivers could include sluice gate management during the most optimal season (in terms of wind and propagule availability), in combination with partitioning of the pond by adding an inner dike to the river-side of the pond and allowing that part to fill up with sediment so that pioneers can colonize (i. e. greenbelt ponds). Planting may be done on pond bunds or at locations where propagule supply of the target species is limited.

(2) Seaward mangrove expansion through planting without additional EMR measures to restore mangrove habitat is likely to be unsuccessful (at low-lying sites) or unneces-

sary (at proper elevations with existing seedlings), unless species need to be (re)introduced. EMR measures to restore mangrove habitat through sediment trapping along eroding mud-coasts could include placement of permeable (brushwood) structures. Along subsiding coasts, these structures may however not be enough to maintain the bed level above the threshold for mangrove establishment. However, mangrove planting with fast growing pioneer species in combination with sediment trapping by EMR-structures could accelerate mangrove recovery at such sites.

4.7 Acknowledgements

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4.8 Appendices

Table A4-1: *High resolution images of the study area in the years of interest that were downloaded from Google Earth for the GIS analysis of mangrove cover change over time. The image type "mosaic" indicates that only a combination of two satellite images was available for the study area in that year.*

Year of interest	Image type	Acquisition date	Image source
2005	mosaic	22-4-2005, 31-5-2003	Maxar Technologies
2010	complete	27-6-2010	Maxar Technologies
2013	mosaic	11-10-2013, 9-9-2013	Maxar Technologies
2014	complete	1-7-2014	Maxar Technologies
2015	complete	13-6-2015	Maxar Technologies
2016	complete	13-9-2016	Maxar Technologies
2017	complete	18-9-2017	CNES / Airbus
2018	complete	9-9-2018	Maxar Technologies

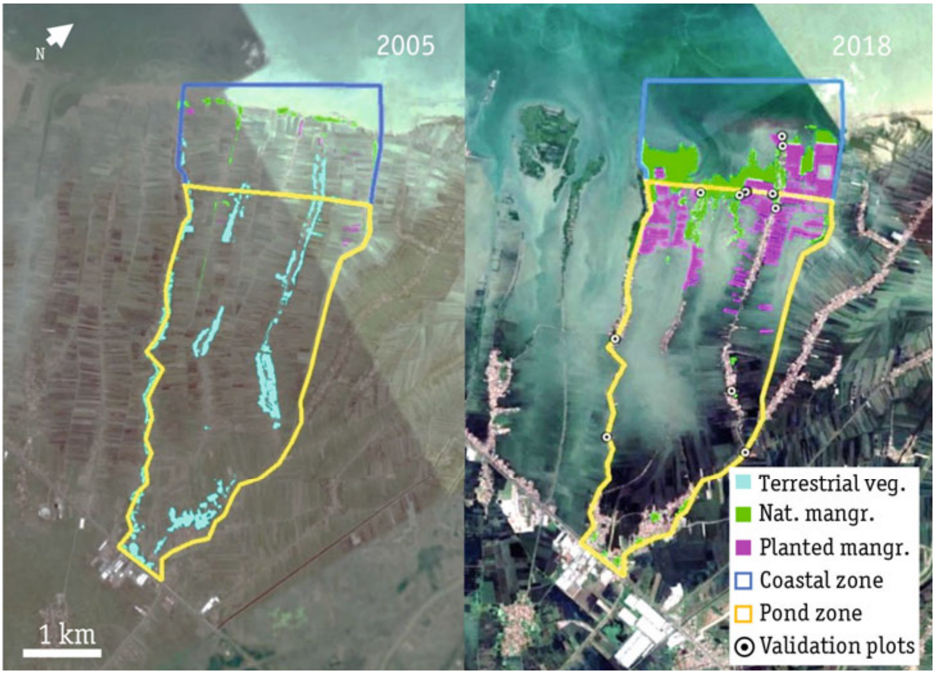

Figure A4-1: *Digitized vegetation types in the coastal zone and pond zone in 2005 and 2018. Locations of validation plots inside the zones of interest from the GIS study (i.e. only those inside the coastal and pond zone not the fresh-water reference zone for 2015) are indicated in the map of 2018.*

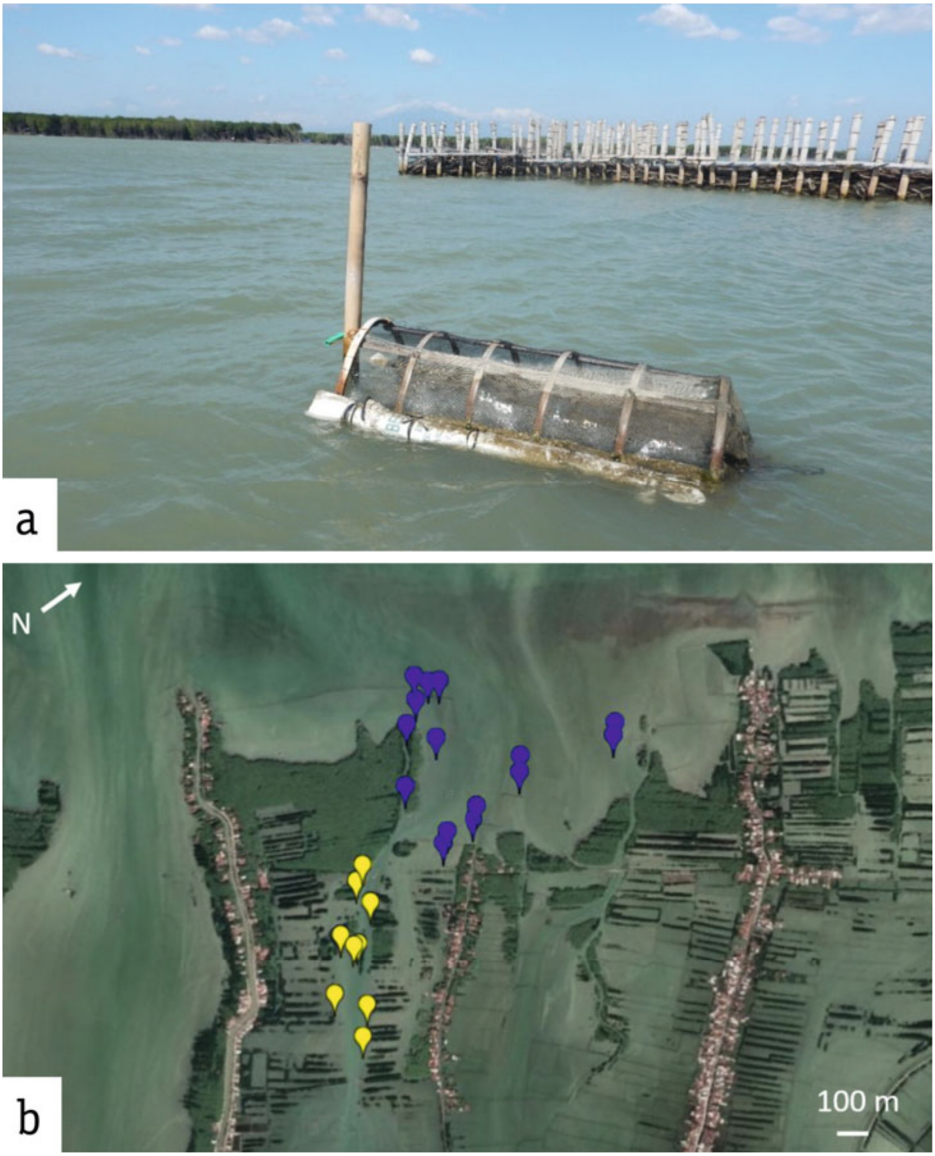

Figure A4-2: ***a.*** *Floating propagule trap, loosely attached to an anchoring pole so that the opening (on the left) was always facing the current and the trap continuously captured floating debris (including propagules).* ***b.*** *Locations of the propagule traps and seedling monitoring sites throughout the pond zone (yellow marks) and coastal zone (blue marks).*

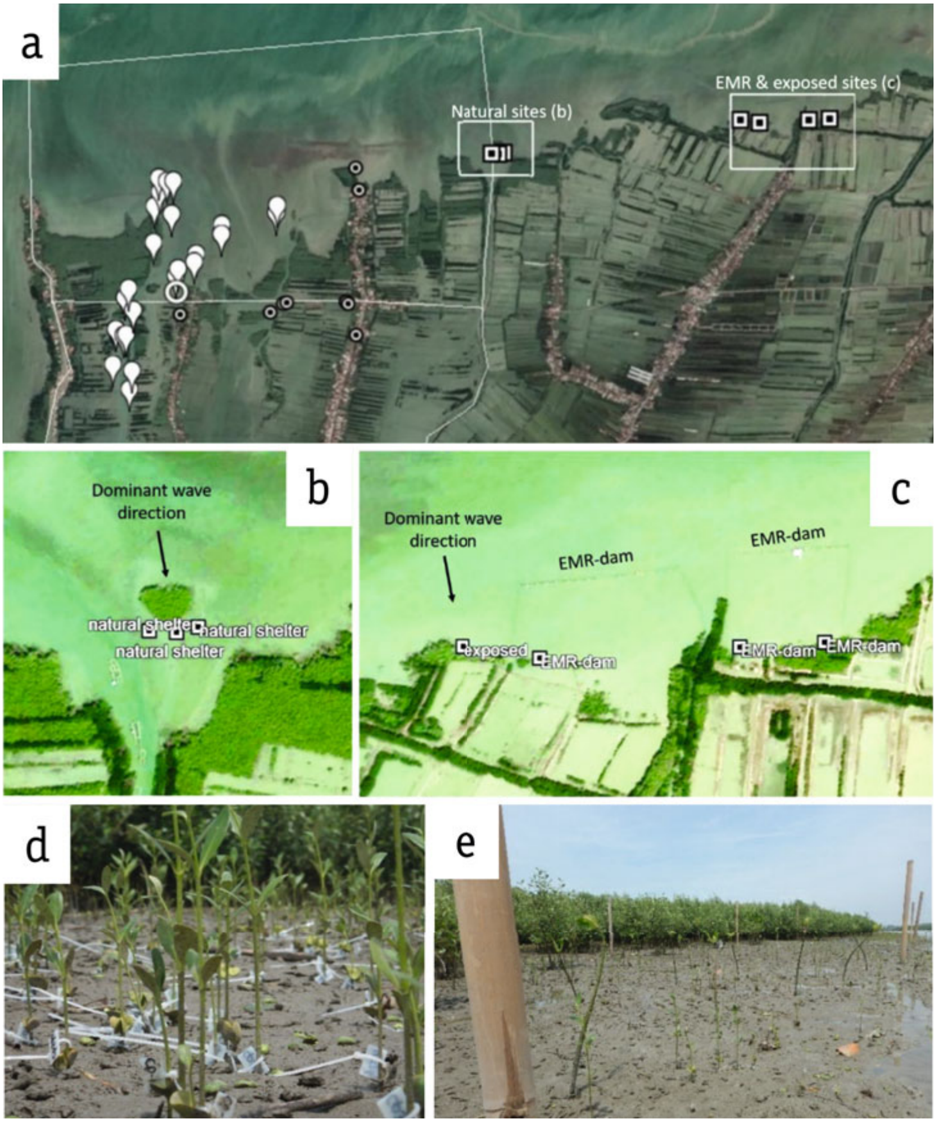

Figure A4-3: a. *EMR experiment locations (white rectangles) relative to the larger research area.* ***b****. Locations of the three blocks behind the natural shelter in Wonerejo.* ***c.*** *Locations of the three blocks behind the artificial shelters (EMR sites) and the exposed block in Surodadi.* ***d.*** *Labelled naturally recruited seedlings in the control plot of the block represented in e.* ***e****. Picture of a block with three plots behind a vegetated chenier (natural shelter).*

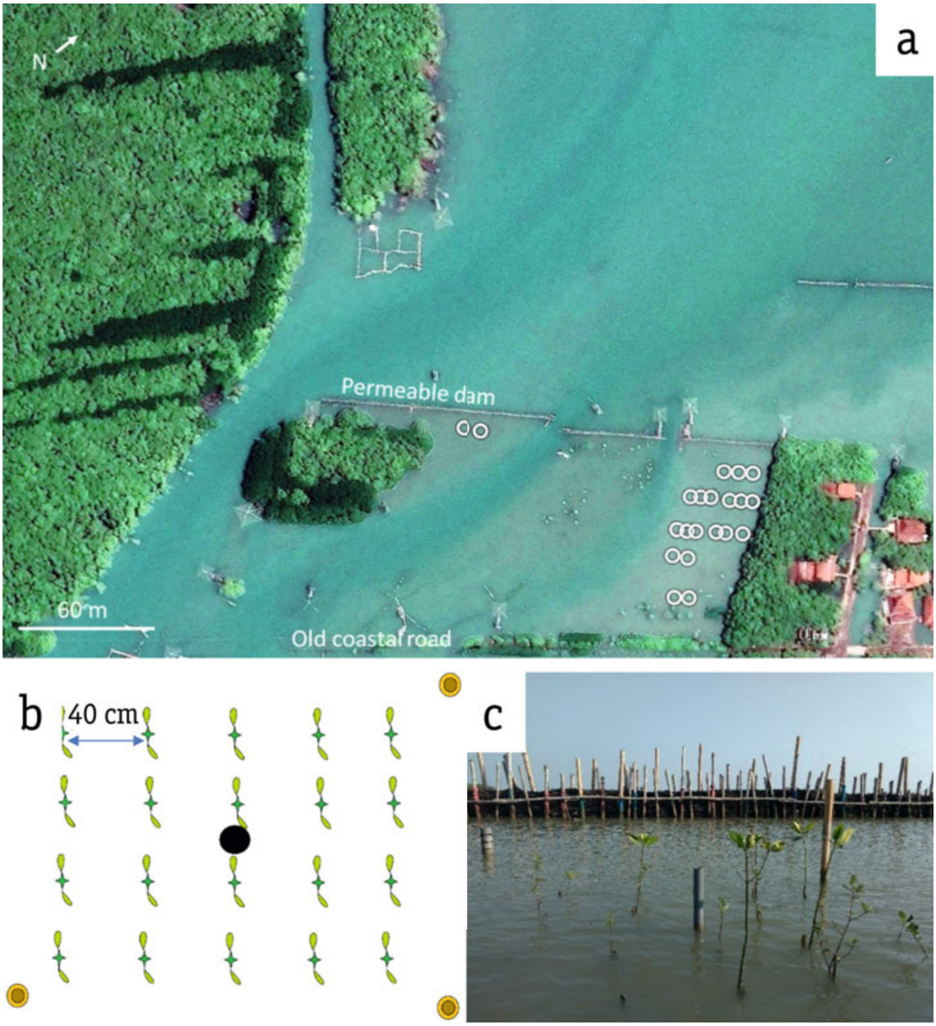

Figure A4-4: a. *Distribution of the 20 plots of the planting experiment behind the oldest permeable structure in the area.* ***b.*** *Schematic drawing of 5*4 planting matrix in each plot.* ***c.*** *Picture of a plot with the four planted species and a sedimentation pole in the centre.*

Table A4-2: *Forest structure in terms of absolute tree density (n ha^{-1}), basal area (m^2 ha^{-1}) and diameter at breast height (cm) in the validation plots of the natural and planted stands. The relative occurrence of each species based on the absolute density, and the relative dominance of each species based on basal area are indicated in brackets.*

Stand type	Species	Absolute density (n ha^{-1}) & (relative, %)	Basal area (m^2 ha^{-1}) & (dominance, %)	DBH (cm)
Natural	*Avicennia marina*	11783 ± 2909 (91.7)	14.3 ± 2.81 (91.1)	3.61 ± 0.12
n = 5	*Avicennia alba*	490 ± 279 (3.8)	0.36 ± 0.32 (2.3)	2.67 ± 0.49
	Rhizophora mucronata	576 ± 217 (4.5)	1.03 ± 0.56 (6.6)	3.58 ± 1.12
	Sum	12849 (100)	15.69 (100)	
Planted	*Avicennia marina*	211 ± NA (1.4)	0.33 ± NA (2.4)	4 ± 2
n = 4	*Avicennia alba*	556 ± NA (3.6)	0.83 ± NA (6.1)	4.33 ± 0.33
	Rhizophora mucronata	14648 ± 5034 (95)	12.4 ± 3.7 (91.5)	3.27 ± 0.08
	Sum	15425 (100)	13.56 (100)	

The effect of plastic on mangrove survival

CHAPTER 5

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Does plastic waste kill mangroves? - A field experiment to assess the impact of anthropogenic waste on mangrove growth, stress response and survival.

5.1 Abstract

The value of mangroves has been widely acknowledged, but mangrove forests continue to decline due to numerous anthropogenic stressors. The impact of plastic waste is however poorly known, even though the amount of plastic litter is the largest in the region where mangroves are declining the fastest: South East Asia. In this study, we examine the extent of the plastic waste problem in mangroves along the north coast of Java, Indonesia. First, we investigate how much of the forest floor is covered by plastic in the field (in number of items per m^2 and in percentage of the forest floor covered by plastic), and if plastic is also buried in the upper layers of the sediment. We then experimentally investigate the effects of a range of plastic cover percentages (0%, 50% and 100%) on root growth, stress response of the tree and tree survival over a period of six weeks. Field monitoring showed that plastic was abundant, with 27 plastic items per m2 on average, covering up to 50% of the forest floor at multiple locations. Moreover, core data revealed that plastic was frequently buried in the upper layers of the sediment where it becomes immobile and can create prolonged anoxic conditions. Our experiment subsequently revealed that prolonged suffocation by plastic caused immediate pneumatophore growth and potential leaf loss. However, trees in the 50%-plastic cover treatment proved surprisingly resilient and were able to maintain their canopy over the course of the experiment, whereas trees in the 100%-plastic cover treatment had a significantly decreased leaf area index and survival by the end of the experiment. Our findings demonstrate that mangrove trees are relatively resilient to partial burial by plastic waste. However, mangrove stands are likely to deteriorate eventually if plastic continues to accumulate.

5.2 Introduction

Even though mangroves are widely valued for the ecosystem services they provide, forests are rapidly declining despite restoration efforts. Mangrove forests provide multiple provisioning, regulating and recreational ecosystem services, but they are most valued for their role in coastal protection (Barbier *et al.*, 2011). In spite of this global recognition of their importance, anthropogenic influences such as land-use change, cause continued mangrove decline worldwide (Alongi, 2002; Thomas *et al.*, 2017). Many restoration and conservation projects have been initiated in recent decades to restore mangrove forests and prevent further loss (Bayraktarov et al., 2016; A. M. Ellison, 2000; Narayan et al., 2016). However, few mangrove restoration projects have been able to achieve stable mangrove canopies (e.g., A. M. Ellison, 2000; Kodikara et al., 2017; Primavera & Esteban, 2008). The lack of system understanding has been the reported cause of various failed restoration projects (Ellison, 2000; Primavera and Esteban, 2008), and achieving such an understanding of the local hydrodynamics and sediment balance has increasingly been recommended to improve mangrove settlement success (eg. Balke & Friess, 2016; Lewis III, 2005). Getting these boundary conditions right has proven to be successful for mangrove seedling planting and even natural mangrove settlement (eg. Lewis III, 2005; Van Cuong, Brown, To, & Hockings, 2015). However, less attention has been devoted to the more-direct anthropogenic stressors that could hamper the growth and survival of the restored young trees. The most notable anthropogenic pollution that might stress mangroves is plastic waste (Smith, 2012).

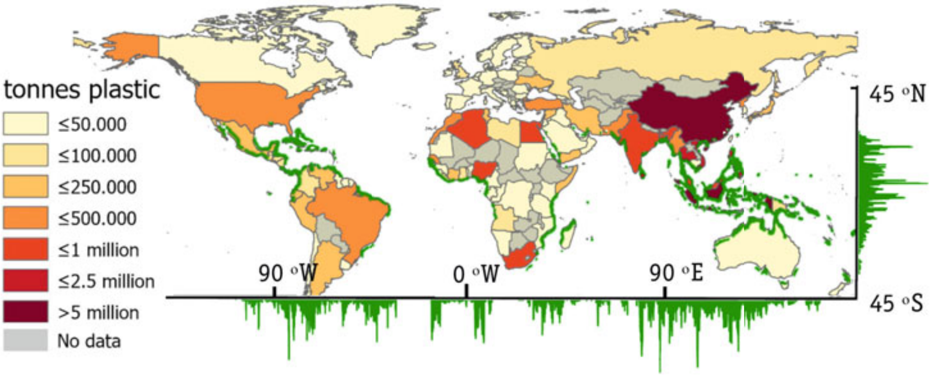

Figure 5-1: *A global map overlay of the estimated mass of mismanaged plastic waste (metric tonnes (MT)) at risk of ending up in the ocean in 2010 from Jambeck et al. (2015) "Plastic waste inputs from land into the ocean" (reprinted with permission from AAAS), and the global mangrove frequency distribution in 2010 along longitude and latitude, courtesy of Bunting et al., (2018) "The Global Mangrove Watch—A New 2010 Global Baseline of Mangrove Extent", Illustrating that the plastic waste problem in the ocean is the largest in regions where mangroves are still the most abundant.*

Worldwide, regions with high mangrove cover often also have serious plastic management issues. Plastic waste entering the ocean due to a lack of waste collection and disposal services is estimated to be 4.8 to 12.7 million tonnes annually (Jambeck *et al.*, 2015). Two-thirds of the global plastic waste enters the ocean via the top 20 most polluted rivers, all situated in Asia (Lebreton *et al.*, 2017). Similarly, Jambeck et al., (2015) found that the top four countries listed, which add the most to the marine plastic waste problem, are China, Indonesia, Philippines and Vietnam (8.8, 3.2, 1.9, and 1.8 million tonnes/year respectively). All of these countries are situated in the general region, where one-third of the world's remaining mangrove cover is found (Bunting *et al.*, 2018)(Figure 5-1). This co-occurrence of large amounts of plastic waste and abundance of mangroves is potentially problematic. The majority of mangroves species possesses some type of aerial roots, which ensure that part of their root system remains exposed most of the tidal cycle (P Barry Tomlinson, 2016). All species rely on their aerial roots to oxygenate their root zone under periodic anoxic conditions. However, the species with upward pointing aerial roots (i.e. knee-roots and pneumatophores bearing species) are especially vulnerable to suffocation by smothering. Smothering by sediment and debris is a realistic threat as aerial roots cause flow reduction of water entering the swamp at high tide (Mazda *et al.*, 2006), which promotes accumulation of sediment and debris in the mangrove fringe (Horstman *et al.*, 2017). With mangrove roots being such efficient traps for particles and objects (e.g. Chen et al., 2018; Horstman et al., 2017; Martin, Almahasheer, & Duarte, 2019), much of the floating plastic debris in the region is bound to end up in mangrove forests, potentially smothering pneumatophores and knee-roots.

To date, few studies have been conducted on the extent of the plastic problem in mangroves, as most of the marine debris studies have focused on zones with more recreational value, such as beaches (eg. Ivar do Sul & Costa, 2007; Li, Tse, & Fok, 2016; Podolsky, 1989; Syakti et al., 2017; Willoughby, Sangkoyo, & Lakaseru, 1997). One of the studies on plastic in mangroves reports marine debris ridges up to 50 cm above the forest floor (Smith, 2012). Marine debris in mangroves is mostly comprised of plastic bags (Ivar do Sul and Costa, 2007; Cordeiro and Costa, 2010; Debrot *et al.*, 2013; Kantharajan *et al.*, 2018). If plastics remain stationary on the forest floor or inside the sediment over multiple tidal cycles (e.g., because of neap tide, or through burial in the sediment), they can create an anoxic environment and could thereby potentially induce tree suffocation (Smith, 2012). In particular, species that rely on upward pointing aerial roots for oxygen supply such as *Avicennia*, *Laguncularia* and *Sonneratia* spp. (McKee, 1996) could be at risk of suffocation caused by burial in plastic. However, despite multiple references to this potential effect in the literature (e.g., Sandilyan & Kathiresan, 2012; Smith, 2012), and some personal field observations of pneumatophore deformation due to plastic burial, to our knowledge, no manipulative studies have yet been conducted to support this hypothesis.

In this study, we aim to understand how the current plastic waste loads on the forest floor and in mangrove sediment might affect mangrove trees. We first assess the size of the current plastic waste problem in mangroves by quantifying the relatively mobile plastic fraction on the forest floor (the percentage of the forest floor covered by plastic, and the number of plastic items per m^2), and by quantifying the more immobile plastic fraction buried in the rhizospheric sediment of eight coastal mangrove fringes. We then investigate how various degrees of plastic loads affect mangrove trees in terms of growth, stress and survival through the experimental application of plastic to the root systems of Avicennia trees.

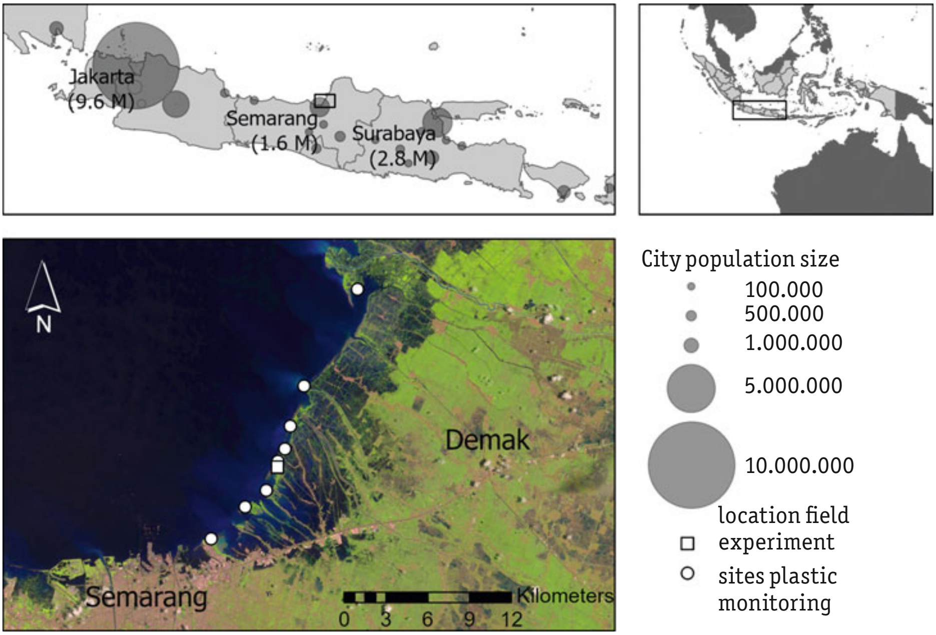

Figure 5-2: *Sentinel-2 satellite image of the study area, with sites of the plastic monitoring campaigns (white circles) and the location of the field experiment (white square) on the coast of Demak, located on the densely populated coast of Java, Indonesia. Population densities of Java's cities with more than 100.000 inhabitants are displayed, proportional to their population size in 2010 (demographic statistics database of the UN).*

5.3 Materials & Methods

5.3.1 Site description

We used the coastline of Demak regency in Central Java as a model site to study the effect of plastics on mangroves, as this area is exemplary for many densely populated mangrove areas found in South East Asia. Java is densely populated (Figure 5-2), and the northern coast holds several big cities (e.g., Jakarta, Semarang, Surabaya) that produce enormous amounts of plastic waste (Willoughby, Sangkoyo and Lakaseru, 1997; Syakti *et al.*, 2017; Maryono *et al.*, 2020). Household waste is largely managed by burial in landfills, but most rural places lack garbage collection services. Demak's coastline stretches along 20 km from Semarang city (1.8 million inhabitants (United Nations, 2018)) in northeast direction (Figure 5-2). The coastal area consists of wide plains with a gentle slope of alluvial deposits, intersected by a few large rivers (Wulan and Buyaran river) and hundreds of streams. The area used to be lined with mangrove forests that were kilometers wide. However, mangroves were replaced by paddy fields long ago, and the last remaining mangrove fringes were removed when shrimp farming was booming in the 1980s. The loss of mangrove forests, in combination with local land subsidence, has led to large scale erosion from 2003 onwards. Since the onset of coastal erosion, many aquaculture ponds have been abandoned, which has led to small mangrove stands (max 500 m wide) reappearing in old ponds along the coast (van Bijsterveldt *et al.*, 2020). The government is investing in mangrove restoration to expand these patches into a greenbelt, but restoration is slow and existing mangrove trees appear to be stressed. This is especially the case in the seaward fringe, where plastic and sediment accumulate, wave impacts are high and erosion is looming. Plastic waste from the villages in the coastal area is not collected by the local authorities, but is instead burned per household, or it is dumped locally in places that are washed out during the monsoon or by the tide. Many of the plastics that have been washed out to sea are subsequently trapped by the coastal mangrove stands. The plastics visibly accumulate in sediment ridges on the seaward edge of the mangrove stands, where the flow velocity of the entering water drops significantly due to the drag caused by prop roots and pneumatophores. Sediment ridges often migrate landward under rough weather conditions, thereby covering mangrove root zones in their path (Chappell and Grindrod, 1984), as the debris that accumulates often rises above the level of the pneumatophores. The trees that are situated inside such a sediment-debris ridge frequently appear stressed or dead (*Figure A 5-1*). However, sedimentation with coarse sediment alone is not likely to kill mangrove trees (Chappell and Grindrod, 1984; Okello *et al.*, 2014). The fact that these sediment ridges consist of both coarse sediment and multiple plastic layers, could potentially explain the tree mortality observed in the field. In order to investigate the nature and the effect of plastic waste in mangroves, we conducted a monitoring campaign and a field experiment.

5.3.2 Plastic monitoring

To assess the amount of plastic present in the field that could pose a threat to mangroves by covering pneumatophores, we quantified the presence of plastic with a size larger than 1 cm (the diameter of a pneumatophore) in two fractions: (1) the amount of relatively mobile plastic covering the forest floor, measured in terms of “the percentage of forest floor covered by plastic waste”, and “the number of items per m^2 of forest floor”. (2) The amount and the burial depth of immobilized plastic that is trapped in the upper layers of the mangrove sediment. This situation is especially expected to occur in the mangroves fringing the sea, where sediment and debris are deposited by the tide and waves. The mobile and immobile plastic fractions were monitored in three zones across eight mangrove fringes during a series of field campaigns in 2018 and 2019. Eight mangrove stands were selected as replicates for this study, all of which consisted primarily of *Avicennia* spp. (*Avicennia alba* and *Avicennia marina*). The mangrove stands were located along a 20 km coastline stretch of the Demak district in Central Java, Indonesia (Figure 5-2). An additional site, situated 100 km westward along the coast without a large city (Semarang) in its direct vicinity, was also added to the monitoring campaign. The mobile plastic fraction, assessed in terms of the percentage of forest floor covered by plastic and the number of plastic items per m^2, was monitored using 50*50 cm quadrats in three different zones per site. Namely, the seaward edge of the sediment ridge in the mangrove fringe, the landward side of the sediment ridge and the mangrove basin landward of the ridge (Figure 5-3a). Quadrat data were collected three times: after the wet season of 2017-2018, after the dry season of 2018 and finally after the dry season of 2019. The percentage plastic cover within each 50*50 cm quadrat was estimated and the number of plastic items in the first 2 cm of sediment was counted. The percentage plastic cover per quadrat was always estimated by the same researcher to ensure consistency. In addition, a validation dataset was obtained by taking pictures of the field quadrats. The pictures in which plastic was clearly identifiable -in practice these were sandy sites where canopy shadow and biofouling of plastic was limited - were subsequently used to manually digitize all visible plastic per quadrat with image processing software (Fiji ImageJ, version 1.52u). The quantified plastic area was then divided by the surface area of the quadrat and multiplied by 100% to obtain the actual plastic percentage per quadrat. The field estimates validated against the plastic percentages obtained from pictures of those quadrats revealed a linear relationship (R^2: 0.88, $p<0.0001$), indicating that the field estimates were perhaps not always 100 % accurate, but at least reproducible across field sites. Thus, the estimated percentage of the forest floor covered by plastic and the number of plastic items were monitored per quadrat in each zone, replicated over nine sites and repeated during three field campaigns, resulting in 27 quadrat observations per zone and amounting to a total of 81 quadrat assessments.

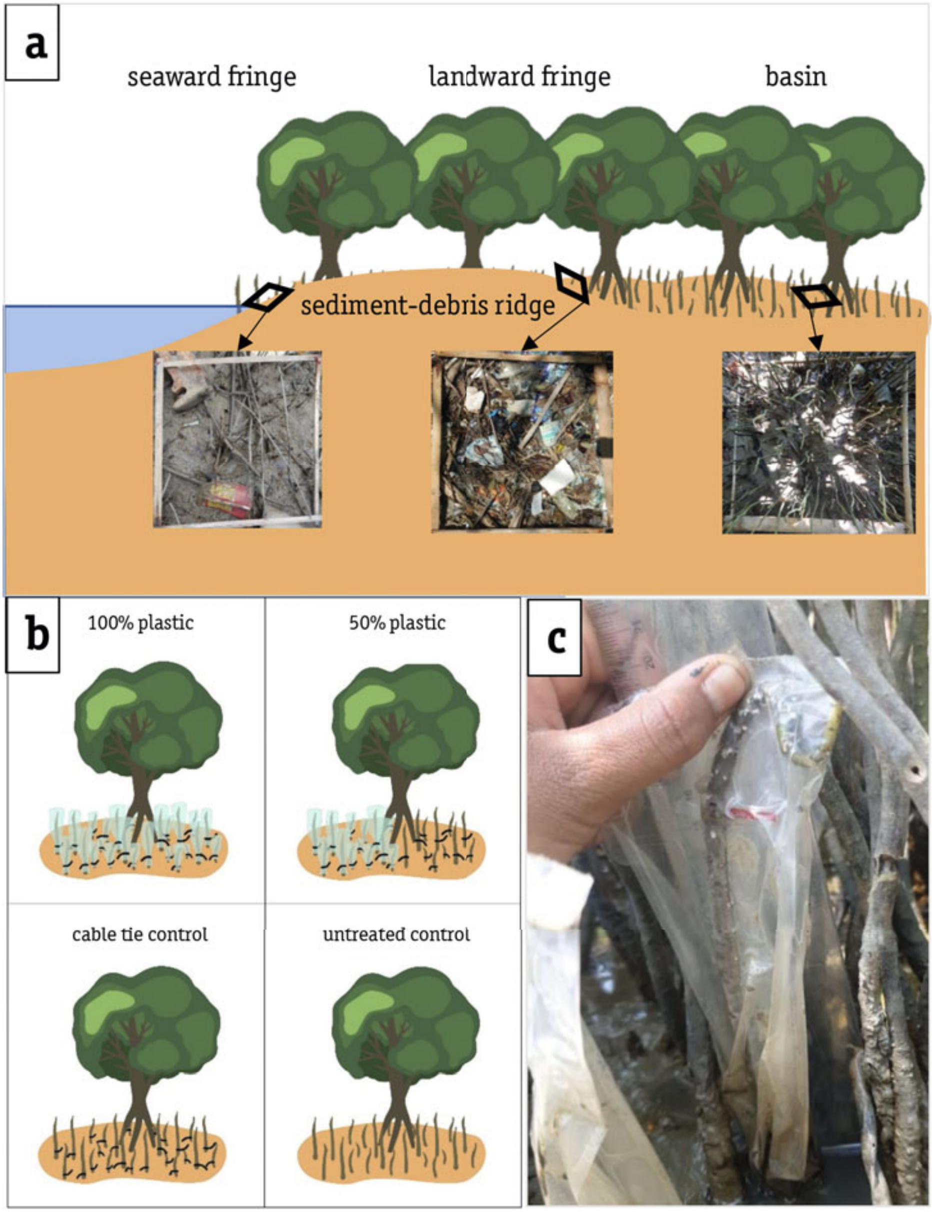

Figure 5-3: a. *Quadrat placement and core locations across a coastal mangrove stand: water line at low tide (seaward fringe), landward of the sediment ridge (landward fringe), mangrove basin (basin).* ***b***. *Conceptual representation of the four plastic suffocation experiment treatments: 100% of the root zone covered by plastic, 50% of the root zone covered by plastic, cable tie control and untreated control (UTC).* ***c.*** *Plastic suffocation method, a plastic bag tied to the base of individual pneumatophores with a cable tie. Additional cable tie markings were used on 10 pneumatophores per tree at five cm from the top (the red cable tie inside the plastic bag in the picture) to quantify pneumatophore extension over the course of the experiment.*

The immobile plastic fraction that was buried inside the sediment was monitored by collecting sediment cores in each zone (*Figure 5-3*a). In total, 9 cores were collected per zone, divided over nine sites and repeated during two field campaigns resulting in 18 cores per zone and 54 cores in total. Cores were collected using a 50 cm transparent PVC tube (Ø 5.3 cm) with a saw head. Coring was done by slowly rotating the saw head into the sediment, cutting through plastic layers instead of pushing them down. The number of plastic layers and the depth of the plastic layers were quantified in each core. The coring depth varied among locations between a depth of 14 cm and 35 cm, due to differences in sediment type or cable root density. Therefore, to standardize across locations, only plastic layers detected within the first 14 cm of the sediment were considered for comparison between the zones.

5.3.3 Suffocation experiment

To investigate the effect of plastic waste on mangrove trees with pneumatophores, we conducted a manipulative experiment on 42 young *A. marina* trees to quantify root growth, litter fall and tree survival in response to various degrees of plastic suffocation over a period of six weeks. The trees that were selected for the experiment were located on the seaward fringe of one mature mangrove stand. This site had been selected because it was an expanding mangrove forest, and the young trees that relatively recently colonized the mudflat were still largely disconnected from each other. This was essential in order to apply treatments to individual trees. The downside of using the seaward fringe of the mangrove stand was that i) the trees were relatively exposed to waves (*Figure A 5-3*) and ii) the trees showed some size and age differences, with larger trees towards the back and smaller trees out on the mudflat. The 42 trees that were selected for the experiment were disconnected from other individuals and were relatively similar in size (mean height: 1.8 m (+/- 0.4 m)). To overcome potential bias in tree mortality caused by confounding variables such as salinity, waves or size rather than by the treatments, the 42 trees were assigned to the four plastic treatments based on their height in such a way that all treatments had the same variation of tree size and tree position relative to the sea. The trees were then subjected to one of the four treatments: 100% of the pneumatophores covered by plastic (13 trees), 50% of the root zone covered by plastic (11 trees), an untreated control (10 trees) and a cable tie control (8 trees) (Figure 5-3b). In the two plastic cover treatments, trees received a plastic bag on individual pneumatophores, tied to the bottom of each root with a cable tie (Figure 5-3c). This method of plastic application does not fully resemble the way pneumatophores are smothered by both sediment and plastic, as observed in the field. However, preliminary experiments revealed that this was the most reliable method to keep the treatments reproducibly in place over longer periods of time. The effect of the cable tie fixation method was accounted for in the cable tie control treatment.

Trees in the 100% treatment received plastic bags on 100% of the pneumatophores in their root zone. Trees in the 50% treatment received plastic bags on 50% of their aerial roots (Figure 5-3c). The 50% treatment was applied to pneumatophores in the root zone in the shape of a semi-circle to mimic the effect of partial suffocation by a migrating sediment-ridge. The effect of different amounts of plastic cover was assessed in terms of (1) pneumatophore growth (new pneumatophore development and pneumatophore extension), (2) tree stress (litter fall and leaf area index) and (3) tree survival.

(1) Root growth was assessed based on two variables: the number of new pneumatophores that were formed over the course of the experiment and the extension of existing pneumatophores. New pneumatophore growth was assessed only at the end of the experiment by counting the number of bright green pneumatophores (at low tide) that were smaller than 5 cm. Root extension of existing pneumatophores was quantified by measuring the increase in distance between a cable tie applied 5 cm from the top of a pneumatophore at baseline, and the distance from the same cable tie to the tip of the pneumatophore six weeks after treatment application. This method was preferred over measuring the total length of pneumatophores at baseline and at the end of the experiment because our preliminary experiments revealed that vertical pneumatophore growth occurs primarily from the root apex. In addition, this root growth monitoring method had the advantage that large changes in sediment level between baseline and the end of the experiment could not influence the measurements in root growth. The cable tie method was applied to 10 randomly selected pneumatophores per tree (red circle in Figure 5-3c). The 50% treatment received cable tie markings on 20 pneumatophores, 10 in the plastic treated part of the root zone, and 10 in the uncovered part of the root zone.

(2) Tree stress was quantified by two independent methods: weekly litter fall underneath each individual tree, and the leaf area index (LAI) at the end of the experiment. Litter fall was quantified by collecting leaves weekly from nets suspended underneath each individual tree, and subsequently weighing the dry biomass (dried in a stove for 48 hours at 60 °C) in the lab. Litter fall data from the third week after treatment application were excluded from the analysis because a storm during that week has likely blown many of the leaves from the nets. Leaf area index per tree was quantified by remote sensing at the end of the experiment. That is, all trees were mapped with a drone (DJI Phantom 4) six weeks after the start of the experiment from an altitude of 50 m. Individual drone images were stitched to one mosaic image of the study area with a 10 cm resolution. The original tree canopy was then digitized by hand, following the contour of branches, twigs and the suspended leaf litter net. Healthy vegetation was subsequently extracted from the original image bands (RGB) using the Normalized

Green Red Difference Index (NGRDI) (Lussem *et al.*, 2018) with a threshold of 0. Overlay analysis of the vegetation extraction with the original canopy polygons then resulted in a LAI value per tree at the moment of final harvest.

(3) Tree survival until the end of the experiment was assessed as a response variable, and compared to the tree survival in the untreated control group. Tree survival in the untreated control group was used as the background survival rate, because tree mortality during the experiment would occasionally occur as result of rough weather conditions. This effect was accounted for by testing the survival rates in the 100%-, 50%- and cable tie treatments against the survival in the untreated control.

5.3.4 Statistical analysis

The distribution of plastic monitored in the field across the three zones of interest was analysed by Kruskal-Wallis non-parametric test for each of the plastic fractions (percentage plastic cover, number of items per m^2, burial depth and number of layers per core). The difference between treatments in terms of root extension was tested by Analysis of Variance (ANOVA), using tree ID as a blocking factor and plastic treatment as the explanatory factor. The effect of the four treatments on the number of new roots was tested with a generalized linear model (GLM) assuming a quasipoisson distribution. The effect of the plastic treatments on weekly litter fall were also analysed by GLM, using treatment and the interaction between treatment and week number as explanatory variables and litter fall per week as response variable assuming a Gamma distribution. Leaf area index, as a proportion, was highly skewed and the effect of the treatments on the leaf area index were therefore analysed with the Kruskal-Wallis non-parametric test. The difference in tree survival and expected natural survival for each treatment was tested statistically with a one-sided binomial test using the survival rate of the untreated control group as the expected natural survival. Statistics were executed in R studio (R Core Team (2013), version 1.0.143).

5.4 Results

5.4.1 Plastic monitoring

The number of plastic items on the forest floor was generally high, ranging from 0 to 236 plastic items per m^2, with an average of 27 items. The number of items differed significantly between the three defined sedimentation zones in mangroves, with significantly higher numbers found in the landward fringe (mean ± SE : 44.8 ± 9.3 number of items) than in the seaward fringe (29.4 ± 8.6 SE) and in the mangrove basin (17.3 ± 4.0 SE) ($p < 0.05$). Surprisingly, no significant effect of season was detected on the number of plastic items in the quadrats. Estimated percentage of the forest floor covered by plastic in Demak's mangroves varied between 0 and 75%, with an average of 17% of the forest floor covered by plastic. However, in contrast to the number of plastic items, no significant differences were found across mangrove zones or between seasons for the percentage of forest floor covered by plastic (Figure 5-4a). Furthermore, cores from the different zones across mangrove stands revealed that there was an immobilized plastic fraction. When considering all data from the cores, plastic was found buried in the sediment up to at least 35 cm depth (Figure A 5-2), which was the maximum core depth. However, due to a large variation in coring depth, only the plastic layers up to 14 cm were taken into account for a comparison between the zones. This comparison showed that plastic layers were present in the upper 14 cm of the sediment in all zones, and no distinction could be made between the zones in terms of average plastic layer depth or the number of plastic layers up to 14 cm (Figure 5-4b).

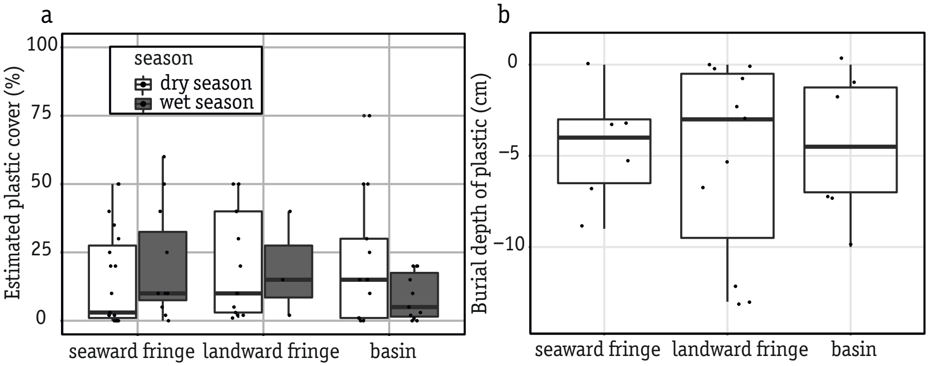

Figure 5-4: *Median (black line within box), 1st and 3rd quantile (box range) and 95% of the observation range (whiskers) of* **a.** *the estimated percentage of the forest floor covered by plastic per zone after the wet season and after the dry season* **b**. *the depth of plastic layers present in the upper 14 cm of the sediment across mangrove zones (seaward fringe, landward fringe and basin).*

5.4.2 Mangrove response to plastic suffocation

Simulation of different plastic cover percentages revealed that mangroves in all treatments developed new roots, with an average (± SE) number of new roots of 73 (± 25), 87 (± 14), 48 (± 15) and 100 (±38) for 100%, 50%, cable tie control and untreated control treatments, respectively, and with no significant differences in root development between the treatments ($p < 0.5$). In contrast, trees did invest significantly in root growth of existing pneumatophores in response to plastic suffocation ($F = 9.4$, df = 4, $p < 0.001$). Trees that had 100% of their root zone covered by plastic showed significantly more pneumatophore extension than trees in the cable tie and untreated control treatments. Pneumatophore growth in the 100% plastic treatment was 6.1 cm ($p < 0.001$), 4.9 cm ($p < 0.05$) and 5.2 cm ($p < 0.001$) higher than uncovered roots in the 50%, cable tie control and untreated control treatments, respectively, over a period of six weeks.

Interestingly, trees that had 50% of their root zone covered with plastic showed a similar growth response, though only in the roots that were covered by plastic, indicating a very localized response to plastic suffocation (Figure 5-5a). In the plastic covered part of the 50%-treatment, roots showed an average pneumatophore extension that was 6.2 cm ($p < 0.001$), 5.0 cm ($p < 0.01$) and 5.3 cm ($p < 0.001$) higher than the root growth in the uncovered half of the 50% treatment, cable tie control and untreated control, respectively, over a period of six weeks.

In addition to the response in root growth, trees also showed other signs of stress in response to the plastic treatments. The average daily litter fall trends of the 100%- and 50%-treatments appear to start higher than the cable tie and untreated control treatment over the first two weeks of the experiment (Figure 5-5b), suggesting an immediate stress response to suffocation. The litter fall in the 100%-plastic treatment then showed a strong drop between week 2 and 4, following a storm in the third week of the experiment (depicted on the second y-axis of Figure 5-5b). Trees in the cable tie and untreated control treatments showed a similar drop in litter fall after the storm, which suggests that the storm removed all loosely attached leaves. In contrast, litter fall in the 50%-treatment remained stable over the same period, and only started to drop after week 4. By the end of the experiment, only the 100% treatment and cable tie treatments showed a reduced LAI as a result of the litter fall, as displayed in *Figure 5-5*c, suggesting that the cable tie fixation method itself may also have induced a stress response. Interestingly, the 50% treatment did not have a significantly lower LAI than the untreated control by the end of the experiment (Figure 5-5c), despite showing high litter fall rates throughout the experiment in general. Overall, these trend differences between the litter fall and LAI data suggest that these stress parameters are less obvious response indicators of plastic suffocation than root extension.

Ultimately, the stress applied to the trees by the plastic treatments resulted in higher mortality rates (Figure 5-5d). The trees in the 100% plastic cover treatment suffered significantly from the plastic and showed a significantly lower survival than the trees in the untreated control treatment ($p = 0.02$). Trees in the 50% treatment also appeared to be mortally affected by the plastic, with slightly over half of the trees surviving the experiment. However, this was not found to be significantly different from the expected survival ($p = 0.2$). The expected survival was based on the survival of the trees in the untreated control treatment over the course of the experiment. The survival of these trees was relatively low as well (70%), which is most likely the result of a storm in the third week. Nevertheless, plastic clearly affected tree mortality on top of the storm.

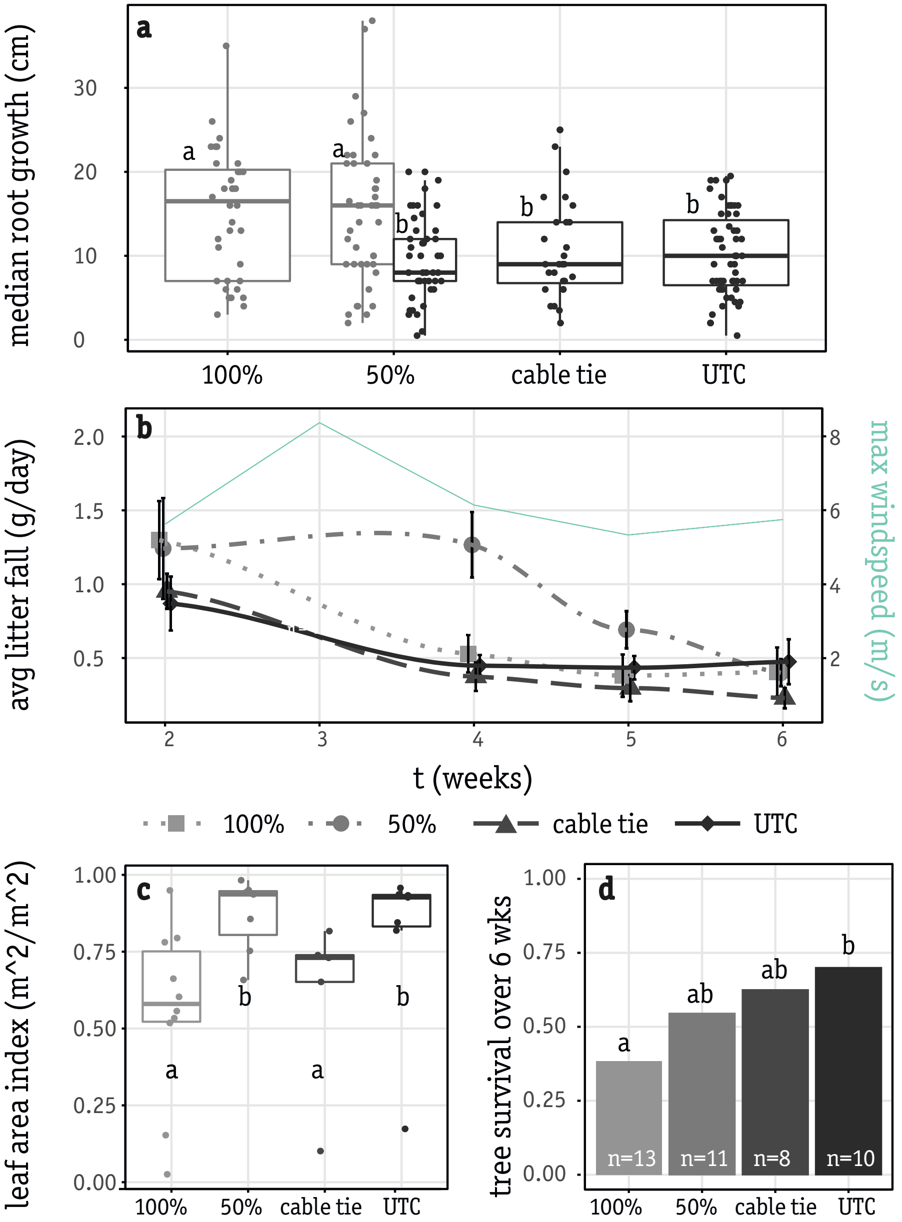

***Figure 5-5: a**. Median (±95% obs range) root extension (cm) over the course of 6 weeks in response to the different treatments. Pneumatophores covered by plastic (grey boxes) extended significantly more than uncovered pneumatophores (black boxes) (p < 0.01). **b**. Tree stress response based on average (±SE) litter fall in gram dry weight per day for per treatment plotted over the course of the experiment. Weekly averages of maximum daily wind speed displayed on the second y-axis. **c**. Tree stress response based on median leaf area index (±95% obs range) per treatment after the experiment. **d.** Tree survival displayed as the proportion of trees per treatment that survived the suffocation experiment (original number of trees per treatment is indicated with “n=”).*

5.5 Discussion

Plastic waste is a growing problem in coastal regions and seas, as plastics accumulate in local ecosystems. Deterioration of coastal ecosystems and the increase of plastic waste in these systems could especially impact people that are directly dependent on these habitats for ecosystem services, such as food and coastal defence. In this study, we attempted to understand the extent of the plastic waste problem in mangrove forests in a rural, though relatively densely populated, area on the coast of Java, where people largely depend on aquaculture and fisheries. These rural communities benefit from healthy mangrove ecosystems for both coastal defence and for shrimp, crabs and commercial fish species. We therefore assessed the extent and the hypothesized negative effect of the omnipresent plastic waste on mangrove growth and survival. Overall, plastic is extremely abundant in mangrove forests and can be found in every plot at multiple depths. In addition, our results indicate that large quantities of plastic negatively affect ecosystem health and tree survival.

In more detail, our results confirm that plastic waste is trapped by mangrove forests. We found multiple sites where a large part of the forest floor (>50%) was covered with plastic waste (Figure 5-4a), and plastic counts were up to 236 items per m^2. These amounts of plastic are not uncommon in mangrove areas near cities. For instance, Rahim et al., (2020) found plastic waste numbers of 378 items per m^2 in Kendari Bay, Indonesia. Although these quantities of plastic are worrisome in and of itself, the plastic that only covers the forest floor and not the exposed pneumatophores probably poses little threat to the mangroves. Aerial roots in the form of pneumatophores are interconnected by below-ground cable roots. Blowing through such a cable root will push bubbles out of lenticels in multiple pneumatophores (Scholander, Van Dam, & Scholander, 1955). This interconnectivity between pneumatophores originating from the same cable root possibly minimizes the effect of scattered plastic on the sediment surface, which might only cover a few pneumatophores per cable root. The remaining pneumatophores can keep the sediment oxygenated at all times. In addition, the plastic can still be resuspended by the tide, thereby only causing temporary suffocation. In contrast, plastic that is deposited on top of pneumatophores during sedimentation events can remain in place for weeks or months, depending on when the next storm hits. Our findings from the sediment cores suggest that a part of the plastic is present in the forest sediment for prolonged periods of time, as more than half of the plastics found in the cores were buried below 2 cm in the sediment (Figure 5-4b). The finding that plastic is not only trapped but also buried in mangrove sediment is in accordance with Costa et al. (2011), who found plastics in various degrees of degradation buried up to at least 20 cm depth in a mudflat of a mangrove fringe in Brazil. If the plastic that was buried in the sediment at our field sites was present in the same abundance as the visible plastic layer on the sediment surface, this could cause a similar suffocating effect as the 100% and 50% treatments in the experiment.

In the field, plastic bags are of course rarely tied to pneumatophores as they were applied in the experiment, and an experimental set-up where the plastic was applied to aerial roots in a more "natural" way could have made the results more easily interpretable. However, despite an apparent loss in LAI in the cable tie control treatment, the tree mortality was not affected by cable tie application method itself, nor did the trees show a pneumatophore growth response when only the cable ties were applied. Furthermore, if the plastic would have been applied in a more realistic manner, covering both the sediment and the pneumatophores, this would only have increased the suffocating effect of the treatments, as oxygen penetration into the sediment through both the sediment surface and the pneumatophores would have been impaired. This would have complicated root respiration by trees even more than the current treatments, making the effects of plastic suffocation with the current set-up conservative estimates. Despite these considerations regarding the methodology, our results show that trees that were treated with 100% plastic cover of their root zone responded with extreme root elongation in an attempt to outgrow the suffocating substance, displayed increased litter fall after treatment application and failed to re-establish a canopy during the experiment. Unsurprisingly, most trees that were completely covered by plastic died in the course of the experiment. In contrast, a more realistic plastic cover of 50% showed that mangrove trees can be resilient to partial plastic suffocation, as the trees of the 50% treatment showed substantial investment in root growth similar to that observed in the 100% trees, but had an intermediate survival compared to the 100% plastic cover, and untreated control treatments.

Resilience to recurring anoxic conditions is a trait of all mangrove species, and in addition to evolutionary root adaptations, most mangrove species can adjust to prolonged inundation events as well (Youssef and Saenger, 1996). Acclimation strategies for coping with inundation events can function as mechanisms to overcome plastic suffocation. However, only few studies report field observations related to mature trees in anoxic conditions. For example, Snedaker, Jimenez and Brown (1981) reported that *Avicennia germinans* trees survived an oil spill, although pneumatophore growth was atypical, with pneumatophores showing a crooked appearance. These root anomalies were similar to observations in our experiment and other heavily polluted field sites (*Figure A 5-4*). Bendy roots may suggest that trees tried to outgrow the suffocating substance. In a study by Ellison (1999), massive sedimentation caused by a hurricane suffocated multiple mangrove species. Mortality was not an immediate result of defoliation, as Ellison noted, because multiple individuals exhibited renewed leaf growth after the hurricane. This side note suggests that the trees did not die immediately from suffocation, but instead reacted similarly to what we appear to see in our 50% treatment; trees that were stressed by suffocation were initially able to endure the anoxic conditions due to heavy investment in root growth and potential foliage renewal. Nevertheless, in Ellison's study, mass mortality occurred sometime between the hurricane and their field observations.

The fact that mangroves can die despite foliage renewal, raises some interesting topics for further research: our experiment was conducted over a relatively short period of six weeks, and although this appeared to be sufficient to observe direct effects of plastic on mangrove trees, the ultimate effect of the 50% plastic waste cover remains unclear. Therefore, it would be interesting to know what the effect is of the current plastic waste levels on the overall life expectancy of mangroves trees. Longer term studies are needed to investigate changes in life span under partial plastic burial and other long term effects, such as the potential leakage of chemicals from decomposing plastic waste (Gao and Wen, 2016), or the effect of plastic during different seasons.

Figure 5-6: *Conceptual figure with a summary of the effects of plastic waste on mangrove growth and survival.*

Notwithstanding the uncertainties regarding long-term effects of plastic on mangroves, our study demonstrates for the first time that mangroves are affected by large amounts of plastic on pneumatophores, and that these quantities of plastic are not uncommon in mangrove fringes (Figure 5-6). In addition, the plastic waste disposal into the environment is only expected to increase in the future (Jambeck *et al.*, 2015). Managers should therefore confront this problem alongside traditional mangrove conservation and restoration. Mangroves are relatively cheap ecosystems to restore compared to other marine ecosystems (Narayan *et al.*, 2016), but median restoration costs for mangroves still range from 1,191 USD/ha in developing countries, and up to 38,982 USD/ha in developed countries (Bayraktarov *et al.*, 2016). Meanwhile, overall restoration success currently remains one of the lowest compared to restoration success in other marine ecosystems such as coral reefs, saltmarshes and oyster reefs (Bayraktarov *et al.*, 2016). The success rate of mangrove restoration could potentially be increased if, in addition to budgets for planting mangroves, budgets for mitigation of potential stressors such as plastic waste reduction are also made available.

5.6 Conclusions

Plastic waste is trapped by mangrove forests, and can be present in high quantities both on the forest floor and in the sediment. Layers of plastic that are deposited on top of mangroves' aerial roots can cause an immediate local response, as trees invest in root growth to outgrow the anoxic conditions. Mangrove trees that are partly covered by plastic show a root-growth response and are seemingly stressed, but appear to be able to endure partial suffocation. Mangrove trees in which the root zones are entirely covered by plastic will ultimately die. Our findings suggest that mangrove trees are stressed by the current plastic pollution levels, especially near sources of mismanaged plastic. Mangrove restoration projects could therefore benefit from plastic management alongside conventional restoration efforts such as planting or habitat rehabilitation.

5.7 Data availability

Data in support of this manuscript are available at: https://doi.org/10.4121/13109561

5.8 Acknowledgements

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5.9 Appendices

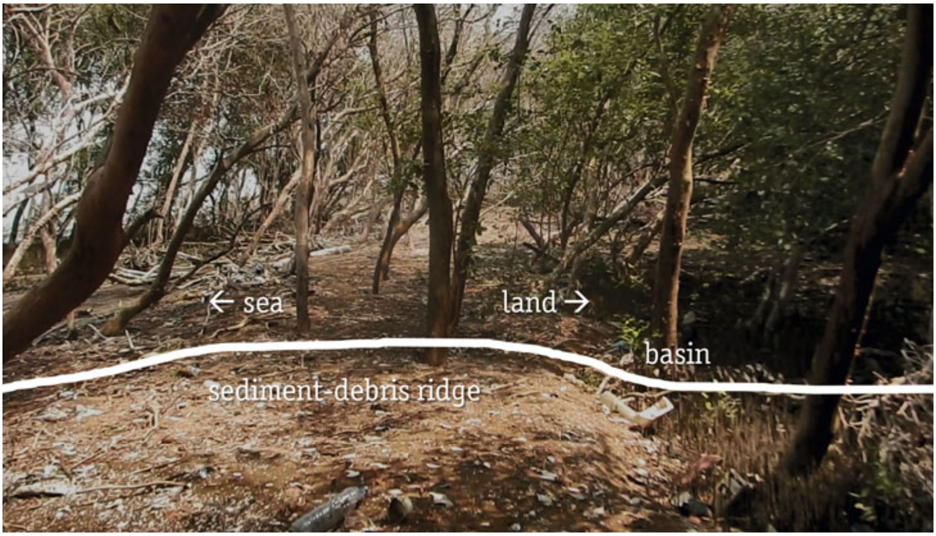

Figure A5-1: *Typical sediment ridge with accumulated plastic waste in a mangrove fringe of Demak regency, Java, Indonesia. Note the reduced canopy cover of trees located inside the sediment ridge.*

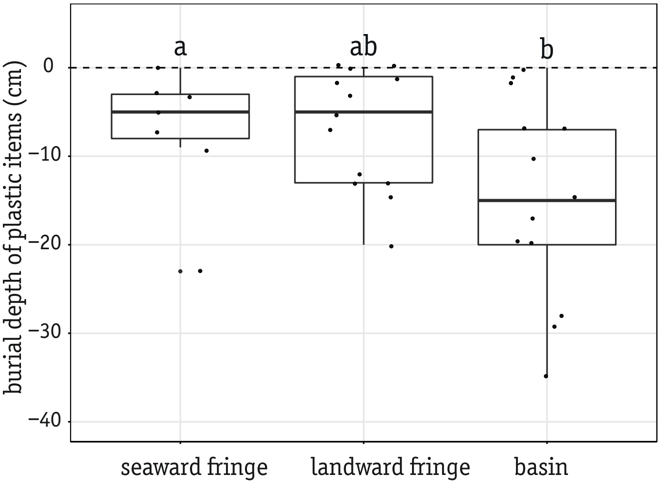

Figure A5-2: *The burial depth of plastic layers across mangrove zones (seaward fringe, landward fringe and basin).*

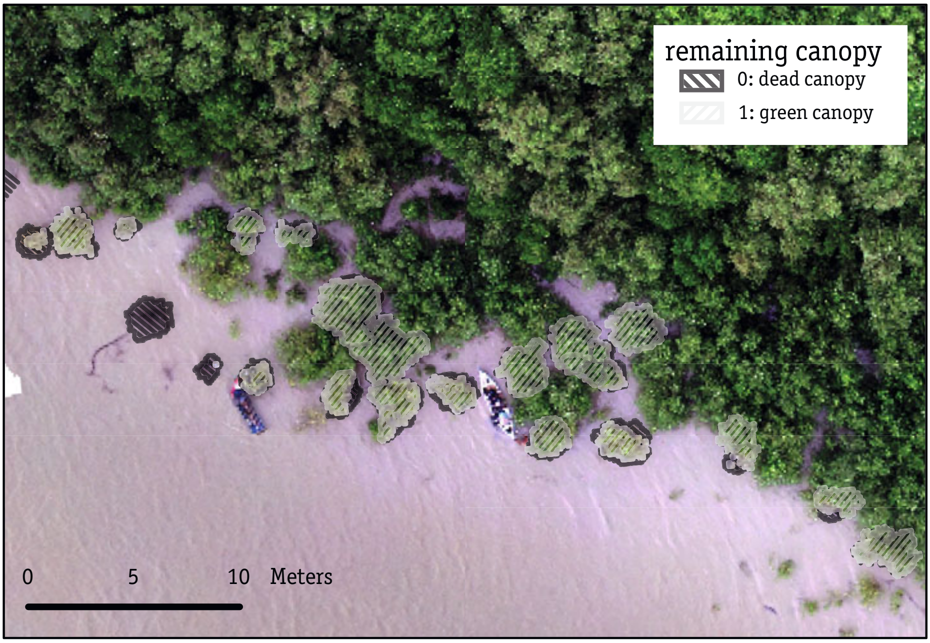

Figure A5-3: *Drone image of part of the study area, white hatched filled polygons (1) indicate vegetation based on Normalized Red Green Difference Index above zero, and black hatched filled polygons (0) display lost canopy cover compared to baseline.*

Figure A5-4: *One of the many mangrove trees with crooked pneumatophores fringing a river mouth. The pneumatophores are entangled in plastic waste, cloths and diapers discharged by the river.*

The effect of rapid sea level rise on mangrove survival

CHAPTER 6

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Submitted as:
A preview of the impact of accelerated relative sea level rise on densely populated mangrove coasts.

6.1 Abstract

Human-induced land subsidence causes many coastal areas to sink with multiple centimetres per year, exacerbating relative sea level rise. While cities combat this problem through investment in coastal infrastructure, rural areas are highly dependent on nature-based flood defences. We studied a 20 km long rural coastline neighbouring a sinking city in Indonesia (8 – 20 cm yr^{-1}) to shed light on the future of low-lying rural areas in the face of sea level rise. By measuring water levels in mangroves and quantifying floor raisings of village houses, we show that while villages drowned rapidly, their protective mangroves sunk much slower and were able to cope with subsidence rates of 4.3 cm yr^{-1} through various root adaptations. However, with subsidence rates four times higher than foreshore accretion rates, lateral mangrove retreat proved inevitable, forcing coastal communities to migrate, sketching a gloomy prospect for marginalized rural communities facing sea level rise.

6.2 Introduction

Land subsidence has recently been recognized as an important magnifier of relative sea level rise, with land subsidence rates on average 4 times higher than global sea level rise (Nicholls *et al.*, 2021). Locally, the effects can be even stronger (Chaussard *et al.*, 2013), causing whole cities to sink with multiple metres in a matter of decades (e.g. Jakarta: 1 m, Bangkok: 2 m, Tokyo: 5 m)(Nicholls, 2010)). Many of the areas subject to land subsidence are productive low-lying areas near deltas and coastal cities where industrial activities require large amounts of ground water extraction (Syvitski *et al.*, 2009; Nicholls, 2010; Chaussard *et al.*, 2013). The combination of water, oil or gas extraction and the weight of the coastal city, often built on top of soft alluvial deposits, then causes the city to sink (Syvitski *et al.*, 2009; Abidin *et al.*, 2011). The rate of land subsidence can be diminished through strict regulation of sub-surface fluid withdrawal, as has been shown in the case of Tokyo (Esteban *et al.*, 2020). However, in many countries this intervention proves difficult for multiple reasons. Most commonly, because alternative water sources, such as seawater and surface water, are either too polluted (Hutton and Chase, 2016), requiring a drastic improvement of basic sanitation and domestic waste management first, or too expensive to utilize (e.g. desalinization of seawater (Hoslett *et al.*, 2018)). The effect of land subsidence is found strongest in South East Asia, with poor surface water quality (Widmer *et al.*, 2013) and many megacities near the coast (Hanson *et al.*, 2011; Nicholls *et al.*, 2021).

For many countries in this region, mitigation of subsidence effects is currently the only option, but it comes with challenges. The most common mitigation response to rising water levels is heavy investment in conventional flood defence structures such as seawalls and dikes to protect the low-lying hinterland from the sea and rivers (Esteban *et al.*, 2020; Yin *et al.*, 2020). These are often accompanied by the creation of expensive and high maintenance polders and their pumps to get the water out. This requires a healthy economy, since everything that is invested in sea-walls, dikes and polders, typically needs to be earned within the area protected by them. If land subsidence extends towards the intertidal area, coastal flood defence structures will sink with the land (Park, Kwon and Hadi, 2016; Yin *et al.*, 2020), increasing the risk of a dike breach, and thus magnifying their maintenance costs (Yin *et al.*, 2020). To date, subsidence research has largely focussed on urban areas, where the problem often originates, while only few studies have investigated how land subsidence propagates to the wider alluvial plain via connected aquifers that lie underneath them (Figure 6-1) (Syvitski *et al.*, 2009; Minderhoud *et al.*, 2019). The impact of land subsidence on poor rural areas may be even worse than on cities, as those areas often lack the financial means for conventional flood defence structures, making them highly dependent on nature-based solutions such as restoration and conservation of vegetated foreshores (Triyanti *et al.*, 2017; Esteban *et al.*, 2020; Menéndez *et al.*, 2020).

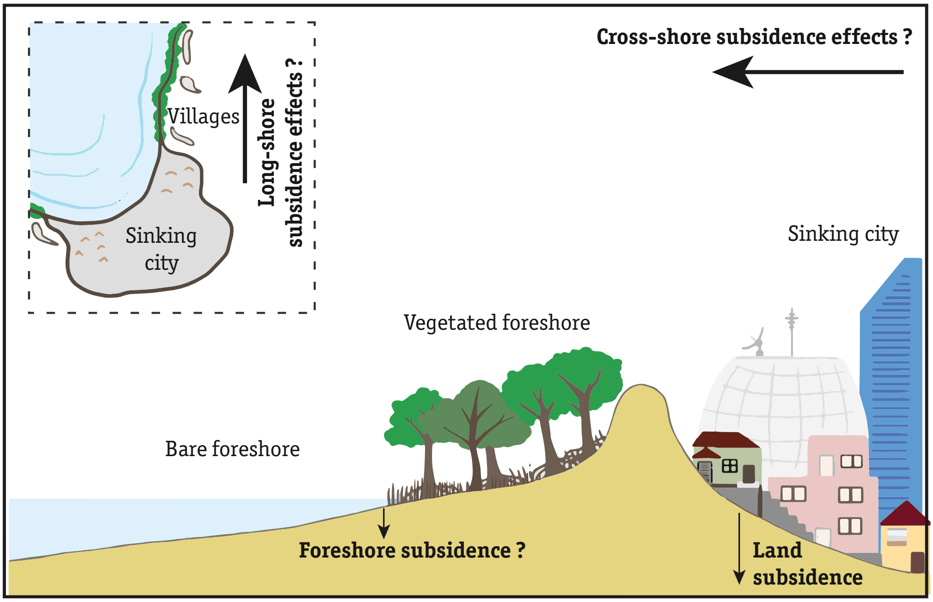

Figure 6-1: *Hypothesized effects of land subsidence in coastal cities (1): propagation of land subsidence in long-shore direction towards adjacent rural areas and villages. (2): cross-shore propagation of land subsidence towards the foreshore potentially threatening the cities' and adjacent villages' flood defence structures.*

Vegetated foreshores like mangroves and salt marshes can attenuate waves (Stark *et al.*, 2015; van Wesenbeeck *et al.*, 2016; Montgomery *et al.*, 2019; Willemsen *et al.*, 2020), and reduce the probability and impact of dike breaches (Zhu *et al.*, 2020). Because these foreshores can trap sediment and/or form peat (Mckee, Cahoon and Feller, 2007) , they can keep up with reasonable rates of sea-level rise provided sufficient sediment is available (Lovelock *et al.*, 2015; Kirwan *et al.*, 2016; Saintilan *et al.*, 2020). However, few studies investigate to what extent mangroves, and intertidal areas in general, can cope with land subsidence, which is often an order of magnitude larger than global sea level rise. If vegetated foreshores like mangroves and salt-marshes subside less rapidly than dikes and land (*Figure 6-1*), or if they are capable of adapting to extreme subsidence rates, they might be a key component in maintaining coastal resilience under sea level rise, especially in rural areas. Using data from a 20 km long coastline neighbouring the extremely fast subsiding city Semarang (8-20 cm per year), a coastal city on the North coast of Java, Indonesia, we demonstrate how the neighbouring rural area and its vegetated-foreshore are affected (*Figure 6-1*), and discuss the implications of mangrove presence for coastal resilience under such conditions.

6.3 Results and Discussion

6.3.1 City subsidence affects adjacent rural areas

In Demak (Java, Indonesia), decades of land subsidence caused by industry in the adjacent city Semarang (Sarah *et al.*, 2020) (Figure 6-2a), has led to extreme land loss (496 ha (Ervita and Marfai, 2017) ,Figure 6-2b). Census data at village level in Demak district (Badan Pusat Statistik Kabupaten Demak, 2021) show that the advancement of the sea and increasing flood frequency forced up to 30% of the people living in coastal villages to move to more crowded inland areas (Figure 6-2 c & d). A questionnaire that we distributed among households of 14 coastal villages along 20 km of coastline reveals that the remaining people are adapting to increased flood frequency through periodic heightening of their houses' ground floor (Figure 6-2f). When the total amount of floor raised over the years is divided by the time since construction of the house, we obtain an approximation of the subsidence rate in these villages, which ranges from approximately 5-20 cm per year near Semarang, to 0-2 cm per year near the Wulan Delta, 20 km further Northeast along the coastline (Figure 6-2e). The village subsidence rates near Semarang that we obtained via this method correspond to the subsidence rates reported in literature for the area, which range from around 8 cm year^{-1} (Kuehn *et al.*, 2009; Chaussard *et al.*, 2013; EO4SD, 2017) up to 20 cm year^{-1} (Marfai and King, 2007; Abidin *et al.*, 2013).

6.3.2 Land subsidence propagates only mildly to intertidal areas with mangroves

The propagation of land subsidence from the city towards the rural coastal area, not only affected the land, but also the intertidal area, though to a lesser degree. A monitoring campaign in which we attached small water level loggers to mangrove tree trunks for a period of 2 years revealed that subsidence of mangrove forests occurred, and that it decreased along the coast with larger distances away from the subsidence epicentre (i.e. Semarang) (Figure 6-3a). This longshore trend, with an exponential decrease in subsidence, was similar to the subsidence gradient observed in the villages (*Figure 6-2*e & *Figure 6-3*a). However, mangroves subsided significantly less rapidly than villages (*Figure 6-3*a, R^2: 0.9, $F = 62.7$, df = 2 & 12, $p < 0.0001$). For instance, where subsidence in the village closest to Semarang was on average 8.2 (95%CI: 6.0-10.4) cm per year, the mangrove subsidence rate in this area was 'merely' 4.3 (95%CI: 2.3-6.3) cm per year on average (Figure 6-3a). The maximal observed intertidal subsidence rate was however still an order of magnitude higher than observed global sea level rise (3.8 mm / year (Nicholls *et al.*, 2021)), but almost twice as low as land subsidence rates.

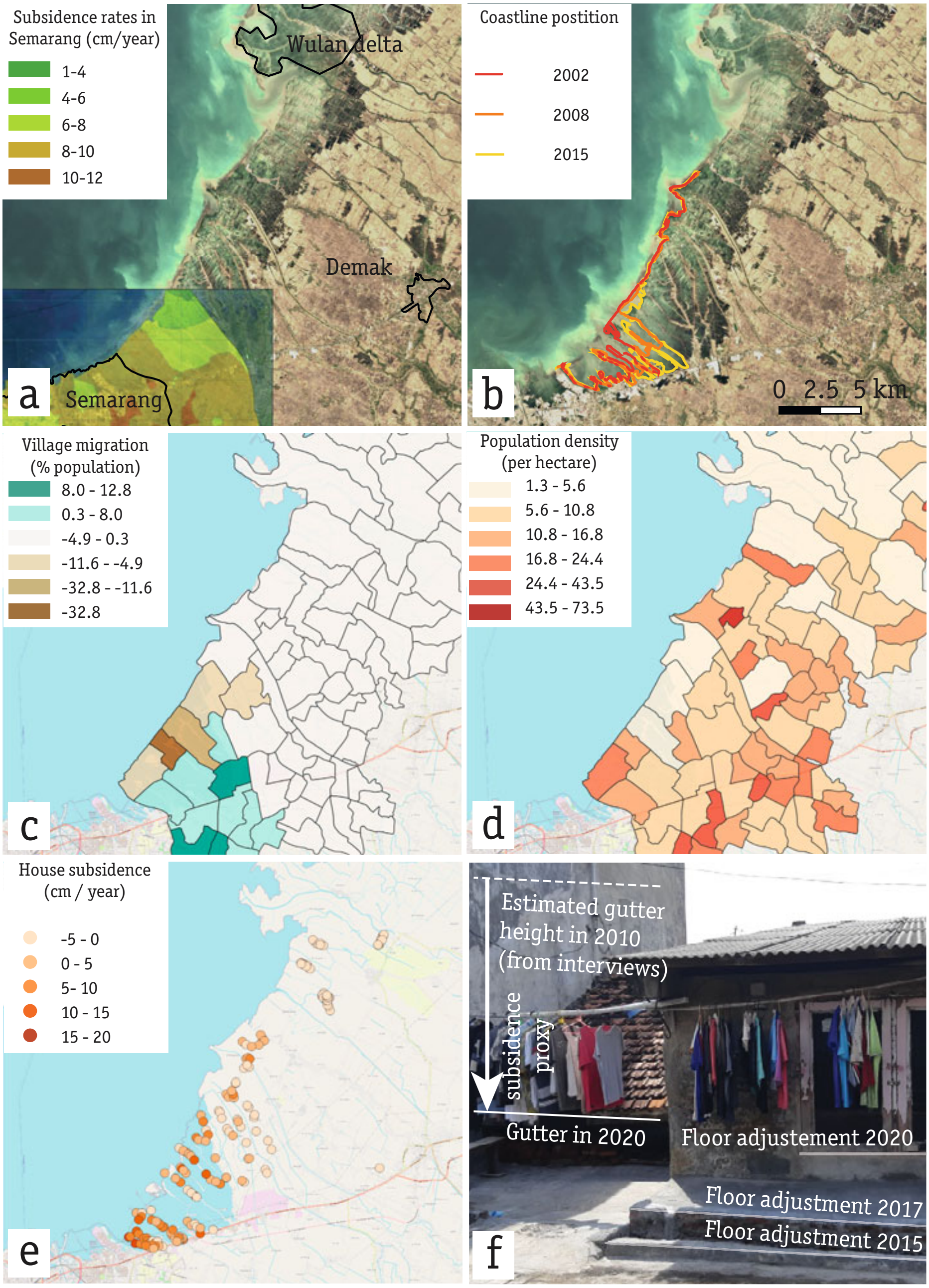

Figure 6-2 a: *Satellite imagery (Sentinel-2) of the study area: the coastline of Demak district, Demak city is outlined in black. The study area comprises the rural coastal area that stretches from the coastal plain East of Semarang harbour, to the Wulan delta, (both outlined in black) 20 km to the Northeast. The overlay map in the lower-left corner is a map made by E04SD (2017) and depicts subsidence rates of the area which are among the more conservative in literature.* ***b*** *Coastline retreat between 2002 and 2015, following the subsidence in Semarang (adapted from (Ervita and Marfai,* *2017)).* ***c*** *Migration fluxes in percentages of the village population in the year 2010 following coastline retreat depicted in (b). People mainly migrated from the coastal villages towards villages further inland (c), where population densities (**d**) are higher.* ***e.*** *Average ground floor elevation increase / gutter height decrease (cm / year), an adaptation to increased flooding frequency by people living in the coastal area* ***f.*** *The process of raising the ground floor every so many years by a certain amount, without roof adjustment, leads to houses with gutter heights close to street level after one decade.*

6.3.3 Intertidal subsidence drives both vertical foreshore erosion and lateral shoreline erosion

When there is not enough sediment to compensate for subsidence, the subsidence of the intertidal area may facilitate accelerated vertical foreshore erosion, as with deeper foreshores, higher waves can reach the shoreline. This indeed seems to be the case in Demak, where the foreshore 50 m in front of the mangrove fringe was consistently deeper at sites close to the subsidence epicentre, than at sites further along the coast (Figure 6-3b, $F = 5$, $R^2 = 0.25$, $p < 0.05$). Such deep foreshores allow high waves to reach the shoreline (Battjes and Janssen, 1978; Padilla-Hernández and Monbaliu, 2001), causing waves to break closer to the shoreline, which can further excavate the foreshore bed (Winterwerp *et al.*, 2013). Erosion monitoring of the bare foreshore, 50 m seaward of the mangrove fringe, indeed revealed that deeper foreshores showed much faster vertical erosion rates (*Figure 6-3*b, $F = 4.8$, R^2 = 0.2, $p < 0.05$) than sites with shallower foreshores further away from the city (*Figure 6-3*b).

Besides vertical erosion of the foreshore, increased wave energy at the shore can also cause mangrove retreat when sediment is rinsed out from between roots. The result of this lateral erosion process is reflected in the mangrove mortality data, where both the mortality of monitored aerial roots (coeff = 0.08, SE = 0.01, $p < 0.001$, *Figure 6-3*c, grey), and the number of dead trees per 50 meter coastline stretch (coeff = 0.02, SE=0.001, $p<0.001$, *Figure 6-3*c, black) increased significantly with foreshore depth along the subsidence gradient. Deep and concave foreshores have indeed been associated with lateral coastal erosion(Mariotti and Fagherazzi, 2013) and mangrove retreat (Anthony *et al.*, 2010; Winterwerp *et al.*, 2013; van Bijsterveldt *et al.*, 2020). This suggests that subsidence in combination with a sediment deficit instigates a self-reinforcing process of lateral coastal erosion (Figure 6-4c).

6.3.4 Mangrove trees can cope with extreme subsidence rates

While at rapidly subsiding sites major vertical erosion occurred at the foreshore in front of the mangrove fringe, high deposition rates were observed inside those mangrove fringes where subsidence was high (*Figure 6-3*d). A similar process is frequently observed along vegetated foreshores where lateral cliff erosion occurs, creating levees along the remaining marsh edge (Zhao *et al.*, 2017; Yang *et al.*, 2020).

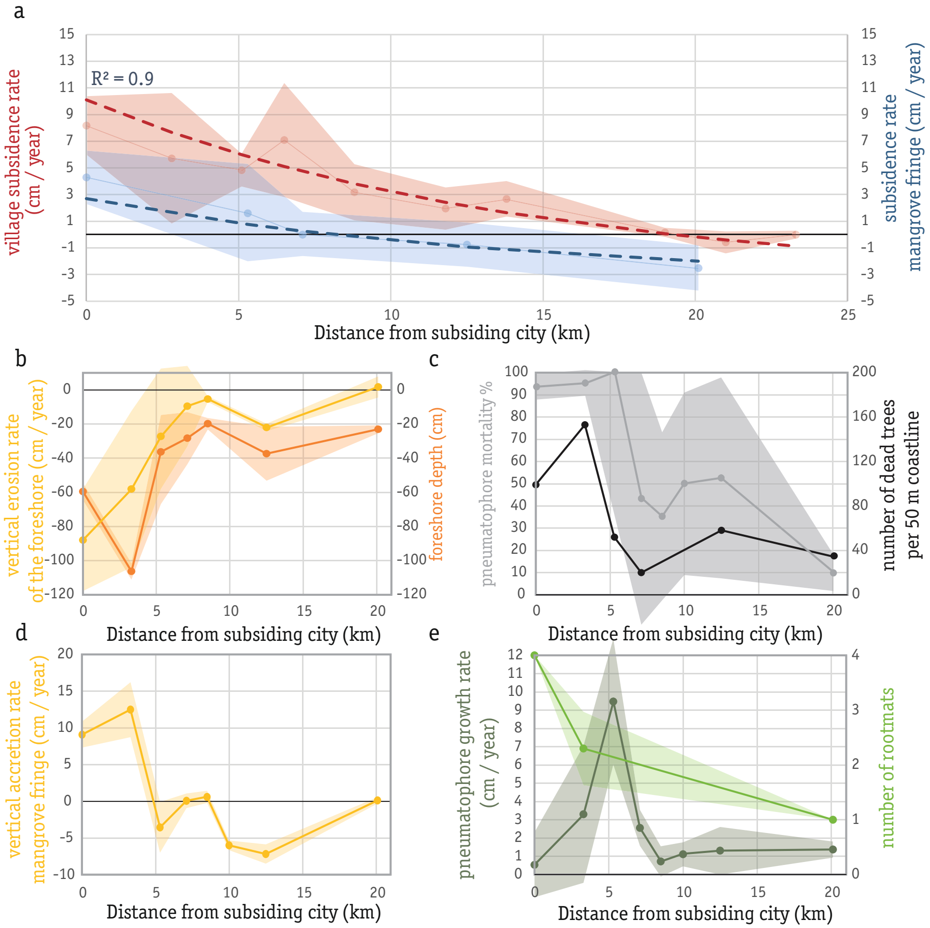

Figure 6-3: *Parameters measured in the mangrove fringe and foreshore of Demak regency, the rural area neighbouring Semarang city. The rapidly subsiding harbour area of Semarang is distance 0 on the x-axis.* ***a****. Average village subsidence rate (cm / year) (+/- 95% CI) and mangrove subsidence rate (cm / year) (+/- 95% CI) of mangrove stands measured over 1.5 years with pressure loggers attached to mangrove trees fringing the shore.* ***b.*** *Average (+/- 95% CI) vertical foreshore erosion rate and average (+/- 95% CI) foreshore depth relative to mean sea level. Both parameters were measured periodically over a period of 1.5 years at sedimentation poles located 50 m offshore from the mangrove fringe along the subsidence gradient.* ***c.*** *Primary y-axis: Average (+/- 95% CI) pneumatophore mortality (%) in the mangrove fringe at the end of the 1.5-year monitoring campaign. Secondary y-axis: number of dead trees observed per 50 m mangrove fringe in longshore direction at the start of the campaign.* ***d*** *Average (+/- 95% CI) sediment accretion/erosion (cm / year) relative to marked pneumatophores of mangrove trees in the fringe along the subsidence gradient monitored for a period 1.5 years.* ***e.*** *Primary y-axis: Average (+/- 95% CI) pneumatophore extension rate (cm / year) monitored in the mangrove fringe over a period of 1.5 years. Secondary y-axis: Number of rootmats found on living trees in the mangrove fringe that were rinsed-out at the start of the campaign.*

These levees are created when water decelerates upon entering the vegetation, causing the suspended sediment to sink to the bottom in the first few meter of the marsh or mangroves (Yoshihiro Mazda *et al.*, 1997; Horstman *et al.*, 2017). Our data suggest that in areas characterized by shoreline retreat, sediment supply to the mangrove fringe is high, and most likely originates from local foreshore erosion (Figure 6-3 b & d, Figure 6-4c).

At the most rapidly subsiding sites, sediment deposition in the fringe was sufficient to keep the forest floor at a constant elevation relative to the water level (*Figure 6-3*d). In principle, under these conditions, mangroves should be able to outgrow extreme subsidence rates, provided that they are able to withstand equal sedimentation rates. A manipulative field experiment in which we applied treatments of 0 cm, 20 cm and 40 cm of sediment to the root zones of young mangrove trees (Figure A 6-1) confirmed that mangroves can survive sudden sedimentation events of 20 centimetres. Trees should therefore be able to keep up with the most extreme subsidence rates reported in literature (Marfai and King, 2007; Abidin *et al.*, 2011), when the space of the subsiding forest floor is filled-up with sediment. A study by Sidik, Neil and Lovelock (2016) revealed that the aforementioned depositions rates (>30 cm per year) can be found in nature. However they also found that high sediment accretion rates (above 10 cm per year) resulted in lower tree growth rates (measured as basal area increment), presumably due to anoxic stress due to covered pneumatophores. Our study shows that one of the mechanisms with which trees may cope with this anoxia brought on by high subsidence and/or sedimentation rates is through new root-mat formation: i.e., the growth of new lateral cable-roots in a fresh sediment layer, from which new pneumatophore can grow upward (Figure A 6-2a). Trees at rapidly subsiding sites with high accretion rates in the fringe (*Figure 6-3*e, distance < 5 km) indeed revealed significantly more root-mats in subsurface layers than trees at the least subsiding site (X^2 = 7.8, df =2, $p < 0.05$). The coping mechanism to keep up with slowly rising water levels without sediment accretion seemed to be through pneumatophore extension (Figure A 6-2b & Figure 6-4a), as pneumatophore growth increased significantly with higher subsidence rates ($F = 7.1$, $p < 0.01$) at the sites with moderate to low subsidence rates and stable or slightly eroding forest-floors (Figure 6-3e, distance > 5 km). These results suggest that in a vertical direction, mangrove trees can cope with the observed subsidence rates of 4.3 (95%CI: 2.3-6.3) cm / year, especially if enough sediment is available (Figure 6-4d). Individual mangrove trees can therefore cope with relative sea level rise that is almost an order of magnitude higher than previously thought based on paleo data (6 mm / year (Saintilan *et al.*, 2020)).

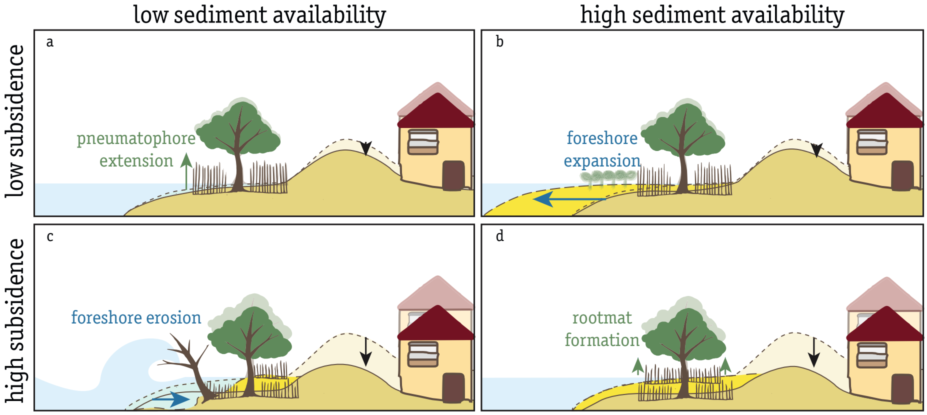

Figure 6-4: *Summarizing schematic representation of the key processes affecting the development of cross-shore profiles under low and high subsidence rates (rows), in the presence of low and high sediment availability (columns). The dotted lines and transparent houses and trees represent pre-subsidence profiles. The solid lines and figures represent sunken profiles. The dashed lines represent sedimentation (yellow) or erosion (transparent (only in c)). Note that foreshore subsidence is lower than land subsidence under all conditions, but the response of the vegetated foreshore to subsidence largely depends on the sediment availability.* ***a.*** *Pneumatophores extend with rising water levels, under low to moderate subsidence rates and limited sediment supply (5-7 km from Semarang).* ***b.*** *When subsidence is low and sediment supply is high, foreshores tend to expand (as reported in literature (Syvitski et al., 2009)).* ***c.*** *Under high subsidence rates (0-5 km from Semarang), sinking foreshores accommodate higher waves, which erode the local bare foreshore and deposit sediment inside the mangrove fringe. Trees in the fringe initially survive due to the local sediment input in which they can create new rootmats but will inevitably die when the coastline retreats due to lateral erosion.* ***d.*** *Vegetated foreshores can remain stable if the sediment availability matches foreshore subsidence, leaving the foreshore profile intact. This foreshore situation was not observed in our study, and thus derived as a hypothetical outcome, partly supported by the literature (Syvitski et al., 2009) and partly by our experimental findings, as in* ***c,*** *our results have shown that trees can survive high subsidence and sedimentation rates by forming new rootmats in freshly deposited sediment, and can survive sedimentation events of up to 20 cm.*

6.3.5 Implications for sustainable flood defence under extreme land subsidence

Even though vegetated foreshores can be resilient to extreme forms of relative sea level rise in a vertical direction, bare foreshores are not resilient, and will eventually cause vegetated foreshores to retreat due to lateral erosion. This highlights the importance of integrating complete foreshore design, both the bare and vegetated ones, in coastal management plans in subsiding areas. The bare foreshores of subsiding coastal areas need to be convex and shallow for the vegetated foreshores to maintain their position (van Bijsterveldt *et al.*, 2020). In non-subsiding but eroding regions, a convex and shallow profile can be restored through the placement of permeable structures parallel to the coastline, to trap the available sediment and thereby elevate the foreshore bed

(Winterwerp *et al.*, 2020). In Demak, this method has indeed led to sediment accumulation directly seaward of the mangrove fringe, where it was most needed, resulting in local coastline stabilization, although the bare foreshore bed could not be raised enough for natural mangrove regrowth with the available sediment (van Bijsterveldt *et al.*, 2022). To truly restore a convex foreshore profile, an enormous amount of sediment is required (Figure 6-4d), and even more to elevate the bed enough to cause coastline propagation similar to non-subsiding scenarios (Figure 6-4b). In the case of Demak, a simple calculation based on historical coastline propagation data (Methods § Sediment deficit calculation for Semarang) reveals that the sediment accretion in the early 1900s, before the onset of subsidence, used to be around 9 mm / year. Assuming that this would be the maximal restorable sediment input into the system, the coastline with that sediment input would still have a sediment deficit that is over 4 times larger than the foreshore subsidence rates that we find today. Similar trends have been observed in deltas worldwide, with the most rapidly subsiding deltas being subject to compromised sediment discharge from rivers (e.g. due to upstream dams and canalization) and subsequent coastal retreat (Syvitski *et al.*, 2009). When a positive sediment balance cannot be restored, mangroves need space to move landward to survive rapid subsidence. A similar conclusion was drawn for salt marshes, which proved resilient to SLR when suspended sediment concentrations were high, but relied on landward transgression to survive when sediment inputs were low (Kirwan *et al.*, 2016). This presents a major societal problem: most low-lying rural coastal areas are highly productive and densely inhabited, so that there is no ‘unoccupied’ space for a landward escape.

6.3.6 The future of coastal communities in subsiding areas

Subsiding cities, culprit of their own demise, continue sinking together with their flood defences. River and tidal floods become more and more frequent and severe inside the city, as runoff can no longer be drained to the sea, while increasingly forceful waves attack sinking dikes and sea walls from the outside. Subsiding cities, however, have the economic power to compensate for these problems (e.g. Jakarta giant sea-wall (Garschagen, Surtiari and Harb, 2018)), at least partly supported by the extracting industries that are at the base of the subsidence problem. Unlike the people within the city’s boundaries, rural communities are often not protected by expensive advanced flood defence structures and associated polders. The migration and floor raising data show that people respond to the advancing sea and frequent flooding in one of two ways: fight (i.e., keep the water out of the house) or flight (move further land inward or elsewhere). These adaptation strategies have been reported in previous studies, ironically, focussing on how flood-prone communities can improve their adaptation strategies (Marfai and Hizbaron, 2011; Marfai, Sekaranom and Ward, 2015). Sadly, 55%

of those respondents indicated not to move to higher grounds due to financial limitations (Marfai and Hizbaron, 2011). This illustrates how marginalized rural communities near populous cities will receive the negative impacts from relative sea level rise, unless their more prosperous neighboring regions start installing measures to address the root cause of the problem and take an integrated landscape approach to coastal protection that connects urban to rural areas, combining for instance hard infrastructure and vegetated foreshores along a gradient from rural to urban areas (Willemsen, Cado van der Leij and Van Wesenbeeck, 2019). Mitigation measures that address the root causes might include smart sediment management to optimize sediment input into the system (Hu *et al.*, 2021), but should always also involve a strict regulation of subsurface fluid extraction to halt subsidence along with providing alternative sources of water use. If relative sea level rise is not addressed in this integrated way, the current small scale refugee crisis will continue to unfold, as communities are left with little choice but to retreat landward with the mangroves that protect them from waves. It is an outcome that may one day occur on a global scale, where developing countries might have to cope with massive evacuation caused by sea level rise, while developed nations can economically afford responsive measures, due to past economic development that exacerbated CO_2 emissions and sea level rise in the first place.

6.4 Acknowledgements

The BioManCO project is a collaboration between TU Delft, NIOZ and UNDIP and makes use of the framework set up by Building with Nature Indonesia, a program by Ecoshape, Wetlands International, the Indonesian Ministry of Marine Affairs and Fisheries (MMAF), the Indonesian Ministry of Public Works and Housing (PU) and other partners. We would like to thank the Alejandra Gijón Mancheño, Bas van Maren, Ad Reniers, Wim Uijttewaal, and Han Winterwerp of the BioManCo team for their fruitful input in discussions about this topic during project meetings.

Furthermore, we are grateful to the group of 32 volunteers, including students from Diponegoro University, State University of Semarang and local volunteers who, coordinated by Sri Ramadhani, conducted the interviews and distributed the questionnaire to the local community in Semarang-Demak area. In addition, we would like to thank Lubabun ni'am and Fajar Rahmawan who provided additional insight into the socio-economy issues in the local communities of the Semarang-Demak area. We would like to thank Aris Ismanto and Rudhi Pribadi for facilitating student participation of Diponegoro University in foreshore field experiments. Additionally, we thank the Wetlands International Indonesia team for connecting us to the community in Wedung so that we could extend our study area towards the border of the subsidence area. In particular, we would like to thank Kuswantoro and his family for hosting us and facilitating fieldwork in a reference area outside Demak regency. Unfortunately, we had to remove this reference site from the analysis because the site had been destroyed for construction of an industrial aquaculture pond in which we lost all valuable water level and tree monitoring data. The Wetlands International Indonesia team has also greatly supported us by identifying and providing us with relevant census data of the Demak regency (thank you Apri Susanto Astra) and by connecting us to local village chiefs for background interviews on the subsidence in the area (thank you Eko Budi Priyanto). The fieldwork in Demak Regency would not have been possible without the hospitality of bapak Sairi and ibu Musaini, and their children, as well as, bapak Slamet and ibu Paini and their family. Both families cordially hosted the researchers and students in their own homes and pak Slamet and his family actively facilitated and participated in field expeditions. We would also like to acknowledge Pak Muis' role as a local translator, which was greatly appreciated. We would like to thank pak Umar for his translation skills, in combination with his transportation services, and enthusiastic participation in field expeditions. We thank bapak Yogie, manager of Combo Putra hardware store in Banyumanik, for his technical advice and enthusiastic participation in experiment and monitoring design with the materials available. Finally, we would like to thank Tjisse van der Heide for helping to bring out the main message of the paper in the cover letter.

6.5 Methods

6.5.1 Village migration data

Village migration and population density data were obtained from the website of the central bureau of statistics of Demak Regency (Badan Pusat Statistik Kabupaten Demak, 2021). The oldest census data available at village level were from 2009, with arrival and departure data per village available for every 5 years since then. Published shoreline change data (Ervita and Marfai, 2017), and a timeline search in Google Earth Pro (version 7.3.3.7786) for the area near Semarang showed that in 2009 a major shoreline change occurred, which would likely affect the livelihood of the local communities. We therefore decided to use the migration data at village level from 2009-2010 to investigate how many people moved away from the area after the event. The migration flux was calculated by the following equation:

$$Migration\ flux\ \% = \frac{arrivals - departures}{population} * 100$$

This equation gives negative percentages when a proportion of the village population left, and positive values when the village population increased through migration.

6.5.2 Village subsidence and community adaptation (questionnaire)

We used a questionnaire to investigate community adaptation to increased flood frequency and inundation along the 20 km coastline gradient between Semarang and the Wulan Delta. In total, 194 households, distributed over 14 villages along the coast responded to the questionnaire, which was distributed by a group of students and local volunteers who spoke the local language. Respondents were asked in what year their house was constructed, and how much they had raised the ground floor of their house over the years. In addition, they were asked to indicate what the original height of the gutter had been when their house was constructed. The current gutter height was then measured by the surveyor in situ. The subsidence rate of the house was then calculated using the floor raising data, and the gutter data with the following formulas:

$$\textit{house(floor) subsidence} = \frac{\textit{total amount the floor was raised since construction (cm)}}{\textit{2020 – construction year}}$$

$$\textit{house(gutter) subsidence} = \frac{\textit{gutter height at construction (cm) – current gutter height (cm)}}{\textit{2020 – construction year}}$$

The outcome of the two subsidence calculations were validated against each other, rendering a clear linear correlation with an R^2 of 0.5 (Figure A 6-3). Five houses were omitted from the subsidence calculation, because they had raised their walls after raising the floor, or because they constructed an entirely new house where they already anticipated the flooding frequency and subsidence to come.

The gutter height subsidence data were then used to compare village subsidence to foreshore subsidence measured in the mangroves. To this end, an average subsidence rate (+/- 95% CI) was calculated from the gutter dataset for each of the villages located directly along the shore (notably: Sriwulan at 0 km, Purwosari at 2.8 km, Bedono at 5.1 km, Timbulsloko at 6.5 km, Surodadi at 8.8 km, Tambakbulusan at 11.8 km, Morodemak at 13.8 km, Wedung at 19.3 km, Berahan Kulan at 21 km, and Berahan Wetan at 23.3 km from Semarang). Distances were obtained using the line measurement tool in QGIS (version 2.18.12) between the centroids of each of the village's administrative boundaries and the start of the mangrove subsidence gradient in North-East Semarang. The relation between village subsidence, mangrove subsidence (see next paragraph for methods) and distance from the city was investigated with the following log-linear model:

$$\log\textit{(subsidence)} \sim \textit{longshore distance} + \textit{cross shore zone}$$

Where longshore distance was the distance from site to start of the subsidence gradient, and the cross-shore zone was a factor variable that contained the site type information: “village” or “mangrove”. Fitted curves from this log-linear model were plotted in *Figure 6-3*a, on top of the observed average (+/- 95% CI) village and mangrove subsidence rates.

6.5.3 Vegetated foreshore / mangrove subsidence

Sinking rates of mangrove forests along the expected rural subsidence gradient were quantified with the use of small pressure sensors (Onset HOBO Water level logger U20L-04), which were covered with a sock (to prevent theft) and tied to the trunks of eight mangrove trees in mangrove stands located along 20 km of coastline. The loggers were deployed for a period of 2.25 years. Due to the way that the sensors were deployed (*Figure A 6-4*a), all sensors were located well above MSL, ranging from 22 until 27 cm above MSL. The sensors were cleaned and redeployed repeatedly. Whenever a sensor-tree was lost by a storm, a new sensor was deployed on a tree further inside the mangrove forest, anticipating future storms.

The regular loss of trees and sensors ultimately led to five continuous pressure datasets that ran for more than 1 year, from which the tree-subsidence rate could be quantified. In order to do so, the raw pressure files were first corrected for air pressure (obtained from the daily emerged window of each sensor), and then converted to water depth, assuming a constant temperature of 30 °C, and a salinity of 30 ppt, which is the average year-round temperature and salinity for this region. Resulting water depth time series were then fitted to the subsidence-corrected tide prediction, which was obtained through a tidal harmonic analysis of freely available data from the tide station in Semarang (see methods § Subsidence of the tide monitoring station in Semarang). The exact height of the logger with respect to MSL was determined by comparing submergence time per day of the loggers, averaged over the first three months, to expected submergence time based on the stations’ tidal signal (Figure A 6-5). After fitting of the logger’s water level series to the tidal curve, inundation time and average water depth recorded by the loggers could then be corrected for expected inundation time and average water depth based on the position of the logger in the intertidal zone at the start of the campaign. A linear regression through all water level logger points per logger over the full deployment time of each logger was unfortunately not possible, because multiple loggers were deployed near a creek or river mouth. We observed a significant increase in water level over the wet season for those loggers, clearly caused by fresh-water run-off during the wet season. We could not correct for this effect because the discharge of each of the individual rivers and creeks was unknown. We therefore excluded the data from the wet-season, and only used tide-corrected water levels during the dry seasons. The significant difference (+/- 95% CI) in average water depth between two consecutive dry seasons (a period of 60 days, exactly one year apart) was then used to calculate subsidence rates (Figure A 5).

$$\textit{mangrove subsidence rate} = \frac{\text{tide corrected waterdepth 2019} - \text{tide corrected waterdepth 2018}}{\text{1 year}}$$

Statistical differences in average tide-corrected water depths between two dry seasons were tested with an independent t-test for each site. One site had an overlapping period of only 15 days, instead of 60 days between the two years, because the tree was lost shortly after re-deployment of the logger. We validated if the subsidence trend along the coastline would be similar if only the 15 day period would be used for all sites, which was indeed the case, although confidence intervals became wider (Figure A 6-6a vs Figure A 6-6b). In addition, we used the four sensors along the gradient that had longer time-series to do two extra validations with 60-days time series later in the dry season (June-August (Figure A 6-6c) and August-October (Figure A 6-6d)). All time series showed similar trends in subsidence and increased daily submergence time along the coastline.

6.5.4 Bare-foreshore dynamics

To monitor foreshore dynamics on the bare foreshore seaward of mangrove stands, PVC sediment monitoring poles were placed at approximately 50 m seaward of each subsidence-monitored mangrove stand (Figure A 6-4 b). Each pipe had a diameter of 10 cm, and was 2 m long. The poles were marked at every 10 cm with tape, and driven 1 m deep into the sediment. We monitored the depth of the foreshore, and the relative bed-level change at each pole three times over a period of 2 years. Scour around the pole was corrected for by using a level to bridge distance from the pole to the undisturbed bed. Water depth measured at the sedimentation pole was corrected to depth compared to MSL, using predicted tide (Methods § Subsidence of the tide monitoring station in Semarang) at the time of water level measurement each campaign.

At the two most eroding sites (0 and 3.3 km from Semarang), the erosion rates were so high that the PVC poles were lost within the first 8 months of deployment. The depth of the original pole location had become too deep (subtidal) for deployment and monitoring of a new PVC pipe. The continued monitoring of a subtidal pole was found to be unfeasible, as locating a submerged PVC pipe with GPS during follow-up visits was impossible due to the turbid water, and marker buoys were stolen or lost. The new location therefore had to be more shallow than the original location, and the most meaningful choice was therefore to place the new pole 50 m out of the retreated mangrove fringe. The placement of the two new poles at a more shallow location, meant that for these two sites there was no continuous bed-level dataset, which would have contained increasingly deeper bed-levels. The foreshore depth data that are displayed in *Figure 6-3*c therefore represent a conservative average foreshore depth at subsiding sites. The loss of the original monitoring poles at these two sites, also meant the absence of bed-level change measurements for those sites. We therefore assumed a conservative 60 cm of vertical erosion compared to baseline between the first two monitoring campaigns. This amount of erosion would have toppled over the pole with its original 1 m anchoring depth. Sixty cm appeared to be a conservative estimate of the amount of erosion that had occurred, based on the observed depth at the original locations of these two poles 8 months after initial deployment.

The bed-level change rate between each monitoring campaign was recalculated to a mean sedimentation/erosion rate per month by dividing bed level change between campaigns by the number of months between campaigns. We then used all available relative bed-level change rates measured per site to obtain a mean and standard deviation of foreshore bed-level change per year. Correlation between foreshore depth and subsidence rate, and the relation between foreshore erosion and foreshore depth were tested with a linear model, using the repeated measures per site as replicates (i.e. each depth measurement at t_n and subsequent bed level change rate between t_n and t_n+1) to include temporal variability. We excluded the site 10 km from Semarang from the analysis, as that sedimentation pole was repeatedly removed by people.

6.5.5 Mangrove response to subsidence

To monitor morphological responses of mangrove trees to subsidence, each tree that received a water level logger, also received markings on ten pneumatophores to quantify responses in root-growth and bed-level dynamics. To this end, a cable-tie was tied to each pneumatophore at 10 cm from the tip, and at 10 cm from the bed (*Figure A 6-4*c). However, due to significant lateral loss of marked trees after the first monitoring campaign, we selected 3 or 4 additional trees per site (also including one tree with a newly deployed water level logger) during the second campaign and monitored those trees two more times for bed-level change and root growth over the period of the following 1.5 years.

Bed-level monitoring

Bed-level change inside the mangrove forest was monitored by measuring the distance from the bed to the cable tie that was fastened 10 cm from the bed at baseline. Net forest floor accretion rates were subsequently calculated by dividing the bed-level change by the time that had elapsed between the monitoring campaign and baseline. Due to lateral loss of monitoring trees in the second year of monitoring, only the mangrove response data of the first 8 months were used to compare the vertical bed-level changes inside the mangrove forest between sites along the coast, using a Kruskal-Wallis test, and pairwise Wilcoxon test.

Aerial root-growth monitoring

Pneumatophore growth was monitored by measuring the length of each marked pneumatophore from the cable-tie (10 cm from the tip) to the tip upon each visit to the site. Pneumatophore growth rate was subsequently calculated from the measured root-tip extension, divided by the time elapsed since baseline. The effect of subsidence on pneumatophore growth was tested at the sites where bed-level change inside the fringe was negative or zero over the period of the study (Figure 6-3d, sites > 5 km from Semarang). Pneumatophore growth per month was then added to a linear mixed effects model, with tree ID and subsidence as explanatory variables (fixed factors), and the monitoring campaign added as a random effect.

Root-mat quantity assessment

To investigate how mangrove trees responded to ongoing subsidence over the past decades, we rinsed out the root-zones of three living mangrove trees at three key sites. (1) High subsidence, high sedimentation rates in the forest, and young trees (0 km from Semarang). (2) High subsidence, high sedimentation rates in the forest, and old trees (3.3 km from Semarang). And (3), low subsidence, low sedimentation rates in the forest, and young trees (20 km from Semarang). At each site, the number of distinctly separate rootmats per tree (*Figure A 6-4*d) was quantified in the top 60 cm of the sediment, deeper excavation and rinsing was not possible. Whether the differences observed in the number of rootmats between the sites was statistically significant, was tested with a Kruskal-Wallis test.

Experiment: mangrove resilience to sedimentation

The effects of extremely high sedimentation pulses were simulated in the field by exposing saplings (height = 60 cm, n=6 per treatment group) and young trees (height = 2 m, n = 4 per treatment group) to a sediment increase of either 20 or 40 cm, with a control group where no sediment was added. A PVC tube (diameter = 30 cm) was put over the saplings and filled with locally available sediment. The young trees were surrounded by fencing, constructed from bamboo and plastic and filled with sediment (*Figure A 6-1*). Survival was assessed after 22 days at the end of that field campaign and after 60 days at the start of the next field visit.

Lateral mangrove erosion

Lateral mangrove erosion was quantified by counting the number of dead trees seaward of the mangrove fringe per 50 m coastline stretch at each of the sites along the coast in the subsidence gradient. In addition, loss of monitored pneumatophores gave a clear indication of lateral erosion as well, as marked pneumatophores were often distributed around the tree, so 50% mortality of marked pneumatophores was often a result of lateral erosion in the field, which left the monitored tree at the seaward edge of the mangrove forest. Pneumatophore survival per site was monitored two times in 1.5 years, average survival per site and 95% CIs were plotted in (*Figure 6-3*c). The effect of foreshore depth on pneumatophore mortality and number of trees per coastline stretch was tested with a generalized linear model, assuming a poisson distribution for the dead tree count data, and with a logistic regression for the pneumatophore mortality data.

6.5.6 Subsidence of the tide monitoring station in Semarang

Tidal harmonic analysis of data obtained from the water level monitoring station in Semarang (Flanders Marine Institute (VLIZ); Intergovernmental Oceanographic Commission (IOC), 2021) was performed to account for the tidal signal in the logger data. Using the UTide Matlab functions (Codiga, 2011) on a water level time series between May 2016 and the end of 2018, amplitude and phase lag of each tidal constituent were determined. Besides expected tidal constituents, the tidal analysis also yielded a linear slope in the water level of 3.9 cm per year. Another sensor showed a linear trend of 7.9 cm per year before 2016, and a slope of 0.8 cm per year after the data gap in 2016 (Figure A 6-7). The lack of data for several months and the lower slope of this linear component after the data gap suggest that the tidal station has been re-anchored at a deeper subsurface layer. Although we have not been able to obtain information about what has happened to this tidal station, the data show that even the deeply anchored structure experiences significant subsidence. In order to use tidal data from the tide station for tide corrections of the water level loggers, we omitted the linear component from the model, resulting in a predicted tidal signal without the effect of subsidence. Subsidence of the tidal station barely impacts tidal analysis, as analysis of a time series before re-anchoring of the monitoring station yields very similar amplitudes and phase lags for each constituent.

6.5.7 Sediment deficit calculation for Semarang

Historic data that report coastline aggradation along the Demak coast were used to estimate the original sediment supply to the system. The coastline of Demak was situated much further land-inward than today. Historic literature reports that the old town of Demak, now situated 12.5 km land-inward, was a coastline port in the sixteenth century (Bird and Ongkosongo, 1980). The coastal plain between Demak and the Muriah Vulcano to the North used to be a sea straight, and did not become a plain until at least the middle ages (Van Bemmelen, 1949). Van Bemmelen further reports that the town of Jepara, at the base of mount Muria became an important port for rice export produced by the Demak region from the 16th century onward. It is therefore likely that the coastal plain of Demak only became inhabitable from the 15th century onward. More recent data, documented on drawn maps from 1908 and 1937, also still show coastline aggradation of 265 m East of Semarang city between these years (Marfai *et al.*, 2008). If we assume that land subsidence around this time was still minimal, coastline aggradation would have been approximately 9 metres per year. Assuming coastal profile of 1/1000, this means that 9 mm/y of vertical accretion was the pre-subsidence sediment supply.

6.6 Appendices

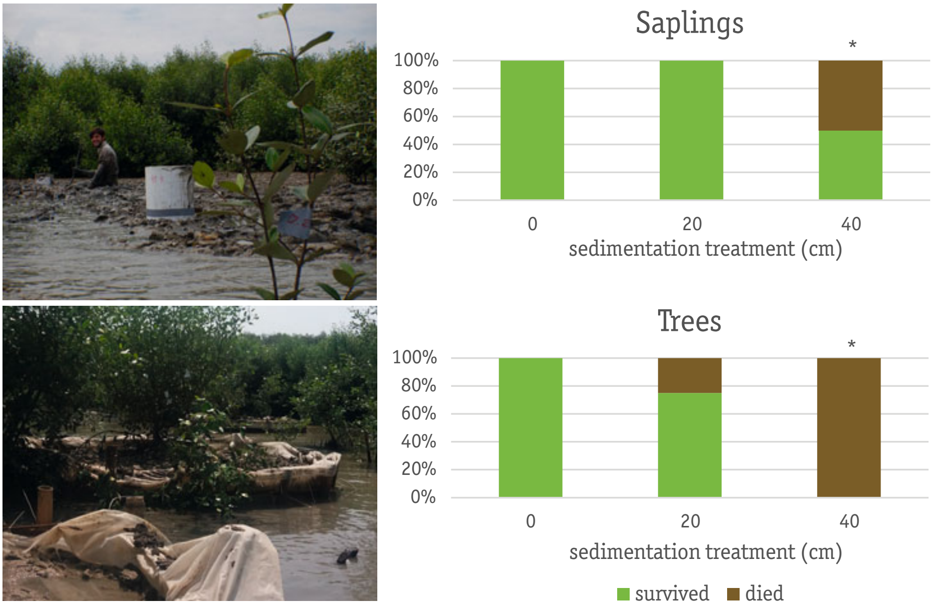

Figure A6-1: *Saplings and young mangrove trees with pneumatophores can survive sedimentation events of 20 cm per event. 40 cm of sudden sedimentation caused survival rates that were significantly lower than the expected survival of 90% * in both saplings and young trees.*

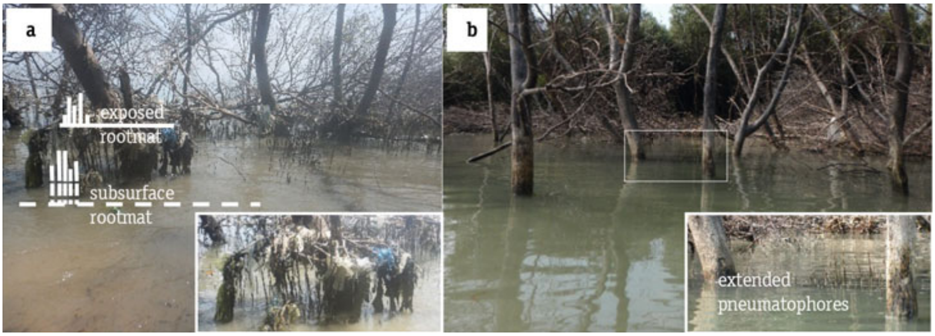

Figure A6-2: *Two subsiding mangrove fringes in which tree mortality occurred following erosion. The erosion reveals morphological adaptations in the root-systems of mangrove trees at these subsiding sites.* ***a.*** *A subsiding mangrove fringe with evidence of multiple root mats per tree. This site has been subject to subsidence, sudden sedimentation and later erosion. The exposed root mat is a remnant of the past sedimentation event in which the existing pneumatophores where buried and trees responded by growing new pneumatophores from fresh cable roots in the top of the new sediment layer. The excess sediment has recently disappeared during erosion, revealing the secondary root mats and even reveals some of the older pneumatophores, which are attached to an older subsurface root mat that now still anchors the tree.* ***b.*** *A subsiding mangrove fringe where only erosion has occurred. These trees do not have multiple root mats, only extended pneumatophores to keep up with the rising water level.*

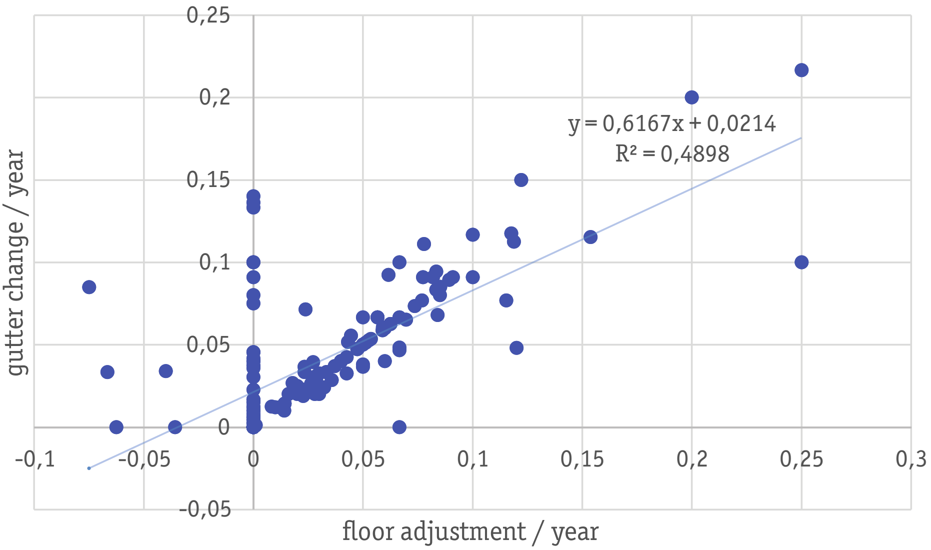

Figure A6-3: *Correlation between the two ways of calculating house subsidence rates.*

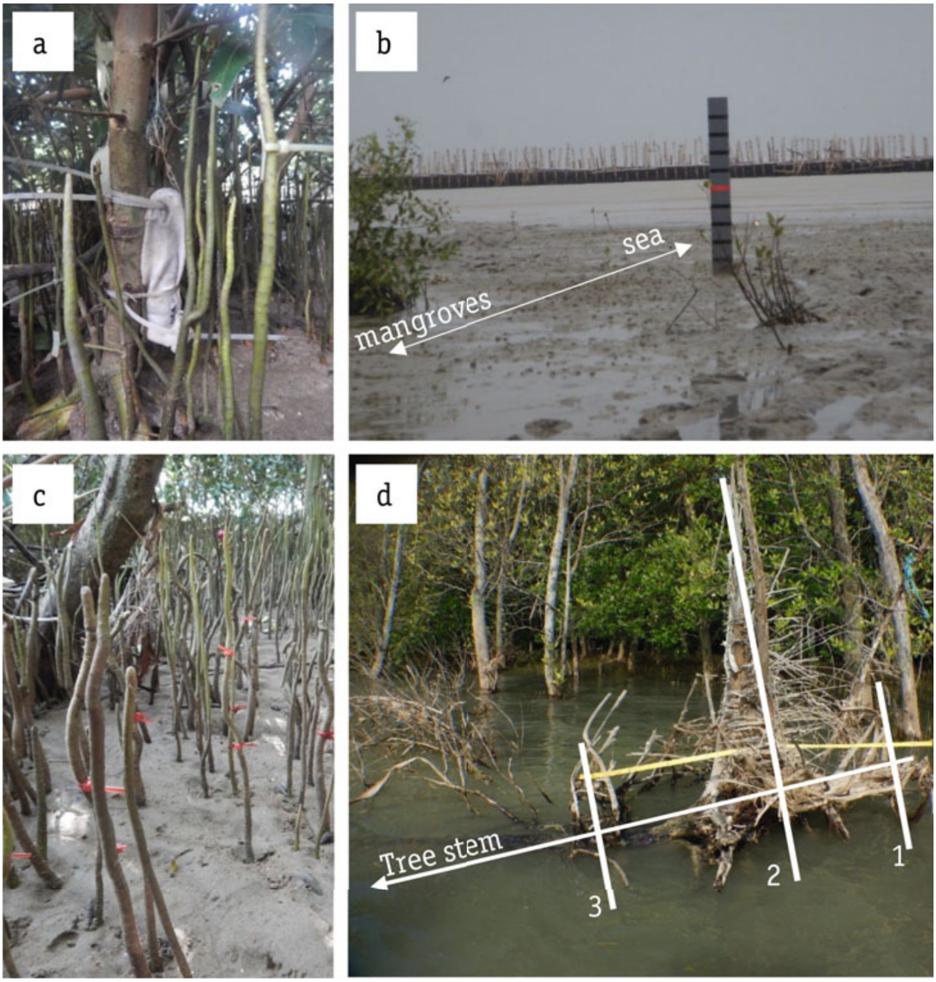

Figure A6-4: *methods applied in the field to monitor effects of subsidence on the mangrove fringe and the bare foreshore.* ***a****. Camouflaged pressure sensor deployed on a young mangrove's tree trunk around 20 cm from the bed.* ***b****. Sedimentation pole approximately 50 meters seaward of the mangrove fringe.* ***c****. Pneumatophore markings in the form of small red cable-ties applied at 10 cm from each root's tip and 10 cm from the bed at baseline.* ***d****. Example of a dead mangrove tree with multiple distinguishable root-mats (1 to 3).*

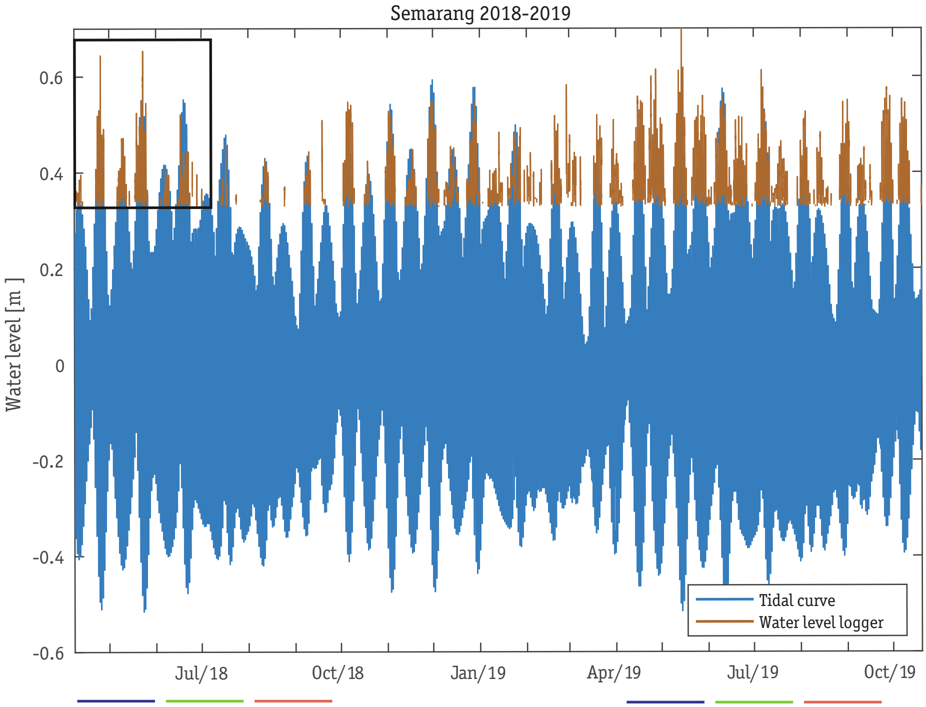

Figure A6-5: *Raw data of one of the water level loggers, fitted to the tidal curve derived from the tide station of Semarang harbour (Flanders Marine Institute (VLIZ); Intergovernmental Oceanographic Commission (IOC), 2021). The black box indicates the first three months that were used to fit the logger to the tidal curve using average daily inundation time. The subsidence of this station was determined using the average tide corrected mean water level per day measured by the logger during the timeframe indicated by the blue line below the x-axis in 2019 with the same period (blue line) in 2018. To validate this trend, the same was done for a timeframe during the mid-dry season (green line below the x-axis) and the late dry season (red line below the x-axis).*

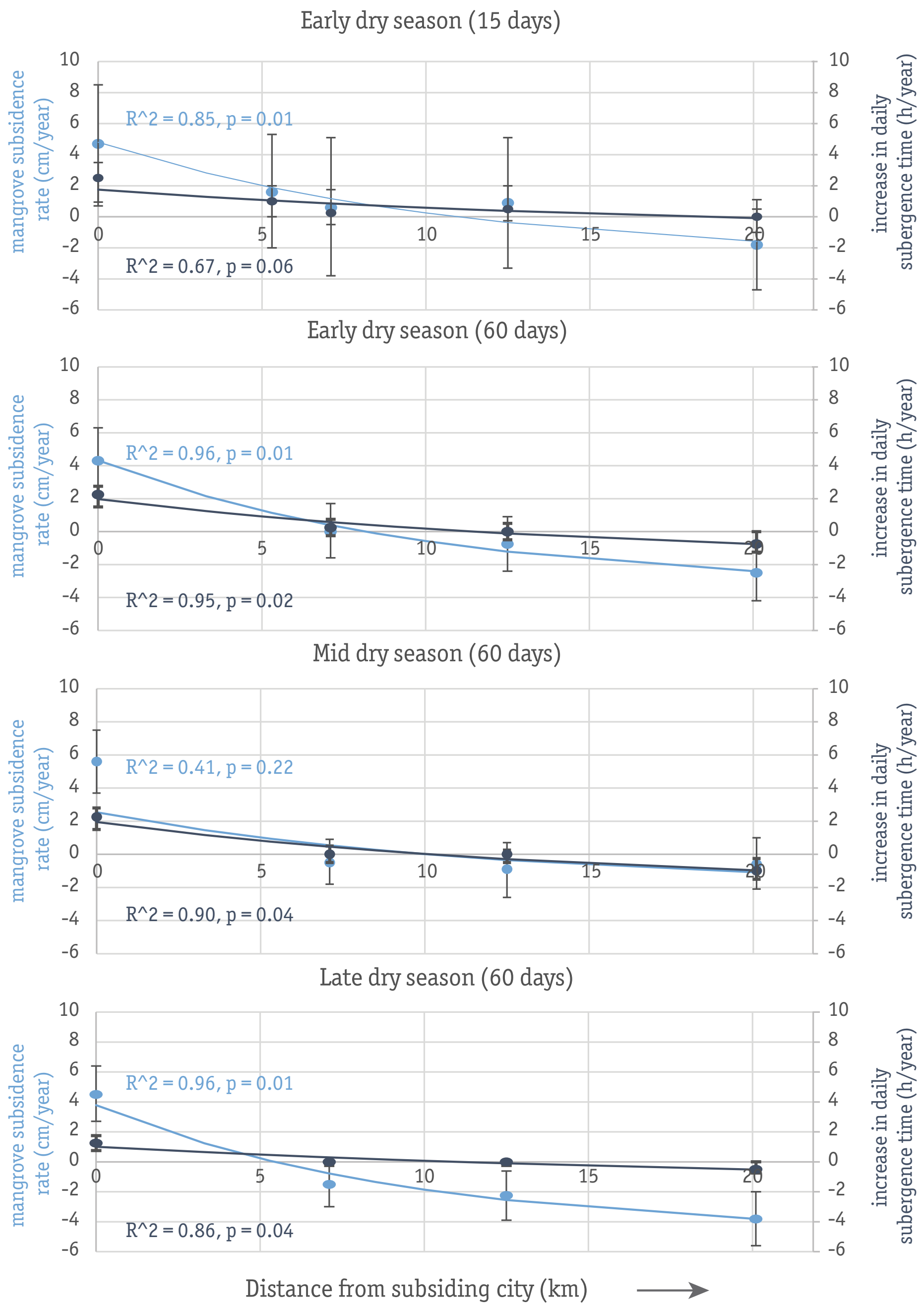

Figure A6-6: *Validation of mangrove subsidence rates along the subsidence gradient, using different dry-season time windows to test for significant water level differences (error bars are 95% confidence intervals) between 2019 and 2018.*

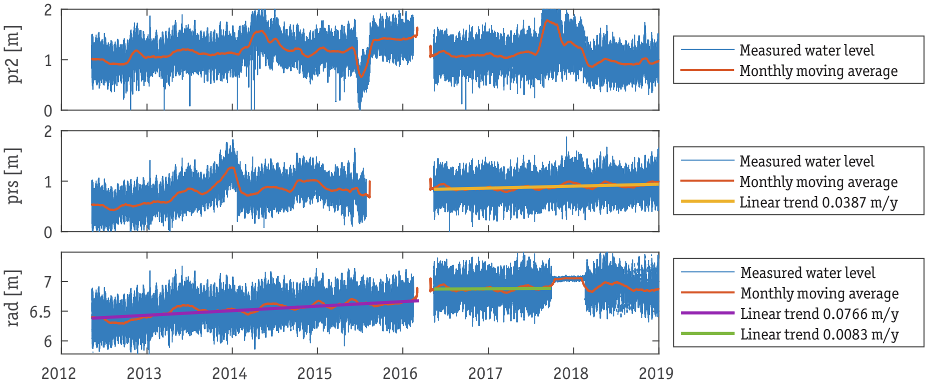

Figure A6-7: *Water levels as measured by the three sensors of Semarang's tide station. The radar signal shows a linear increase in water level of 8 cm per year before the data gap in 2016. After the data gap the linear increase in water level has decreased to 8 mm per year. It is unclear what has happened during the data gap, but it seems plausible that the station was anchored at a deeper sediment layer.*

Synthesis

CHAPTER 7

Synthesis

In this dissertation I sought to elucidate how mangroves can best be restored along vulnerable coastlines based on three main questions. (1) What are drivers of natural mangrove dynamics along degraded mangrove-mud coasts? (2) How can that knowledge be used to assist mangrove expansion? (3) How do indirect anthropogenic stressors affect survival of established and restored mangrove vegetation?

In order to answer these questions, I structured the main results from each chapter in the discussion based on the location where mangrove restoration is attempted: the seaward edge or the landward edge of a mangrove fringe (Figure 7-1). For each mangrove edge, I first discuss natural factors (negative and positive) that were identified in chapter 2 & 3 to drive establishment and survival. Factors that drive natural landward edge dynamics could not be investigated in Demak but may be important to understand general limits to landward mangrove expansion, and thus restoration. Natural drivers of mangrove dynamics along the landward edge are therefore discussed based on literature. After exploring the natural processes, I discuss how anthropogenic factors (positive Chapter 4, and negative (Chapter 5 & 6)) influence these processes and thereby potentially drive establishment and survival along each edge in degraded systems.

Within each subcategory (e.g. seaward edge & natural processes that have a positive influence on mangrove expansion), I first briefly reiterate the main conclusions of the chapters that apply to that subcategory (superscript numbers in Figure 7-1). I then discuss relevant observations from the field and experimental results that apply to those topics but were outside the scope of the associated chapters. Finally, some potentially relevant topics were not addressed in any of the chapters. In those cases, I turn to literature to fill in the gaps or to remote sensing to estimate how important the factor might be for mangrove restoration (Figure 7-1 factors with question marks). Conclusions are drawn where possible, or opportunities for further research are indicated.

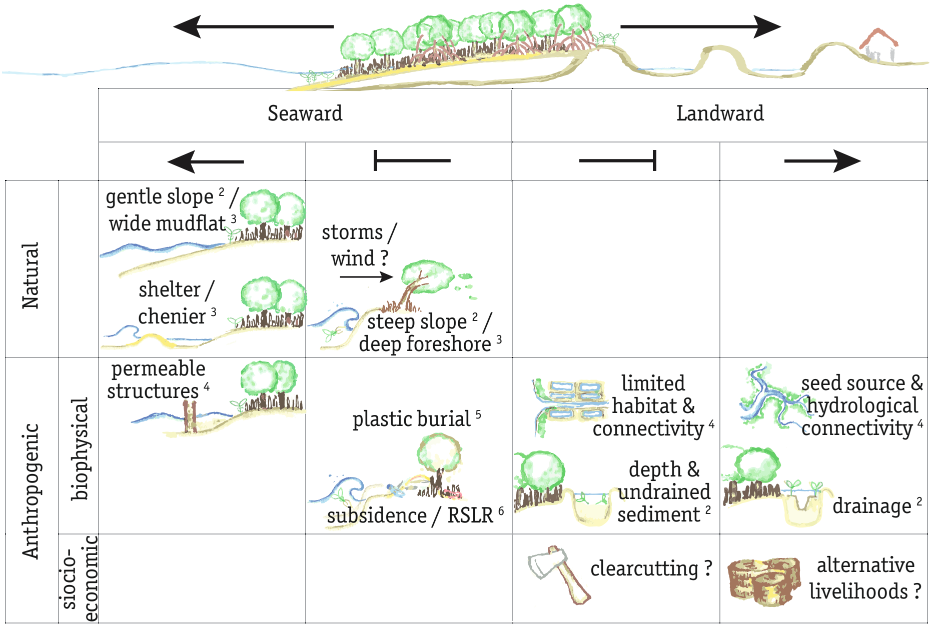

Figure 7-1: *Factors (positive → and negative —I) presented in this dissertation that drive seaward and landward edge dynamics of mangrove greenbelts along vulnerable rural coastlines. Relevant chapter numbers are indicated in superscript with each factor. Factors that were outside the scope of the chapters, but potentially important drivers to understand for greenbelt restoration purposes are indicated with a question mark. These will be elaborated on in the discussion.*

7.1 Seaward edge dynamics

7.1.1 Natural processes that drive seaward edge dynamics

Mangrove-mud coasts are often found to be dynamic along the seaward edge in natural systems (e.g. Anthony et al., 2010; Balke et al., 2013; Lugo, 1980). Understanding the drivers of these natural dynamics in degraded systems, may in the first place, help to identify foreshore characteristics that can be targeted to promote mangrove expansion in seaward direction. Secondly, a profound understanding of the factors that drive natural mangrove die-back, such as storms, may help predict and account for natural dynamics in the design and maintenance of mangrove greenbelts for coastal resilience.

Mangrove retreat along the seaward edge

Tree survival along the seaward edge

The resilience of mature mangrove trees to the forces of the wind and waves is an important aspect in understanding and predicting natural mangrove retreat. Receding mangrove forest are frequently observed as a natural phase in forest dynamics (Augustinus, 1978). Chapter 3 revealed how the absence of sheltering features increased the probability of landward mangrove retreat. However, hurricanes and cyclones have been known to cause mass mortality events, destroying up to 100% of the trees in a forest, not only in the fringe (McCoy *et al.*, 1996; Krauss and Osland, 2020). Understanding driving conditions of mangrove die-back, can give a better understanding of the potential resilience of mangrove greenbelts for coastal protection. For instance, what amount of mangrove retreat could be expected for a once in a 100 year storm? In terrestrial systems, tree stability has already been studied for decades, either in light of tree safety in urban areas or windthrow in commercial silviculture (e.g. (Somerville, 1979; Coutts, 1986; Read and Stokes, 2006; Achim and Nicoll, 2009; Gardiner, Berry and Moulia, 2016)). For terrestrial trees, the overturning moment of a tree is often described as a function of wind velocity, air density, aerodynamic drag, crown surface area and its distribution along the stem, and the natural frequency factor (which is the natural frequency at which a tree will sway in the wind) (Wessolly and Erb, 1998). All these factors will also play a role for mangrove trees. In addition, mangrove canopies are also subject to wave forcing at high tide, especially during storms.

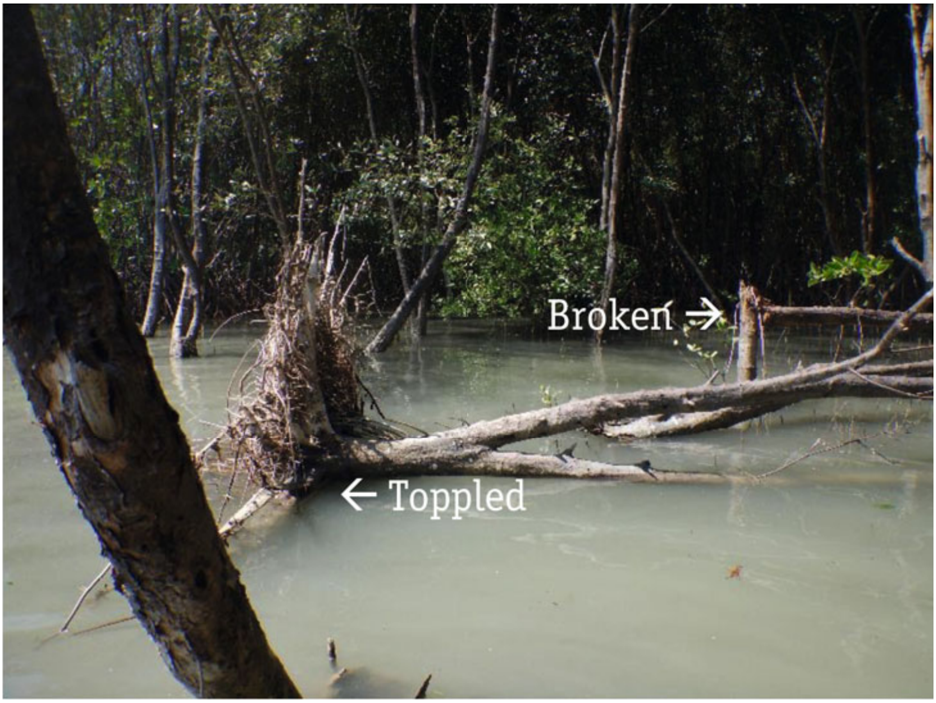

Figure 7-2: *Two dead trees at the seaward edge of the mangrove fringe along Demak's coastline. The mechanism of failure was different for both trees. The tree in the front has been uprooted by the wind, and the tree in the back is still anchored at its roots, but the stem has snapped.*

This is actually viewed as a service by mangroves for coastal protection, as was revealed in the chenier transect of Chapter 3 (Figure 3-4), where wave attenuation through the vegetated chenier was much more pronounced during the high water levels of the storm surge (i.e. high enough for the water to go through the shrub like canopy) than during regular high tide (i.e. where the water only passed through the pneumatophore field). However, the reverse effect (i.e. the effect of waves on the tree) has not yet been investigated, which is another factor that will play a larger role on tree stability in mangroves than in terrestrial forest is erosion. When mangrove soils erode laterally, cliffs are formed at the border between vegetation and sea (Othman, 1994), trees that are located at this border miss anchorage capacity. When erosion is more gradual and rinses the sediment out from between the roots, a process called sheet erosion (Semeniuk, 1980), trees loose anchorage in terms of depth. In both cases, trees become more prone to wind damage (Gardiner, Berry and Moulia, 2016). Another gap in knowledge is the mechanism of failure when mangrove die-back occurs. When do trees topple and when does the stem break? In the field, we have observed both types of tree death, sometimes within meters from each other (Figure 7-2).

We conducted a field experiment with tree sized mangrove mimics in the Netherlands to shed some light on factors that influence the overturning moment of mangrove trees. We placed 60 mangrove mimics with various root morphologies (cable roots, stilt roots and various lengths), at various depths (mimicking the effect of erosion) and at sites with two different sediment types (mean (±SD) grainsize sandy site: 153.54 µm (±11.62), grainsize muddy site 35.96 µm (±12.36)), and did a pulling test with a winch on each mimic to test its stability (Raven, Bijsterveldt and Hespen, 2020). Sediment type was the most important factor determining the stability of the tree. In sand, the mimics required much higher forces to reach 15 degrees of deflection than in mud. What is more, the mimics in sand frequently broke before they had tilted more than 50 degrees (visible in Figure *7-3*c as the truncated lines). In contrast, the mimics that had been placed in muddy sediment, required significantly lower forces to deflect to 15 degrees, and were then pulled out of the sediment (i.e. uprooted). Although these findings lift a tip of the veil with regard to factors that contribute to tree stability during storms, pulling experiments in the field need to be done to obtain reliable overturning moments for mature mangrove trees. Overturning moments with various baselines of erosion would give valuable information to predict the speed and extend of mangrove retreat under severe storms, an interesting topic for future experiments, and important information to obtain for the maintenance of mangrove greenbelts.

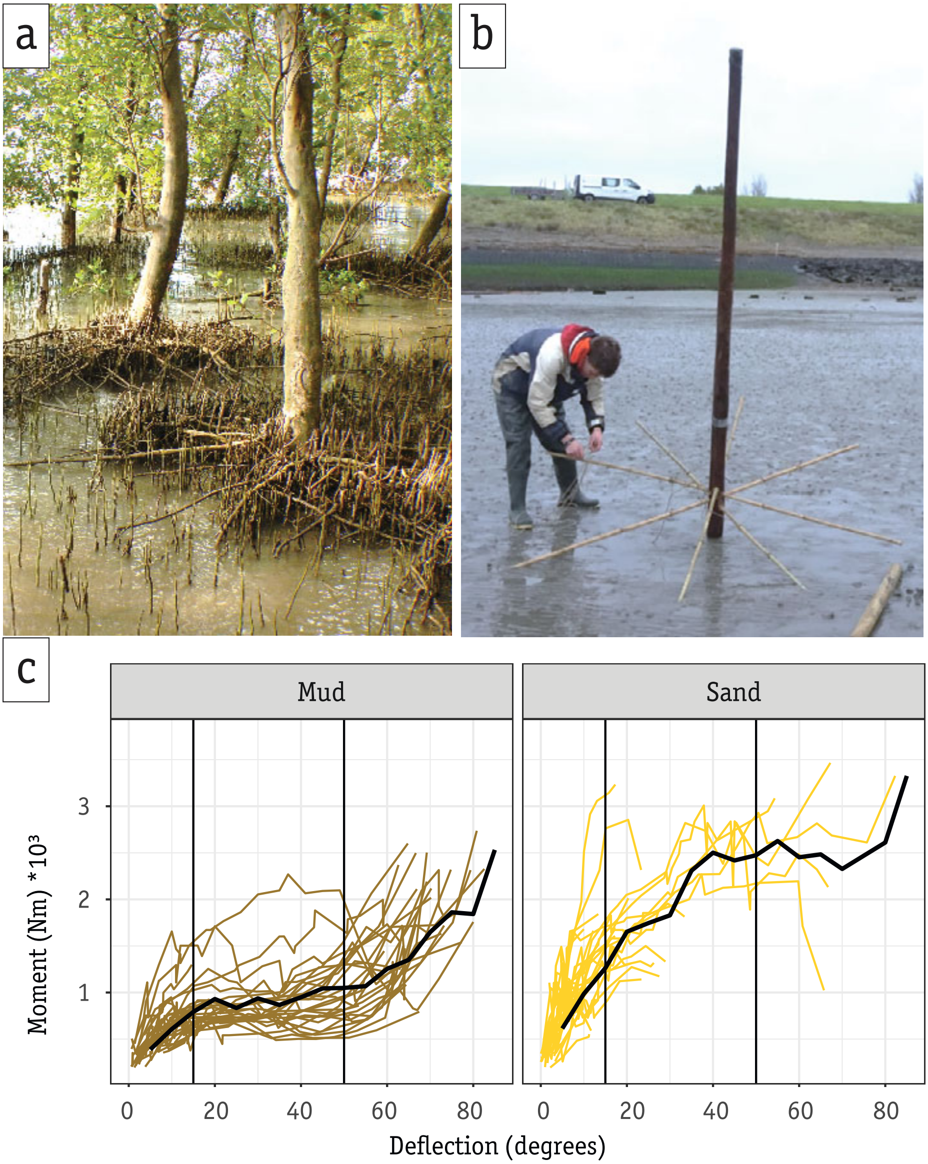

Figure 7-3: *Mangrove tree mimic uprooting experiment.* ***a.*** *exposed roots of Avicennia spp. in the field after sheet erosion.* ***b.*** *a mimic of (a) in the Netherlands before placement in the mud. Cable roots were mimicked with bamboo rods (star shape) and larger hairroots were mimicked by interconnecting those bamboo roots with sisal twine.* ***c.*** *Moment (force * distance) needed to deflect mimics by n degrees in sand versus mud, adapted from (Raven, Bijsterveldt and Hespen, 2020).*

Mangrove expansion along the seaward edge

Natural drivers of seaward edge expansion are also important to identify along degraded coasts, as they are the key to understanding how mangroves may be expanded at retreating locations where a wider greenbelt is required for coastal resilience. Chapter 2 revealed that at seaward sites, local slope was an important factor in establishment; A steep slope, like a cliff at the seaward edge did not allow for seedling establishment on the forest floor, most likely due to wave forcing by breaking waves. In contrast, a gentle slope that provides habitat and allows for more gradual wave-energy loss seaward of the mangrove edge increased the chances of seedling presence. Chapter 3 revealed that the width of this gentle slope greatly determined the chances of seaward mangrove expansion. For instance, the likelihood of seaward expansion increased from 17% to 44 % between a mudflat of 10 m versus a mudflat of 370 m wide. These numbers are already useful when foreshore restoration is attempted to promote seaward mangrove expansion. However, we did some additional field observations and experiments that might give useful insights for seaward mangrove restoration.

Habitat availability: cheniers as mangrove nursery

Along eroding shorelines, the cross-shore slope between the mangrove edge and the foreshore is generally steep (Winterwerp *et al.*, 2013). This means that there is very little habitat for natural seedling establishment along the seaward edge. Cheniers, although detached from the shore, can develop well above mean sea level (Tas, van Maren and Reniers, 2020), which means that they may function as suitable mangrove habitat. Cheniers were indeed observed to provide habitat for mangrove establishment in the field. Especially, the landward side of cheniers appears to be a popular landing site for mangrove propagules (*Figure 7-4*). In fact, cheniers may slow down the loss of propagules to the open ocean, especially since cheniers in this region are often observed in front of creek mouths. These cheniers stretch parallel to the shore for multiple kilometres, essentially creating a lagoon on their landward edge. The water draining from this lagoon -and all the inland aquaculture area- during ebb flow slows down significantly by the long sandy barrier (Tas *et al.*, 2022), this likely extends the stranding window of propagules substantially. Cheniers might therefore also have a retaining function for propagules, a factor that was not investigated, but might have merit to explore further in future research. Propagules that were observed to strand on a chenier, often germinated and grew into seedlings. Sand would in principle allow for sturdier anchorage of seedlings than mud, as it is much less erodible. Indeed, in situ uprooting and critical erosion depth experiments in the field revealed that 50 cm tall (approx. 90 days old) seedlings of *Avicennia* species could be uprooted from sand with pulling forces of 84 N. In mud, uprooting a similar sized seedling took much less effort (28 N, Figure 11 in (van Rees, 2019)), suggesting that cheniers could provide a suitable habitat despite their exposed position. However, chenier colonization only occurred at two locations in the area over a period of 7 years (*Figure A 3-5*). These events seem to have occurred only at locations where another chenier was already developing further seaward, and during the same year that mangrove expansion occurred throughout the whole area (Chapter 3)(van Bijsterveldt *et al.*, 2023): a wet season with exceptionally low wind speeds. Chenier colonization therefore seemed to be relatively rare, not in space, but in time.

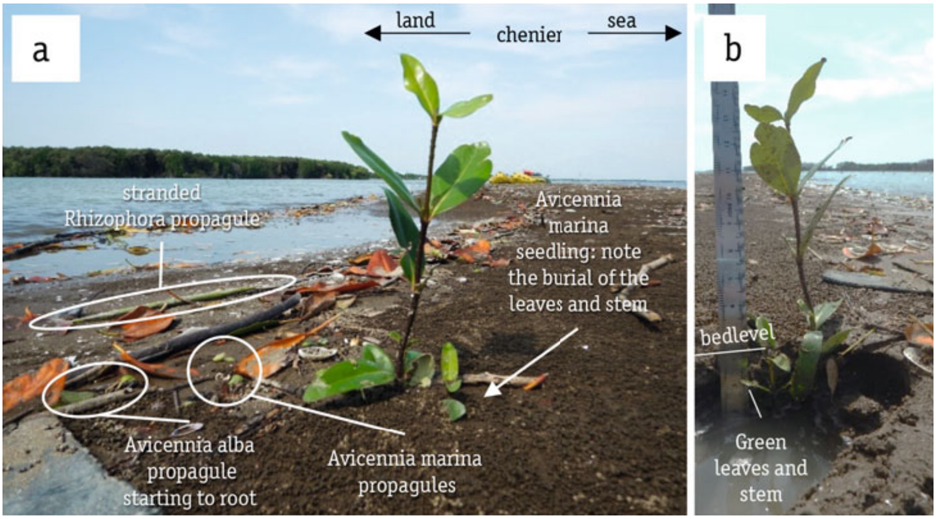

Figure 7-4:* a.** *Stranded mangrove leaves and propagules of various mangrove species, and a seedling of the species Avicennia marina on the landward side of the chenier, illustrating that the chenier itself is a natural establishment site for mangroves.* ***b. *The same seedling but photographed from a different angle; the green subsurface leaves that have been dug out for the picture suggest that this individual was recently buried with at least 4.5 cm of sediment.*

Seedling survival in space and time

The fact that mangrove colonization at cheniers was only successful in the year that mangrove expansion throughout the whole coastal zone was successful, actually indicates that the location of seedling establishment in cross-shore direction (and thus the maximum wave forcing that a seedling experiences) has very little direct influence on seedling survival. This sounds counter intuitive, but it is actually consistent with the discrepancy between the dislodgement forces that were measured in the field (84 N & 28 N resp. for a 50 cm seedling in sand and mud) and the maximum wave forcing that equally sized seedlings would have experienced at the most exposed sites, which were no higher than 0.35 N (van Rees, 2019). Even if we use the wave parameters measured at the chenier during the December 1st storm surge (waveperiod = 8 s, H = 0.45, z = 0.7), the wave forcing on a (assumingly rigid) 50 cm seedling at that site

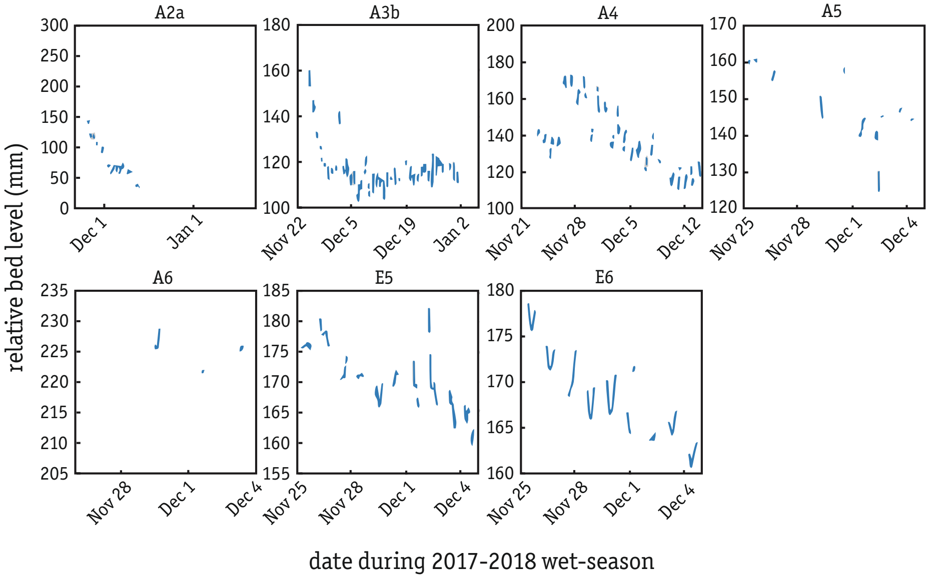

Figure 7-5: *Relative bed level change per station along the transects of Chapter 3.*

does not exceed 1.5 N, which is not even close to the dislodgment forces needed to uproot seedlings anchored in sand (84 N), or even mud (28 N). It therefore appears that sediment dynamics is a much more important factor driving seedling dislodgement than direct wave forcing. There is a clear gradient in terms of sediment dynamics in cross-shore direction, with the largest changes in bed-level at the steepest and most exposed sections of the profiles (Figure 7-5). While seedlings are well adapted to rapid accretion (Chapter 6), erosion is a different matter (Balke *et al.*, 2011). A couple of decimeters of erosion over the course of a few days are not uncommon in the field (Figure 4 (Tas, van Maren and Reniers, 2020) & Figure 7-5), which would significantly reduce the anchoring capacity of a 50 cm seedling. Assuming a 1:1 root-shoot ration, 20 cm of erosion, would effectively result in a 70 cm tall seedling exposed to wave forcing, with the anchoring capacity of a 30 cm tall seedling, making the seedling much more vulnerable to dislodgement from wave forcing. In addition, seedling mortality is not only determined by total dislodgement. Toppling of seedlings is achieved before total dislodgement in critical erosion depth experiments (van Hespen *et al.*, 2022) and is likely fatal as well at exposed sites. Another mechanism of failure under the continued back and forth forcing of the waves is tissue damage. The seedling might perhaps not be dislodged in such a case, but it’s stem might break, or it might be defoliated. Although the precise composition and magnitude of individual factors that determine failure of establishment along the seaward mangrove edge remain unknown, it is clear that the combination of increased wave forcing and erosion during the 2017 wet-season was fatal for the seedlings that had established that season regardless of their cross-shore location (Chapter 3). For instance, even the most landward station of the chenier transect lacked saplings, seedlings that would have settled in the season prior to the field visit. Similarly, the GIS analysis in Chapter 3 revealed that conditions had not been favorable for seedling survival along the seaward edge of the entire study area in six out of the seven years that were studied. Longer timeseries of high-resolution imagery might reveal other years that were suitable for seaward mangrove expansion and thus shed more light on the tipping point between success and failure of seedling establishment on the seaward mangrove edge but the knowledge that seasonal differences are more important may be useful for restoration purposes, as the timing of a restoration effort could greatly determine the chances of success.

7.1.2 Anthropogenic factors along the seaward edge

The natural processes described in the previous sections can be influenced both negatively and positively by human interference. Positive human interference involves interventions that support mangrove expansion along the sea-ward edge, as discussed in Chapter 4. Negative human influences on the seaward edge are seldom direct effects, such as the direct removal of mangroves. More often, mangroves are impacted indirectly. For instance, through interference by regional stressors (e.g. as described in Chapter 5 and 6) with one of the natural processes described in the previous sections. Gaining understanding of how anthropogenic stressors interfere with natural mangrove dynamics, and what the size of the impact is, can help communicate the importance of addressing these issues, which will help to make mangrove restoration efforts more successful and long-lasting.

Negative anthropogenic influence on edge dynamics

Two common human induced stressors that are poorly addressed in coastal regions worldwide are household pollution and land subsidence (Syvitski *et al.*, 2009; Jambeck *et al.*, 2015; Nicholls *et al.*, 2021). Chapter 6 revealed how land subsidence caused both the vegetated and bare foreshore to sink, which exacerbated erosion in locations where sediment deposition could not keep up with subsidence rates. Subsidence thus made the foreshore slope steeper (Chapter 2), and shelter width smaller (Chapter 3). A deepening foreshore allows ever larger waves to pound on the seaward mangrove edge. Subsidence can therefore leave the mangrove fringe even more vulnerable to storms, which are expected to increase in frequency and severity due to human induced climate change (Knutson *et al.*, 2010). The combination of land-subsidence and the absence of sediment to compensate for it will likely cause mangroves to continue to retreat landward.

The fatality of a prolonged subsidence-sediment mismatch

Indeed, the sites with ongoing rapid lateral erosion were situated close to the subsidence epicentre (Figure 7-6, 0 & 3.3 km). Interestingly, the site at 3.3 km from Semarang showed more rapid lateral erosion than the site closest to Semarang. The foreshore at the 3.3. km site was consistently measured to be deeper throughout the study period than the foreshore in Semarang. This suggests that the mismatch between subsidence rates and sedimentation rates was larger at this site. Based on the absence of buildings and existing subsidence heatmaps (Kuehn *et al.*, 2009; Abidin *et al.*, 2013; Chaussard *et al.*, 2013; EO4SD, 2017), it is unlikely that this site would sink faster than the site at Semarang. A more likely explanation for the subsidence-sediment mismatch, and thus the more rapid lateral erosion at this site, would be the absence of a local sediment source (creek or canal). It appears that even the mangrove trees that stand in the basin of this forest suffer from the combination of land subsidence and low sediment supply. Figure 7-6 shows how the trees at this site (3.3 km from subsiding city) are turning brown. We have not been able to verify the cause of this apparent die-back in the field, but the site is relatively remote and it has a protected status to facilitate eco-tourism. It therefore seems unlikely that trees were purposefully killed for pond conversion.

The apparent mismatch between available sediment and sinking bed-level seems a likely reason for tree die-back at this site. Bed level monitoring and long term sea level change measurements in the Indo-Pacific region have already led to predictions of mangrove drowning commencing from “as early as 2070” onwards (Lovelock *et al.*, 2015). However, actual drowning of mangrove forests has only been reconstructed from paleo data (Saintilan *et al.*, 2020). If the trees at 3.3 km are indeed drowning, this would be the first reported case of present-day mangrove die-back caused by relative sea level rise. A sad observation, although it is encouraging to realize that the tipping point for short-term mangrove drowning likely lies in the order of cm per year rather than the reconstructed mm per year previously reported in literature (Saintilan *et al.*, 2020).

Mangrove resilience to subsidence through “floating”

Another encouraging find in Chapter 6, is the fact that mangroves appear to sink less rapidly than land. Of course, land subsidence in this case was measured by the floor elevation corrections that were done in houses to cope with the floods that became ever more frequent. Therefore, one possible explanation for the discrepancy would be that concrete and brick houses weigh more heavily on the subsiding and compacting sediment then trees, and therefore sink faster. Mangrove trees have shallow root systems (P Barry Tomlinson, 2016) that are densely intertwined in a mature mangrove stand, effectively forming one big shallow root plate that anchors the forest. Mangrove roots are extremely light weight, and consist mostly of aerenchym, and thus air (Scholander, Van Dam and Scholander, 1955). Even the wood of tree stems can be inherently buoyant, depending on the species (Rumbold and Snedaker, 1994). In the field, mangrove sediment inside the basin of mature mangroves was often so soft that a person would sink in until their hips. In a way, mangroves thus appear to be "floating" on this soft sediment. A large-scale network of Sedimentation Erosion Tables (SET) throughout Indo-Pacific mangrove forests revealed that shallow compaction (i.e. compaction of the sediment layer between the depth of living roots and the depth of the first layer of resistance such as consolidated sediment or bed-rock) does not exceed 10 mm / year in 65 % of the cases, of which 15% is actually subsurface expansion rather than compaction (Source data, extended Data Figure 1 in Lovelock et al., 2015). Unfortunately, the SETs that I installed in Demak to understand the difference between mangroves and foreshores in terms of accretion, shallow compaction and deep subsidence were lost during the campaign; Any relation between mangroves and soil compaction therefore remain unclear.

Figure 7-6: *Lateral erosion at the three sites closest to the subsidence city.*

Nevertheless, the data from Lovelock et al. (Lovelock *et al.*, 2015) imply that nor the weight of a mangrove forest nor its evapotranspirating capacity cause the underlying sediment to drain and compact much. If soft sediment below mangroves drains and evaporates poorly, as also seemed to be the case based on field observations, and mesocosm experiments we conducted with seedlings (Broeze, 2017), then the removal of mangroves could expose large amounts of easily erodible sediment. A better understanding of the interaction between mangroves and soil compaction could help predict erosion volumes when mangroves are lost, and thus help communicate the importance of these ecosystems to prevent further degradation.

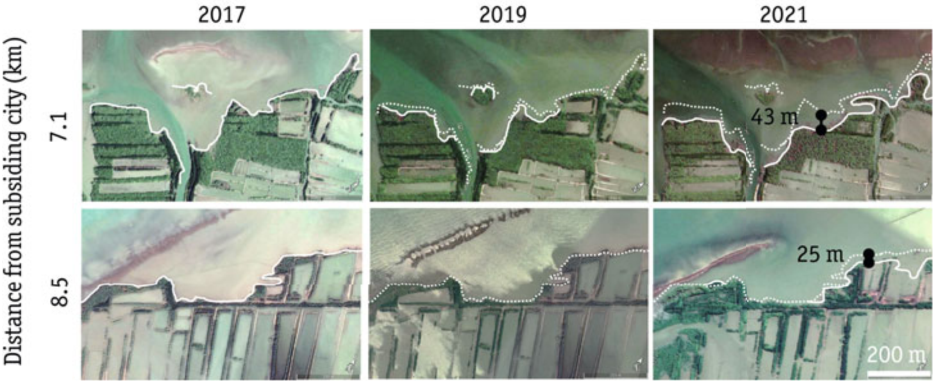

Figure 7-7: *Lateral erosion at non-subsiding sites.*

Subsidence and plastic, a deadly combination

Another important human-induced stressor to mangrove growth and survival proved to be the presence of plastic from household waste (Chapter 5). The abundance and continued accumulation of plastics in mangroves are well known (Costa *et al.*, 2011; Sandilyan and Kathiresan, 2012; Smith, 2012; Debrot *et al.*, 2013; Ivar do Sul *et al.*, 2014; Li, Tse and Fok, 2016), and it has even been suggested that trapping of household waste could be an ecosystem service that mangroves provide. However, the plastic suffocation experiment in Chapter 5 revealed that when all a tree's pneumatophores are covered by plastic debris, this can cause tree death. When a tree's pneumatophores are partially covered, leaf loss appears to be the response. Forest floor monitoring throughout the research area further revealed that a plastic cover of 50 % was not uncommon, and even reached 75 % at some locations (Figure 5-4). Moreover, plastic was found to be buried in the sediment at all locations, trapping the plastic in the forest. Although there was no significant difference in surface plastic cover in cross-shore direction, especially the trees in the fringe seemed to suffer from it (*Figure 7-8*a). This was most likely because of the difference in sedimentation rate between fringe and basin; Trees in the fringe trap sediment that is loosened from the foreshore bed. This rapid deposition of sediment fixates plastics and debris that were already trapped by aerial roots (Figure 7-8a), a process that was most clearly visible at the most rapidly subsiding and eroding sites (0 & 3.3 km from Semarang). It appears that the layers of plastic and sediment significantly stress trees in the fringe even before the erosion front arrives and gives the trees along the seaward edge a final blow (Figure 7-8b). This shows that the combination of these two common, and generally poorly addressed anthropogenic stressors have a detrimental effect on the seaward mangrove edge. Without addressing these issues, mangrove restoration along the seaward edge becomes extremely challenging.

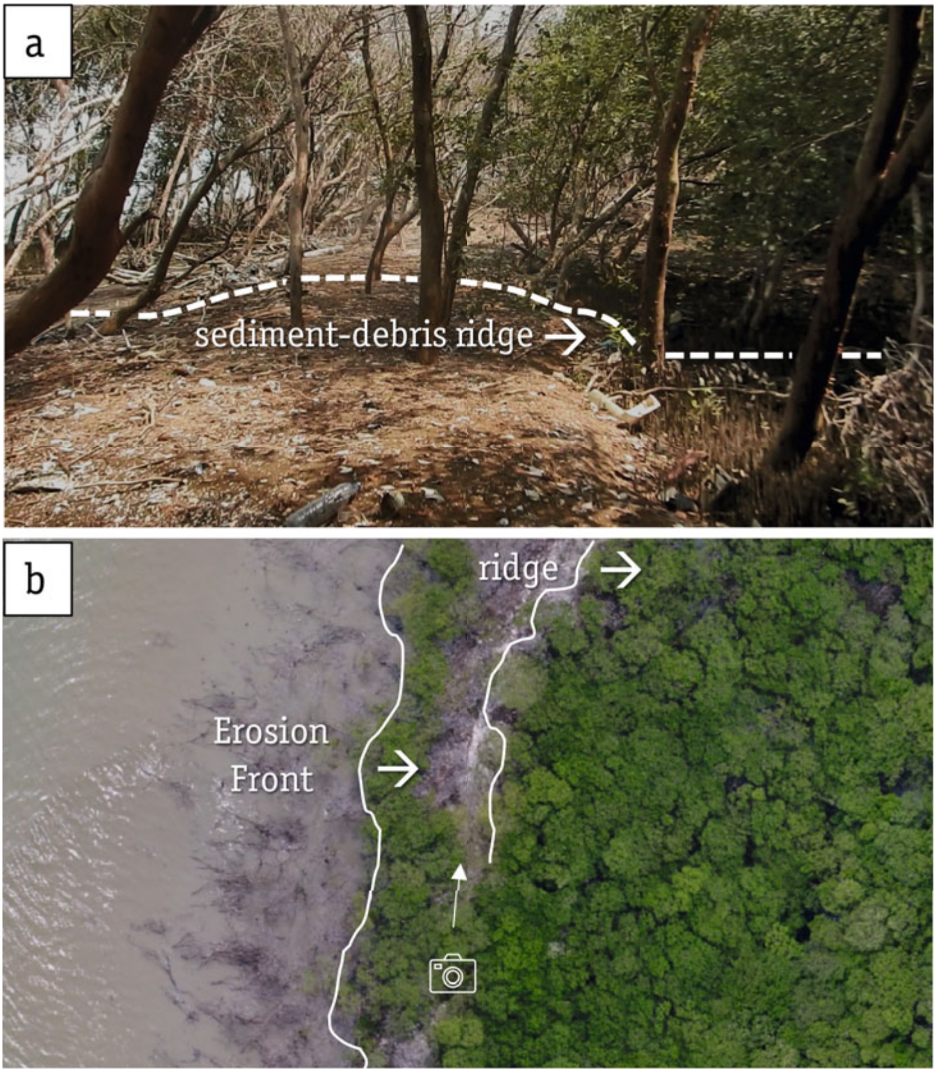

Figure 7-8: *Mangrove trees under stress from plastic and (subsidence induced) erosion.* ***a.*** *Sediment deposited on top of in plastic entangled aerial roots. Note how the trees inside the ridge have a low canopy cover. The sediment in the ridge originates from the foreshore bed seaward of the fringe and is deposited within the first 20 m of viable mangrove fringe. As the outer trees die and the old forest floor erodes (erosion front), the ridge also moves further land inward.* ***b.*** *Drone mosaic (altitude ~100 m) of the site in which both the erosion front and the sediment-debris ridge are clearly visible, the direction of (a) is indicated.*

Positive anthropogenic influences

While seaward mangrove dynamics are heavily influenced by anthropogenic factors, Chapter 4 also shows that it can be influenced for the good. Traditionally, efforts to expand mangrove cover in seaward direction involved large-scale seedling planting (Wodehouse and Rayment, 2019). However, without regarding the natural recruitment process and its limitations, planting efforts are set-up for failure (Chapter 4). When limitations of natural recruitment are accounted for, planting is often no longer necessary. Restoration of a suitable mangrove habitat (EMR) has therefore demonstrated to be more affective (Chapter 4). In eroding areas where land subsidence is not an issue, the placement of permeable dams has proven a valuable restoration measure for mangrove habitat (Winterwerp *et al.*, 2020). In Demak, the structures indirectly caused stabilization of the coastline, as they limited ongoing deepening of the foreshore with subsidence, but they were not able to raise the bed level permanently to levels needed for recovery of a healthy mangrove forest. Similarly, sediment amendments on the bare foreshore have a positive influence on seaward mangrove edge dynamics (Cohen *et al.*, 2021), as this shelters the mangrove edge from waves in a similar way as natural processes discussed in Chapter 2 and 3. Sediment nourishments inside coastal vegetation have been suggested to maintain surface elevation of protective greenbelts near large cities where landward migration is not an option (Rogers *et al.*, 2019).

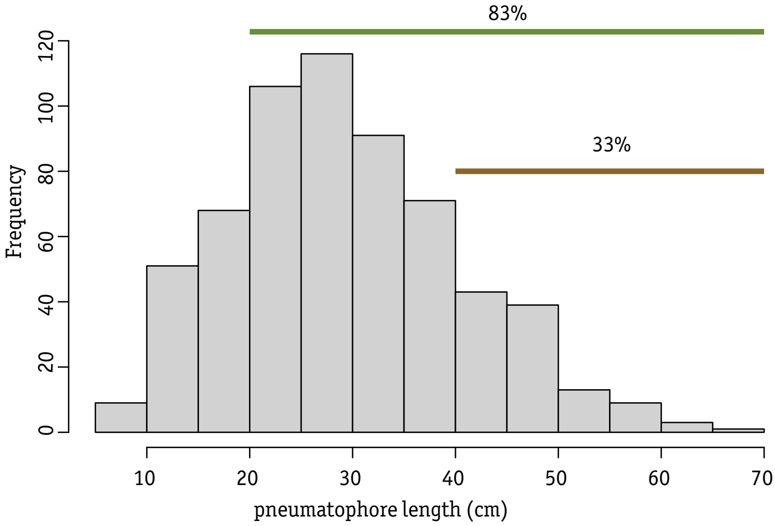

Figure 7-9: *Frequency distribution of pneumatophore length (sediment to tip) of all monitored pneumatophores in the research area for a period of 2 years (dataset from Chapter 6). The percentage of pneumatophores that would still be exposed after hypothetical sediment amendments of 20 cm (non-lethal) and 40 cm (lethal) are indicated.*

These so called thin-layer sediment additions, dredged material sprayed in a layer between 3 and 7 cm thick on top of the marsh surface, have indeed been successful in salt marshes (Ford, Cahoon and Lynch, 1999; Slocum, Mendelssohn and Kuhn, 2005; Stagg and Mendelssohn, 2010; Puchkoff and Lawrence, 2022). In fresh water swamps this method has been less successful (Middleton and Jiang, 2013), causing the death of several *Taxodium distichum* trees at one of the study sites. According to the authors, this was likely because the knee roots of those trees were completely buried by the sediment amendment, which does not bode well for a potential application of nourishments in mangroves. So far, there is no literature on experimental sediment nourishments to compensate for RSLR in mangroves, but there have been accounts of sedimentation events of both natural (e.g. due to floods and hurricanes) and artificial (e.g. marine dredge spoils) origin. These have reportedly led to mass tree mortality (Ellison, 1999), likely because of aerial root suffocation. Although a maximum amount of sedimentation was not mentioned, all events that caused tree mortality were above 10 cm of sudden sedimentation. The length and type of aerial roots were not reported in Ellison’s review of these cases but these root characteristics would likely determine the maximum amount of sediment amendment that a forest can handle.

The *Avicennia* trees that we experimented with in Chapter 6, were able to cope with 20 cm of sudden sedimentation, but died under a sedimentation pulse of 40 cm. The total of 773 pneumatophores of 46 mangrove trees that were monitored in the study area had an average (±SD) pneumatophore length of 29.9 cm (±11.2), of which 645 pneumatophores were larger than 20 cm, and 261 were larger than 40 cm. This suggests that *Avicennia* trees should be able to survive sediment amendments that leave 83% percent of their pneumatophores exposed but will not be able to tolerate nourishments that only leave 34% of their pneumatophores emerged from the sediment (Figure 7-9). The tipping point likely lies somewhere in between but using both the distribution of aerial root length and an 80% “emerged” threshold could be a safe starting point to test if sediment nourishment can help maintain mangrove greenbelts under rapid RSLR.

However, the presence of large amounts of plastic on the forest floor might contra-indicate nourishments, as Chapter 5 revealed that this would increase the likelihood of suffocating the trees. Unfortunately, a regression analysis on the plastic abundance data from Chapter 5 performed by Raven (2020) revealed that the percentage of built environment within a buffer of 200 m around rivers and streams in the catchment upstream from each study site in Demak was positively correlated ($p<0.001$) with the number of plastic items observed on the mangrove forest floor (Figure 7-10). The most polluted areas were thus found at the sites in the most rapidly subsiding area. This is unfortunate, as Figure 6 shows that these sites could potentially benefit from sediment nourishment. This co-occurrence of plastic pollution and subsidence is most likely not unique, as the global distribution maps of Chapter 5 and 6 show that both plastic discharge from rivers (Jambeck *et al.*, 2015), and land subsidence rates (Nicholls *et al.*, 2021) are highest in South-East Asia. Hence, plastic pollution should be addressed first, before attempting to counteract RSLR in mangroves with sediment amendments. Another difficulty with sediment amendments in mangroves is that mangrove coasts have extremely gently slopes and thus shallow waters in the coastal area. This might make it difficult to get the necessary equipment close enough to the vegetation to apply sediment to the forest floor. Alternatively, sediment nourishments to the foreshore to create a kind of mud motor might be worth investigating. A mud motor is an artificial mud bank from which mud is gradually dispersed landward by the waves and tide (Baptist *et al.*, 2019). However, it has only been applied in the Wadden Sea. Research with this method in other systems yet needs to be done, but it might be worth studying as it could be a temporary solution to where mangrove forests are drowning, but need to be maintained until the underlying problem, in this case subsidence, can be resolved.

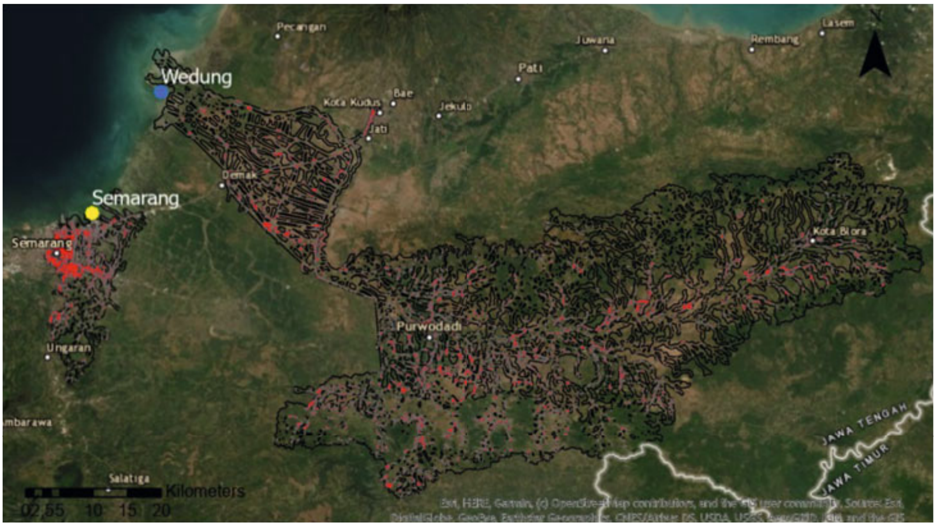

__Figure 7-10:__ Map with catchment area of rivers (black lines) that flow into the Java sea within 5 km from each study site. The two most distal sites within the larger study area are displayed (Semarang & Wedung). The area of built environment (red) within 200 m from each river positively correlated with the number of plastic items found at each site.

7.2 Landward edge dynamics

Like the previous sections, mangrove restoration is often focused on the seaward edge. This is logical, as the seaward edge is dynamic, and restoration efforts are then used to overcome periods of erosion. In addition, they are less complicated than the landward edge in terms of land tenure (Primavera and Esteban, 2008). However, Chapter 6 and the synthesis in the previous sections elucidated that the seaward edge of mangrove forests will very likely retreat under continued sea level rise. If mangrove buffers are desired between ocean and land, it is important to understand the factors that drive landward mangrove edge dynamics as well, so that possibilities for landward mangrove escape with rising water levels may be optimized.

7.2.1 Natural processes that drive landward edge dynamics in pristine systems

Pristine mangrove ecosystems often display zonation of species (Watson, 1928), the composition of which is shaped by abiotic factors such as gradients in salinity (Clarke, Kerrigan and Westphal, 2001), inundation time and microtopography (Van Loon *et al.*, 2016), as well as by biotic factors such as species traits in buoyancy and shape of propagules (Van der Stocken *et al.*, 2015). The location of the landward mangrove forest edge is often determined by competition between mangrove and terrestrial plant species (Smith, 2013). Mangroves and salt marsh plants are facultative halophytes (Wang *et al.*, 2011), which means that they can grow under fresh water conditions, but they are usually outcompeted by terrestrial species at supratidal sites (Smith and Goetz, 2021). Terrestrial vegetation cannot encroach into the upper intertidal zone due to intolerance to the saline environment and is therefore abiotically bound to the supratidal zone (Smith and Goetz, 2021). This intolerance becomes visible when storms and rising water levels cause a shift in the salt limit (Jiang *et al.*, 2014), which results in ghost forests of terrestrial trees at the boundary of the salt and fresh water line (Schieder and Kirwan, 2019). At such sites, salt tolerant vegetation such as salt marsh plants or mangroves then take over and can migrate inland (Smith, 2013). In arid regions, the inland extent of mangroves is determined by hyper salinity (Hutchings and Saenger, 1987). At locations high in the intertidal, evaporation rates far exceed flushing by precipitation, freshwater discharge or even tidal flushing, which results in the formation of brines and salt pans, which are devoid of any vegetation (Terchunian *et al.*, 1986; Lovelock *et al.*, 2017). Lastly, the landward mangrove edge is also strongly determined by slope and topography. The steeper the slope, the narrower the zone between MSL and high-high water (Watson, 1928), and thus the smaller the potential mangrove habitat (Pontee, 2013).

Sites in the upper intertidal zones may have advantages for mangroves; upland grasses retain propagules after storm surges (Peterson and Bell, 2012) and may thus facilitate landward mangrove migration. In the cases of freshly deposited sediment in upland zones (e.g. due to a hurricane), these sites may also be beneficial for mangrove recruitment, as evaporation causes sediment in the high intertidal to drain and compact more quickly, which makes it more stable than saturated sediment (Fagherazzi *et al.*, 2017), and therefore more favorable for seedling establishment (Chapter 2). Finally, inland sites are far less exposed than the seaward edge, further diminishing the chances of seedling dislodgement. The relative suitability of sites along the landward mangrove edge might explain why the main propagule peak of all pioneer species in Demak started right before the wet-season (*Figure 4-5*), the most unfavorable season in terms of wave and wind forcing. Water levels are higher during this season due to increased water discharge from rivers and creeks. The high water levels could actually be beneficial to the seedlings as this allows them to be carried further inland and / or a higher intertidal zonation, safe from waves and erosion, and towards more stable sediments. The co-occurrence of mangrove fruiting and wet-seasons appears to be a global trend; mangrove trees situated in tropical zone, but above 10° latitude, all drop their propagules in the wet season (Van der Stocken, López-Portillo and Koedam, 2017).

7.2.2 Anthropogenic influences along the landward edge

Along many shorelines, natural processes no longer determine how far inland mangroves can propagate. In fact, the mangrove species that are naturally found in the highest intertidal zones are among the most threatened (Polidoro *et al.*, 2010), either because the landward edge of the forest is converted to other land uses (e.g. Béland et al., 2006; Ilman et al., 2016; Thomas et al., 2017), or because the freshwater supply is disrupted by (climate change induced) droughts (Mafi-Gholami, Mahmoudi and Zenner, 2017) or upstream canalization (Donders *et al.*, 2008). Mangrove restoration interventions along the landward edge should therefore be aimed at addressing these issues.

Negative anthropogenic influences on the landward edge.

In urban areas, hard coastal infrastructure often constricts mangroves from expanding landward (Pontee, 2013; Rogers *et al.*, 2019; van Zelst *et al.*, 2021). In rural areas, reasons for restricted landward expansion vary on regional and time scales. Traditionally, the landward edge in rural areas used to be mainly threatened by direct logging of mangroves for firewood or charcoal production (e.g. Semesi, 1998), but over the past decades conversion to other land-uses has become a more serious threat to mangroves. In South-East Asia, the main threat has been the conversion of mangroves to aquaculture ponds, although conversion to other land-uses such as oil palm, and rice agriculture have become increasingly important threats in recent years (Richards and Friess, 2016). In other regions, invasive climbers, grazing cattle (Saenger and Siddiqi, 1993; Biswas *et al.*, 2009; Wodehouse and Rayment, 2019), or conversion to salt pans (Semesi, 1998), can impact the landward edge, and thus greenbelt restriction. Land-use change however remains the biggest threat to mangroves worldwide, which also makes it the biggest opportunity for mangrove rehabilitation (Worthington and Spalding, 2018).

Positive anthropogenic influences on the landward edge

Biophysical measures along the landward edge

In urban areas, where dikes or seawalls line the landward edge of the forest, the landward mangrove edge is defined by infrastructure. However, where space behind a dike allows for mangrove restoration, a mangrove refuge might be designed through creation of double dikes, in which dikes are set-back a certain distance from the coast to create space for landward mangrove retreat following sea level rise, as has been designed for salt marshes (Temmerman *et al.*, 2013). This will have another positive effect, as the tidal prism within the forest (and thus the sediment supply to the forest) will increase (Winterwerp et al. 2013), making the forest more resilient to sea level rise.

Along rural coasts without conventional flood defence along the landward mangrove edge, mangrove restoration traditionally mainly involved planting. Chapter 4 revealed that planting on pond bunds of active aquaculture ponds can be a successful measure in terms of tree survival and mangrove cover, although it does not result in a functional forest. When ponds have become disused, mangroves may start to expand landward on their own accord when the direct stressor for mangrove existence (i.e. clearcutting) has been removed. Chapter 2 showed that abandoned aquaculture ponds along the landward edge often harboured naturally established mangrove seedlings, although not all areas within a pond were equally suitable for recruitment; Locations with a relatively high elevation and with relatively compact sediment were the most favourable for seedling establishment. Chapter 4 furthermore revealed that propagules were abundant in canals of the aquaculture zone, hundreds of meters from their source along the coast. This information is useful for mangrove restoration in disused ponds. For instance, the excavation of additional drainage canals, or removal of remaining pond dikes may help speed up the infilling process and enhance propagule entrance at high tide and sediment drainage at low tide. Both measures have been shown effective to promote natural seedling recruitment in abandoned aquaculture ponds in other regions as well (Lewis III and Brown, 2014; Balke *et al.*, 2020). Landward mangrove restoration measures for other land-use types are often based on the same principle, and are designed to target hydrological reasons for failure of seedling establishment (Lewis III, 2005; Teutli-Hernández *et al.*, 2020).

Socioeconomic measures along the landward edge

The above-mentioned hydrological measures to restore mangroves along the landward edge, all assume that the land has been abandoned. However, since the largest threats to mangroves along the landward edge in rural areas are related to land-use change (Thomas *et al.*, 2017). This means that in rural areas, positive human interventions that will have the most impact and the most long-lasting effects are those that address socio-economic issues and provide alternative livelihoods to local communities. This is to ensure that the mangroves that are restored are not removed again (Biswas et al. 2009; Damastuti and de Groot 2019; Kusmana 2017). In the building with nature project in Demak, socioeconomic issues were addressed by the implementation of both coastal field schools and the bio-rights approach. These measures were aimed at active engagement of local communities to prevent conversion of mangroves to aquaculture. This first entailed empowering communities through coastal field schools, which informed them about the importance of mangroves and more sustainable forms of aquaculture. Then, the communities designed village plans in which they designated existing non-productive ponds for greenbelt restoration, productive riverine ponds for mixed-mangrove aquaculture, and ponds further inland for implementation of sustainable aquaculture practices. Finally, pond owners that gave up (part of) their ponds for greenbelt restoration were financially compensated through the bio-rights approach. While the effectiveness of the biophysical processes has been studied in depth in Chapter 2 and 4, the effectiveness of the bio-rights interventions in the aquaculture zone was not yet quantified in terms of mangrove cover. I therefore looked into mangrove cover change in (1) the ponds that participated in the coastal field schools and bio-rights programme, (2) the wider aquaculture zone within the village boundaries (landward of the shoreline), and (3) the wider study area (administrative village boundaries and coastal zone), all over the period of 3 years preceding (partial) pond designation for mangrove restoration (2015-2018) and the period after pond designation (2018-2021), making use of Sentinel-2 imagery.

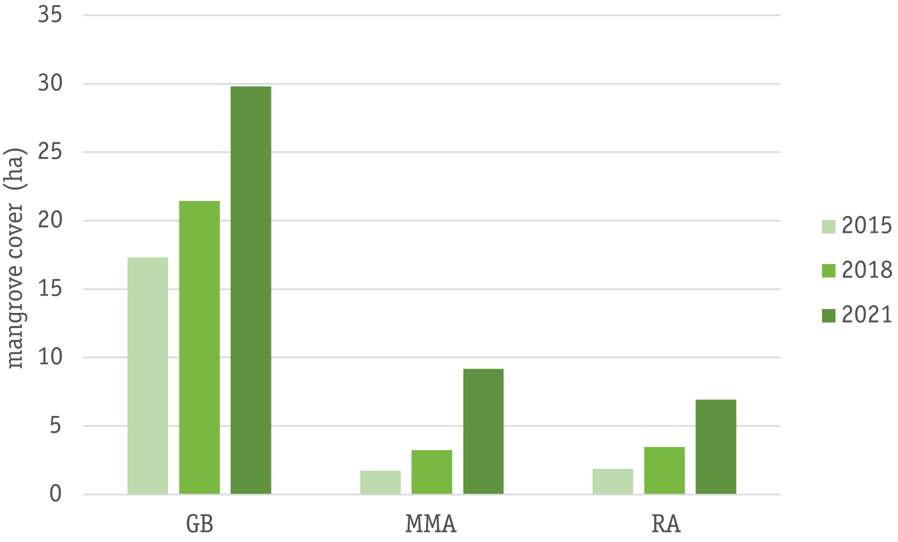

Figure 7-11: *total mangrove cover increase in greenbelt ponds (GB), mixed mangrove aquaculture ponds (MMA: riverine ponds that sacrificed part of the pond area for mangrove restoration along the creek), and sustainable aquaculture ponds (RA).*

The specific aquaculture ponds that participated in the bio-rights programme all showed an increase in vegetation cover over the entire project period, both before and after the official designation of project ponds in 2018 (*Figure 7-11*). However, mangrove cover increased significantly faster throughout all pond types after implementation in 2018 than before (p=0.016, pairwise Wilcoxon rank test).

Although mangrove cover increased over all pond types, the amount of pond area that was assigned for greenbelt restoration contained considerably more mangroves than the other pond categories. By the end of the project, 48.6% of the pond area set aside for greenbelt restoration had successfully recovered. When excluding ponds that were outside the initial target villages or were situated seaward of the shoreline in 2021 (thereby excluding ponds that had been subject to coastal erosion or had been situated seaward of the shoreline already since the start of the project), 72 % of the pond area assigned for greenbelt restoration had been successfully restored (Figure 7-12). This illustrates the importance of community engagement along eroding coastlines to convert ponds back to mangroves, as it is key to establish a mangrove greenbelt when coastal erosion cannot be halted and seaward mangrove restoration in challenging.

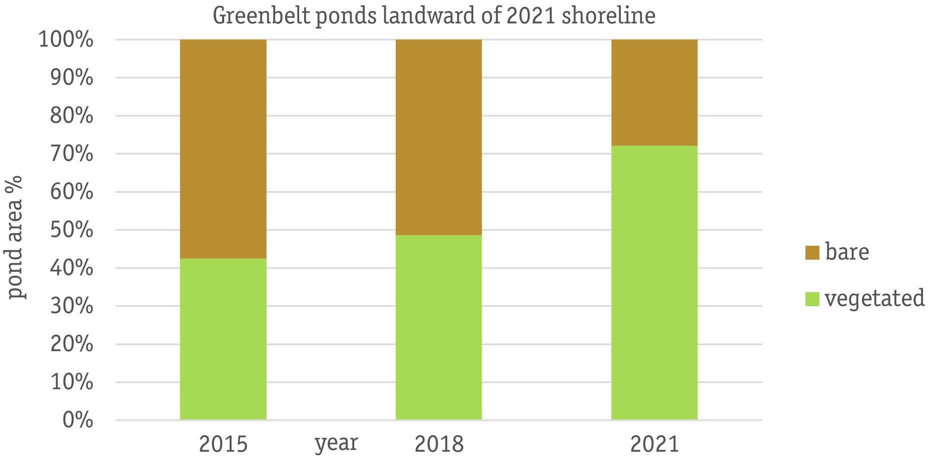

Figure 7-12: *total pond area designated to one of three categories (greenbelt restoration (GB), mixed mangrove aquaculture (MMA), or sustainable aquaculture (RA))*

Outside the aquaculture ponds of owners that participated in the project, the general project area (village boundaries) also showed an increase in mangrove cover, suggesting that substantial awareness and alternative livelihoods were created during the project allowing more space for mangrove recruitment. Over these 6 years of project activity in the region, the mangrove cover in the project area showed a net increase with 26 hectares from 225 ha to 252 ha in the 3 year pre-project implementation period. In the 3 years following project implementation, the mangrove expansion occurred more rapidly, with mangrove cover increasing with another 83 hectares so that the total mangrove cover had become 321 hectares in 2021. These numbers are net mangrove cover changes. In absolute numbers the mangrove area was revealed to be dynamic, as the new mangrove recruitment over the entire six year period was actually 208 hectares, while at the same time a 100 hectares of mangroves were lost between 2015 and 2018 (Figure 7-13).

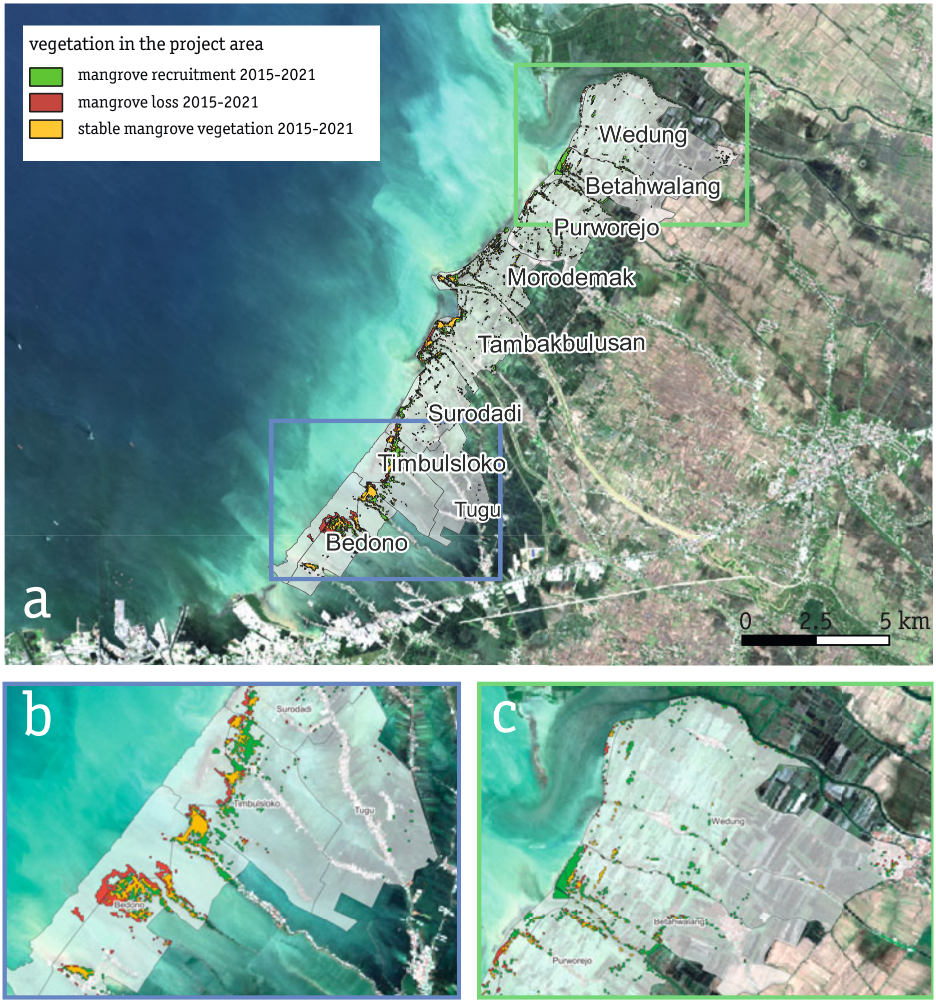

Figure 7-13: *Mangrove change in the project area between 2015 and 2021, note how the villages in the south-west (blue rectangle and panel b) show a landward retreat of mangroves over the years, while the villages in the north-east of the study area (green rectangle and panel c) show a stable shoreline and even seaward expansion in one location.*

This loss was not the result of mangrove to aquaculture conversion, but mostly caused by erosion in the coastal zone. This is illustrated in Figure 7-14, in which the mangrove cover per village has been separated in coastal mangrove cover and landward mangrove cover (pond zone). Overall mangrove cover decreased only in Bedono, which is situated closest to the subsidence epicentre in the area and was subject to substantial coastal erosion. All other villages displayed an increase in mangrove cover over the years, of which the largest amount of mangrove increase was in landward direction (green bars). This mangrove cover increase was more pronounced after project implementation in 2018 than in the years prior to project implementation. This illustrates how intensive community engagement through coastal field schools and benefit sharing schemes, can both limit the continued conversion of mangroves to aquaculture ponds, and create space for landward mangrove expansion, a crucial step towards generating the necessary greenbelt width for coastal protection.

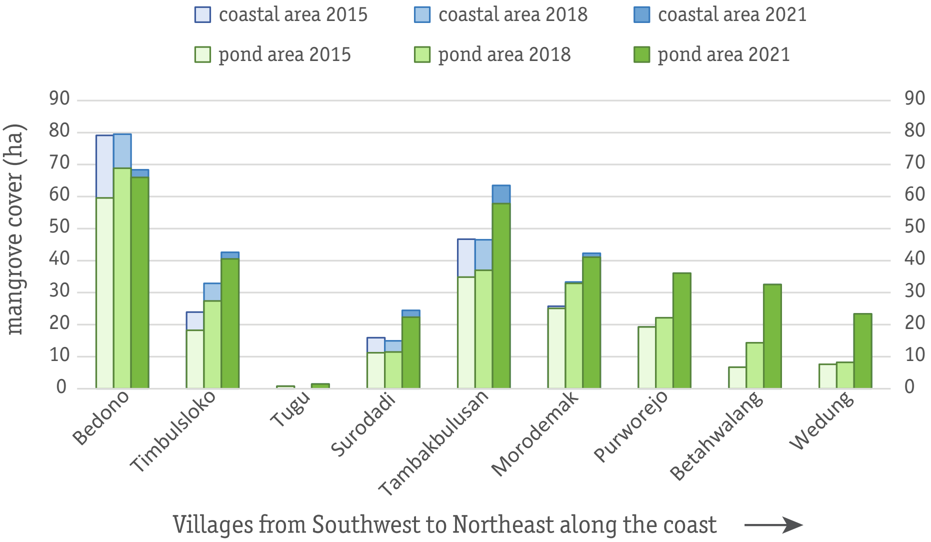

Figure 7-14: *mangrove cover (ha) change over the years for each village. Mangrove cover in the coastal zone seaward of the shoreline in 2021 is indicated in blue. Mangrove cover landward of the 2021 shoreline and in the pond area, and thus sheltered from erosion prior to 2021, is indicated in green. Note how the village in the south-west, and thus close to the subsidence epicentre, is subject to substantial erosion along the seaward edge.*

Conclusions

CHAPTER 8

Conclusions

The research presented in this synthesis and the preceding chapters was aimed at understanding how we can restore mangroves for coastal protection along vulnerable tropical coastlines. Understanding factors that limit natural mangrove expansion in both natural and degraded systems (Question 1) naturally point towards factors that remove those stressors and may thus be targeted in restoration attempts (Question 2). Ultimately, underlying anthropogenic stressors for mangrove growth and survival need to be addressed as well to maintain a mangrove greenbelt once restored (Question 3).

Along the seaward edge, foreshore depth and subsequent erosion are the main constraints of mangrove expansion in both natural and degraded systems; Negative anthropogenic influences such as relative sea level rise and sediment depletion only exacerbate the effect of natural factors by deepening the foreshore and thereby worsening erosion (*Figure 8-1*). Plastic waste can potentially increase the speed of mangrove retreat as it weakens the trees in the fringe before the erosion front hits. The increasing intensity and frequency of storms also aggravate mangrove retreat, but these cannot be addressed with local mangrove restoration measures. Although storms should be considered in the design of a resilient mangrove greenbelt. Biophysical restoration measures along the seaward edge that slow down erosion are likely most effective to counteract mangrove retreat under continued anthropogenic stress. These measures should be designed to mimic natural promotors of mangrove expansion, such as restoration of sheltering mudflats or cheniers (*Figure 8-1*). Measures that elevate the foreshore bed, either through sediment trapping with permeable structures or potentially by sediment nourishments, could therefore stabilize the mangrove fringe. However, it is unlikely that these measures will be able to expand the mangrove fringe in seaward direction under continued rapid relative sea level rise.

Mangrove restoration in landward direction has much greater potential, especially in rural areas where wide mangrove greenbelts are needed to shield villages from the most devastating storm impacts. In natural systems, the landward mangrove edge is determined by topography and freshwater input, limiting mangrove expansion only by hyper salinity in arid regions or by competition with terrestrial vegetation in humid regions. However, along densely populated, and vulnerable tropical coastlines, a mangrove greenbelt will nearly always be constrained by other land-uses on the landward edge and therefore rarely stretches to its full natural potential. Land conversion to agriculture or aquaculture often changes the topography and the water influx to a site (*Figure 8-1*). When aquaculture ponds become disused, natural mangrove recruitment is limited by depth and poor soil consolidation (*Figure 8-1*). Biophysical measures that support mangrove expansion into disused ponds or rice paddies are thus aimed at restoring surface elevation and hydrological connectivity, so that a site's access to sediment and propagules is improved. Mangrove planting is often superfluous when these biophysical measures have been installed and is only necessary when there is no upstream propagule source (*Figure 8-1*), or when species need to be reintroduced. When the landward migration of mangroves is restricted by artificial topography such as traditional flood defence structures, landward mangrove expansion may be supported by the creation of a double diked system, in which the main defence structure is reconstructed further land inward. The outer structure is then breached, and the intermediate area is allowed to fill-in and recolonize. The creation of this green-grey flood defence system may significantly increase the durability of both the grey infrastructure and the greenbelt under continued sea level rise. However, one immeasurably important aspect that is implied with these biophysical measures is that the land along the landward edge has become available for mangrove restoration. This can only be achieved if communities are willing to sacrifice land for restoration. The case of Demak shows that with the help of socioeconomic measures such as benefit sharing schemes and education to promote alternative livelihoods (*Figure 8-1*), mangrove expansion in landward direction can surpass seaward expansion.

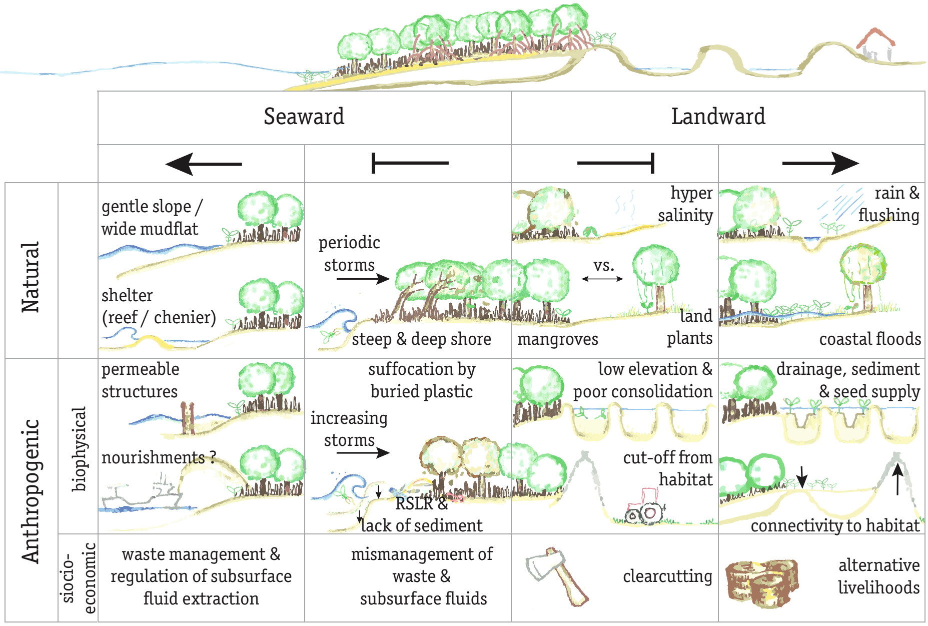

Figure 8-1: *An overview of the natural and anthropogenic factors (rows) that hamper (—I) or promote (→) mangrove expansion in seaward or landward direction (columns).*

Although the findings presented in this dissertation might not necessarily translate one on one to other locations, both aquaculture and relative sea level rise are major threats to mangroves worldwide. The rapidly subsidence aquaculture coast of Demak therefore provides useful insights. The Demak case shows that mangrove restoration in seaward direction is restricted due to rapid relative sea level rise, which limits the possibilities of widening the existing mangrove fringe. Greenbelts typically need to be at least a hundred meters wide to attenuate wind-and swell waves, and even up to a kilometre wide to protect the hinterland from flooding. The case of Demak therefore shows that besides focussing on how we can restore mangroves in seaward direction, we should also be focussing on the landward edge, where kilometres of potential mangrove habitat still exist in the form of other land-uses. With the proper biophysical and socio-economic restoration measures, a sustainable and functional mangrove greenbelt may be restored in this direction even along very challenging coastlines. However, if the underlying issues for mangrove degradation are not addressed, people and mangroves will move further and further inland. Ultimately, interventions with the largest impact and the most long-term effect on coastal protection, will be those interventions that halt regional and global stressors of mangrove establishment and survival: pollution, subsurface fluid extraction, and climate change.

Appendix:
How Can Nature Protect People Against Sea-Level Rise?
a core-concept paper for childeren

CHAPTER

9

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How Can Nature Protect People Against Sea-Level Rise?

9.1 Abstract

Almost one third of people on Earth live near the coast, at risk from floods. Coastal areas are often protected from flooding by human-built ‘gray’ flood protection structures, like dikes and seawalls. Now that our climate is changing, sea-level rise and storms are becoming more intense and more frequent. This increases the risk of flooding. Therefore, we need to develop bigger flood defense structures to stay safe from flooding. However, this is very expensive. Is there an alternative? It may sound surprising, but nature can help us out. Around the world, ecosystems like mangrove forests, salt marshes and coral reefs can help to protect our coasts from flooding. They can contribute to ‘greener’, more natural and biodiverse coasts, and make living along Earth’s coastlines safer and more sustainable. Using these natural systems is called nature-based flood defense. In this paper, we explain how it works.

9.2 Main text

9.2.1 Why do we need to protect ourselves against sea-level change?

The coast is a nice place to live. One third of the people on Earth live at the coast: in small fishing towns, in big cities like New York and Shanghai, or even in entire countries, like the Netherlands. Many of these places lie where a river enters the sea, so called river-deltas. River-deltas are great locations for harbors, agriculture and fishing, but are also low-lying areas that can easily flood. Half of the Netherlands, for example, is several meters below the sea and would be flooded today if some of its flood protection breaches! Living at the coast is risky, and it’s becoming riskier with climate change. Because it has all kinds of consequences, like sea-level change, and more or stronger storms. How does this work exactly?

The continued addition of gases, like carbon dioxide, to our atmosphere causes global temperatures to rise. This leads to sea-level change in two ways. Glaciers and ice sheets, like Antarctica and Greenland, are melting. All that melted ice flows into the oceans. More water in the oceans means higher sea levels. At the same time, the oceans capture some of the heat from global warming. Because oceans get warmer, they expand, and sea levels rise even more. But that is not all. Some cities are literally sinking. One of the main reasons this happens is because we pump up water, gas or oil from underground. The weight of buildings and houses then pushes the emptied soil downward. Because the land is now lower, the water is relatively higher. This effect is called land subsidence. In some cities, land subsidence makes the water level rise more than ten centimeters per year! Land subsidence makes sea-level rise an even bigger threat. The effect of land subsidence and sea-level rise together is called *relative sea-level rise* (Figure 9-1a) (1). *Relative sea-level rise* increases the flood risk to coastal areas.

Beyond contributing to (relative) sea-level rise, climate change also has other consequences. For example, it causes more frequent and intense storms. Stronger storm winds can blow more water to the coast, leading to even higher water levels than with relative sea-level rise alone. Because of this, the waves that reach the coast are also higher. Why? Waves consist of an above water part and an underwater part (Figure 9-1b). Normally, when the underwater part of the wave hits the seafloor or dike, it collapses and the whole wave breaks. You can see this for example at sandy beaches. However, when the water level is too high, waves do not break and stay big. Storms with such high waves can cause floods, damage buildings and sometimes even cost human lives. How can we protect ourselves against them?

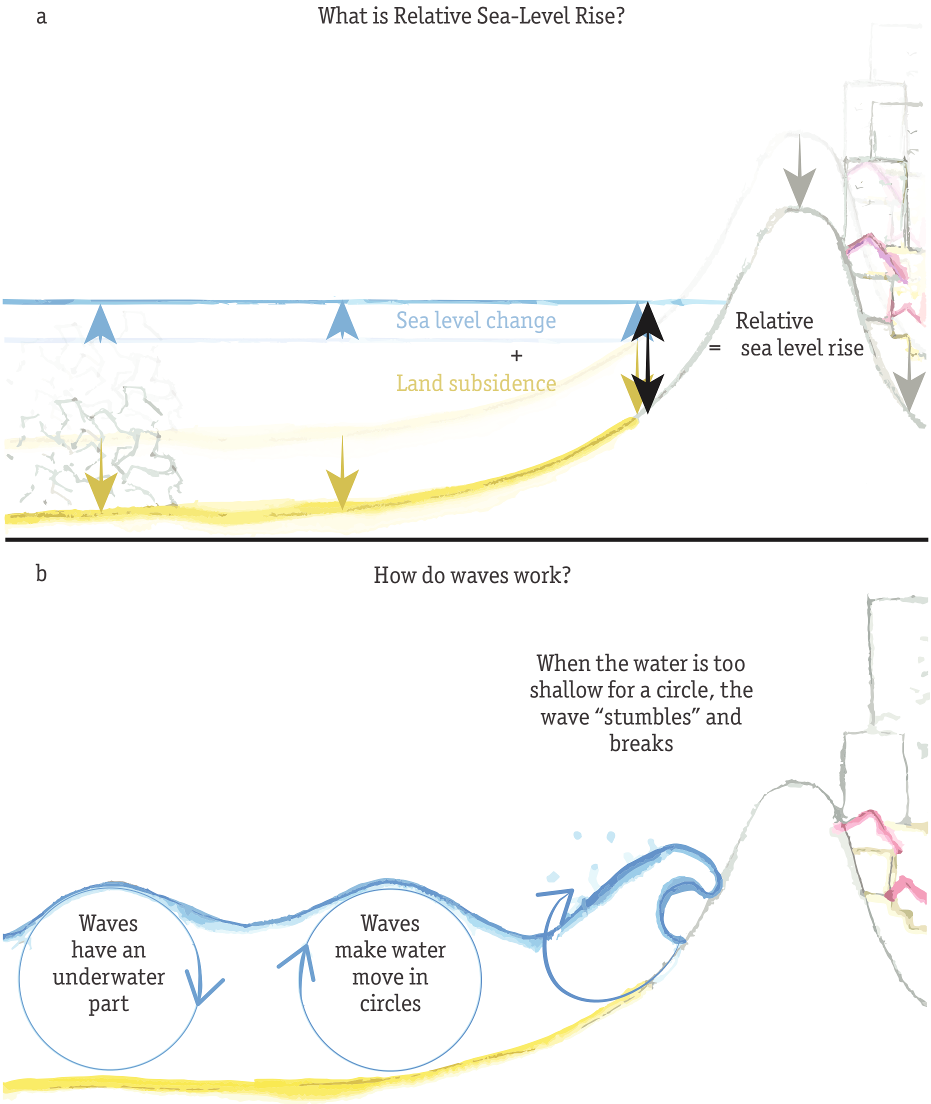

Figure 9-1:* *a. *Relative sea-level rise is a combination of sinking land and rising sea levels.* ***b.*** *Waves have an above and underwater part. When there is no longer enough space for the underwater part, the wave breaks.*

9.2.2 Is it time to update our flood defenses?

Coastal cities are protected by concrete structures often called "gray flood defenses" (Figure 9-2a). These defenses can be dikes, seawalls and breakwaters. A breakwater protects the coast from the force of the waves by forming an elevated barrier in the sea. The underwater part of waves hit this barrier and lose some of their energy. This is kind of like stumbling over a curb when you run. Dikes or sea walls are barriers on the shore. They protect the land from flooding, especially when the water level is higher than normal. This happens during storms in combination with really high tides.

Gray defense structures are built to survive heavy storms, but they are not so effective when sea levels rise. For example, breakwaters are less effective because the underwater parts of waves do not hit the breakwater anymore. Instead, the waves have enough space to travel over the breakwater and break on the shore (Figure 9-2a).

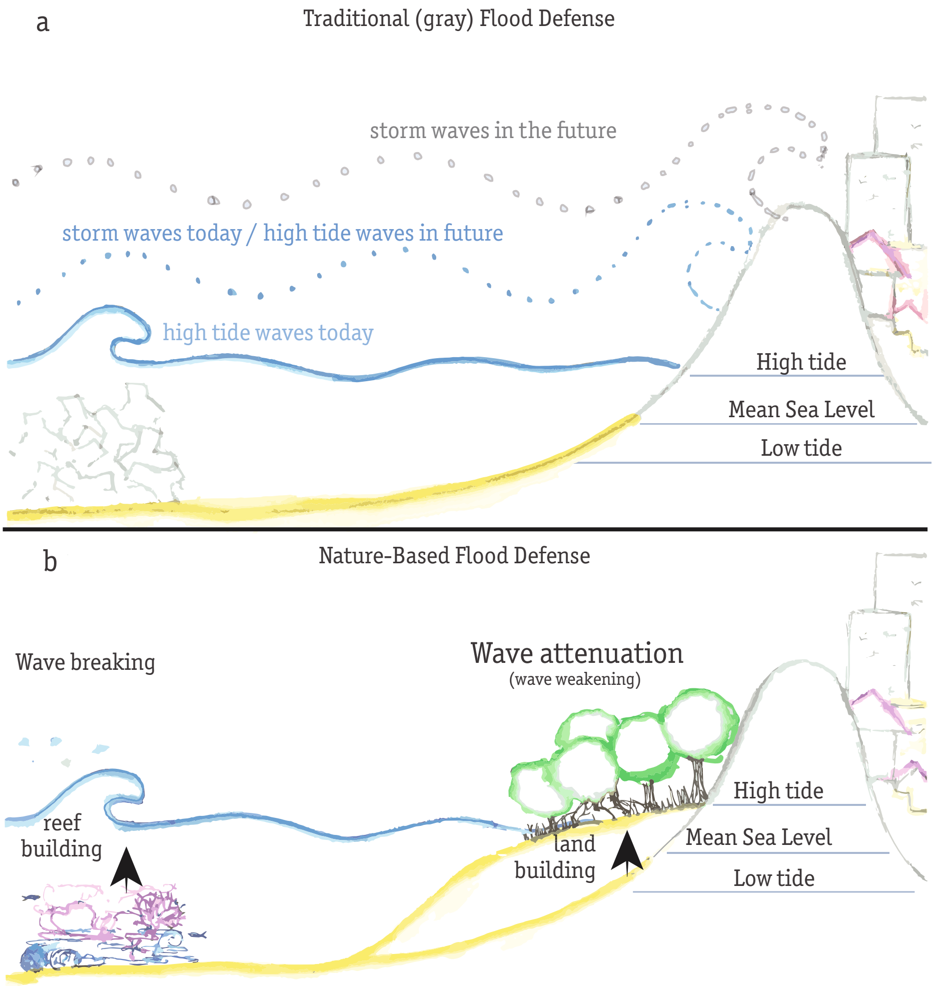

Figure 9-2: a. *Traditional gray flood defense protects the land from waves and flooding, but relative sea-level rise makes gray flood defenses less effective.* ***b.*** *Nature based flood defense uses living ecosystems to help protect the coast. Different plants and animals can be used depending on where you are in the world (see Figure 9-3).*

Dikes and seawalls are also less effective. If relative sea level rises too much, water level becomes too high and land behind it is flooded. Also, dikes and seawalls can be damaged by very strong waves and storms. That also makes them less effective. Sinking lands, rising seas and increasing storms means that we need higher and stronger flood defenses. But that will cost lots of money. What if we could use systems that keep up with relative sea-level rise by themselves? We can! Using living systems to help protect the coastline is called nature-based flood defense (Figure 9-2b) (2).

9.2.3 How nature can protect us from relative sea-level change

On coasts around the world you can find ecosystems that can be used for nature-based flood defense. These ecosystems are made up of plants and animals such as mangrove trees, salt marsh plants, oysters, mussels and corals (Figure 9-3). These plants and animals are adapted to living in salty seawater and can protect us from floods. Some do this by slowing down waves, some do this by building land, and some can do both!

Coral reefs are not just pretty; they are sturdy and grow close to the coast. Their location is very similar to that of the gray, concrete breakwaters. They also work the same way: by forming an underwater elevated barrier that breaks waves (Figure 9-2b). Oysters and mussels can build shellfish reefs that break waves in a similar way to corals. Although reefs and breakwaters protect the coast in the same way, reefs can do one thing more. They grow! This way they can keep up with sea-level rise and stay high enough to keep breaking waves, while gray breakwaters need to be maintained.

Mangroves and salt marshes often grow outside dikes and seawalls. This is helpful, because they can reduce the size of the waves before they reach the dike or seawall. It works like this: if a wave rolls into a forest (or salt marsh), it runs into many obstacles: the mangrove trees or salt marsh plants. Every time it hits a plant the wave weakens and loses a bit of its energy. You can imagine that if a wave hits all of the plants in a marsh or forest, it will lose quite some energy. By losing its energy, the wave becomes smaller. We call this effect wave attenuation. If a mangrove forest or salt marsh is large enough, it can reduce wave height a lot (Figure 9-2b). This is so effective that you can build a smaller dike, saving a lot of money!

Mangroves and salt marshes do not just slow down waves, they can also raise the earth on which they grow. Without plants, seawater can flow so fast that it can carry mud and sand particles. But when water flows through the coastal plants, its speed is reduced and the particles sink down onto the soil. Over time, this makes the soil surface higher (Figure 9-2b). This is very useful for keeping up with sea-level rise (3). And there is another benefit of this process. Because the surface is keeping up with sea level rise, the area is flooded less often than lower lying areas. Mangrove trees and salt marshes like to grow in less flooded areas, so they keep building land there and the surface level keeps increasing. That is very useful, because the elevated surface level can act like a breakwater, protecting the coast even more (4).

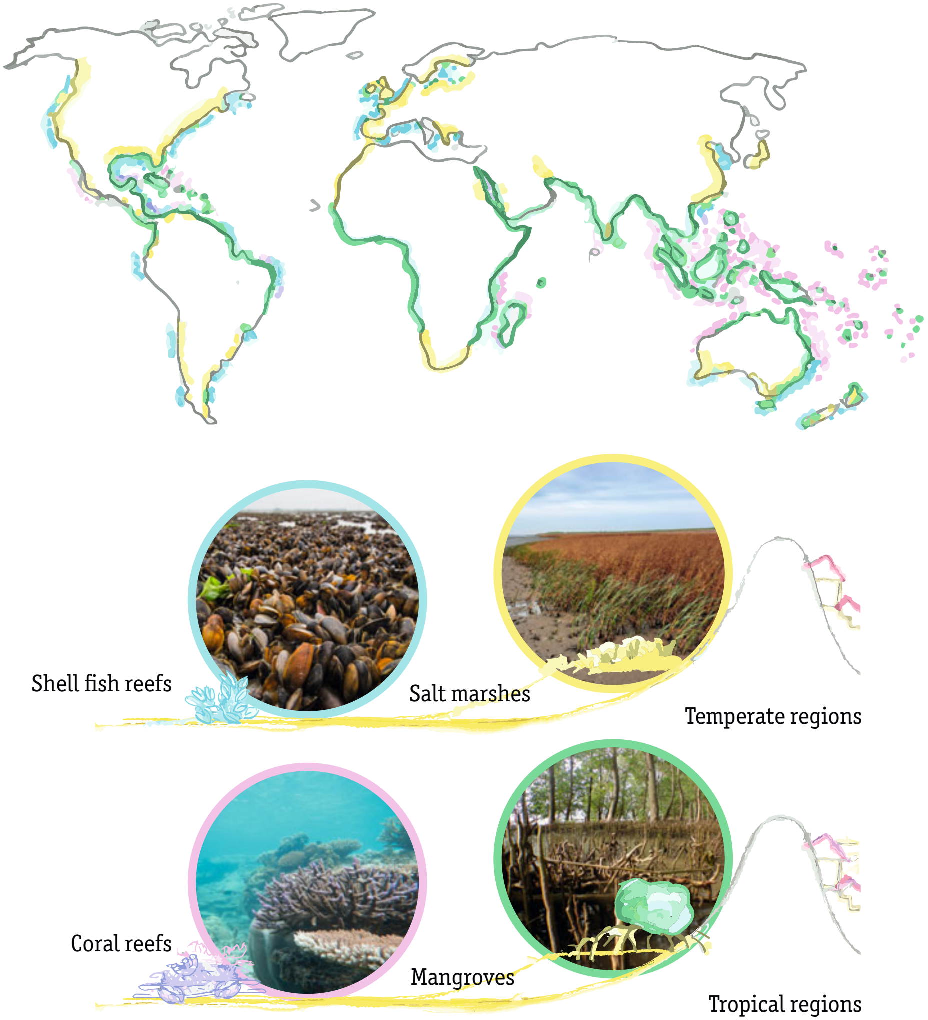

Figure 9-3: *Global distribution of coastal ecosystems for nature-based flood defense. Salt marshes and shellfish reefs are mostly found in temperate regions, while mangroves and coral reefs are found in the tropics. Photos by Jildou Schotanus (mussel reef), Marte Stoorvogel (salt marsh) and Celine van Bijsterveldt (coral reef and mangrove).*

9.2.4 More advantages of using coastal ecosystems

Besides integrating coastal ecosystems in nature-based flood defense, coastal ecosystems have more advantages. As we saw before, their surface level can rise with the sea (5). This makes them cheaper because they need less maintenance than gray flood defenses. These ecosystems can also store large amounts of carbon. This lowers the amount of carbon dioxide in the atmosphere. Lower carbon dioxide levels mean less climate change, so less sea-level rise. Finally, these ecosystems provide a nice area for (rare) plants and animals to live. This supports biodiversity and you can even enjoy nature yourself!

Unfortunately, coastal ecosystems have been declining over the last 50 years. Therefore, scientists are now learning how to protect and restore these coastal ecosystems. We can do this by studying what the plants and animals that live in these ecosystems like. How often do they like to get flooded? How much wave energy can they handle? And how salty should it be? Lots to learn (Box 1)! The more we learn about them, the better we can protect them. So they can protect us!

BOX 1

As authors we like to study nature-based flood defense because we live in a country that is largely below mean sea level. Without flood defense, our houses would be flooded today. For example, Tjeerd would only stay dry if living on the 2nd floor of his house. With the rising sea levels worldwide we are interested to find ways to protect ourselves from flooding by building with nature. Are you also interested in nature-based flood defense? Or would you like to learn more about the ecosystems you can use? Here are some things you can do as a young scientist:

- Give a presentation at your school about nature-based flood defense
- Find and meet your local scientists. Or meet them online (e.g., Skype a Scientist, Letters to a Pre-Scientist).
- Do you want to become a scientist yourself? Learn more about biology, mathematics and physics!
- Read articles on Frontiers for Young Minds about sea level change (https://kids.frontiersin.org/articles/10.3389/frym.2017.00041), global ocean climate change (https://kids.frontiersin.org/articles/10.3389/frym.2021.495240), or seagrasses (https://kids.frontiersin.org/articles/10.3389/frym.2019.00114).

9.3 Glossary

Climate change: Long-term changes in global temperature and average climate conditions. Earth's climate changes naturally, but human activity is causing the world's temperature to rise higher and faster than any time before.

Tides: During high tide the sea-water level is highest and closest to shore. During low tide it is lowest and further away from the shore. Tidal ranges vary around our planet.

Ecosystem: Plants and animals that form a distinct piece of nature. This can be the Amazon rain forest, the Great Barrier Reef, but also your local forest.

Ecosystem restoration: Ecosystems can suffer from damage, just like people can get very ill or injured. Ecosystem restoration tries to repair the damage and make an ecosystem fully functional again.

Summary

Summary

Oceans are rising, our global population is increasing, and nature is under pressure. This is problematic as nearly two-third of the world's population lives within 50 km from the coast. Mangrove forests have wave attenuating and sediment trapping properties. Therefore, mangroves have potential to increase the flood-resilience of low-lying rural coastal areas either by stabilizing eroding shorelines or by complementing conventional flood defence infrastructure, make those more affordable. Mangrove restoration has traditionally been approached from the sea-ward edge, mostly through large scale planting. The success of this approach was often limited. In many of these locations, the reason that mangroves were being restored, is because the coast is subjected to erosion. This makes mangrove restoration in seaward direction challenging. By understanding natural and human induced processes that drive mangrove fringe dynamics along an eroding, subsiding and polluted aquaculture coast, I aim to understand how mangroves can best be restored and maintained for coastal protection along vulnerable coastlines. In this thesis, I study this using the coast of Demak, Indonesia, as example of a heavily impacted coastline suffering both from coastal erosion following mangrove conversion to aquaculture and from rapid land subsidence. The location has however a large sediment supply, which may support mangrove restoration.

The aim of chapter 2 and 3 was to elucidate what natural factors drive mangrove dynamics along the landward and seaward edge of a mangrove greenbelt fringing aquaculture coasts. Chapter 2 revealed that landward mangrove expansion into disused aquaculture ponds is hampered by long inundation times and soft sediment. These factors are associated with pond bathymetry and poor flushing. Seaward mangrove expansion was strongly associated with foreshore morphology: mangroves only expanded in the presence of an elevated mudflat and along gentle slopes. In chapter 3 the effect of different foreshore morphologies was investigated more into depth to obtain targetable foreshore parameters that can be used in coastal zone management to enhance the chances of mangrove persistence and even (re)establishment. By comparing two cross-shore transects, a retreating and an expanding mangrove fringe, we found that cheniers (i.e., elevated sand lenses atop de muddy foreshore) significantly decrease waves that enter the mangrove fringe. A satellite time-series confirmed on a larger scale that the wave-sheltering effect of cheniers in combination with a wide mudflat resulted in a lower chance of mangrove retreat and a higher chance of mangrove expansion. A stable chenier (present for at least two years in a row) was found to promote mangrove expansion, even in combination with a negligible small mudflat of 16 m (95%CI: 0 - 43 m). Without a chenier, a minimum mudflat width of 110 m (95%CI : 76 – 183 m) was required to make mangrove expansion more likely than mangrove

retreat. Although mangrove expansion remains event driven and dependent on a calm reproductive season.

The aim of chapter 4 was to first investigate the effect of ecological restoration measures (EMR) that counteracted stressors of mangrove expansion found in chapter 2 and 3, and then to compare those EMR measures with traditional mangrove planting. Satellite data revealed that over the last decades mangrove planting has resulted in landward mangrove expansion into the active pond zone, although planting was done only on pond dikes and therefore did not result in a functional mangrove forest. EMR measures proved more effective at creating connected forest expansion in landward direction. Propagule trap data from the field revealed that propagules of pioneer species were up to 21 times more abundant in creeks of the pond zone than near their source in the coastal zone, especially during the wet season. This suggests that EMR in landward direction could be achieved through smart sluice gate management in the right season to trap seeds in target ponds, such as associated mangrove aquaculture (AMA) ponds of which a part is sacrificed for riverine mangrove rehabilitation. Remote sensing data that are discussed in the synthesis (chapter 7) indeed revealed that mangrove cover in AMA ponds significantly increased over the project life-time. This suggests that socioeconomic measures aimed at promoting alternative livelihoods, and sustainable mangrove-aquaculture have been successful at creating mangrove habitat along the landward edge. Along the seaward edge, field experiments revealed that EMR with permeable brushwood dams, aimed at expanding mangrove habitat, could not sufficiently overcome subsidence rates to increase natural mangrove expansion in the seaward direction, but did significantly increase the survival of already established (planted) seedlings compared to more wave-exposed sites. The structures thus provided shelter but did not create a sustainable mangrove habitat. This suggests that mangrove restoration in seaward direction will become increasingly difficult under continued relative sea level rise.

In chapter 5 we investigated a potential threat of plastic waste to mangroves once they have been restored along populous and vulnerable shorelines. Southeast Asia is home to both the most extensive (rapidly declining) mangrove forests, and the largest amount of mismanaged plastic waste. Yet, the impact of plastic waste on mangroves is poorly understood. By monitoring plastic on the forest floor and conducting a field experiment, we found that aerial root suffocation by plastic caused immediate pneumatophore growth and leaf loss. This happened both with trees in the field that were subjected to naturally accumulated plastic, and with experimentally treated trees in which 50% of their rootzone was covered in plastic bags. Trees in the 100%-plastic cover treatment had a significantly decreased leaf area index and survival by the end of the experiment. Our findings demonstrate that mangrove trees are relatively resilient to partial burial by plastic waste. However, mangrove stands are likely to deteriorate eventually if plastic continues to accumulate.

Chapter 6 deals with understanding the long-term mangrove greenbelt resilience in the face of continued rapid relative sea level rise. To quantify subsidence rates of the land and foreshore throughout the research area, we collected data on floor raisings in houses and monitored water levels in adjacent mangrove fringes. These two datasets revealed that while the villages closest to the subsidence epicentre sank with 8.2 (95%CI: 6.0-10.4) cm per year, their fringing mangroves sank nearly twice as slow with 4.3 (95%CI: 2.3-6.3) cm per year on average. Nevertheless, this relative sea level rise is multiple times faster than current global sea level rise. However, trees were able to adjust to the rapidly rising water level through pneumatophore extension and the creation of new root mats. A sedimentation experiment subsequently revealed that these adaptations likely enable trees to outgrow subsidence rates of up to 20 cm per year providing that there is sediment available to compensate for subsidence. However, mangrove fringes prooved threatened by deepening of the bare foreshore under high subsidence rates and low sediment supply, as deep foreshores allow larger waves to arrive at the forest fringe. The presence of sufficient sediment to maintain shallow foreshores thus determines the long-term fate of flood-safety from vegetated foreshores. Unfortunately, most subsiding deltas have a negative sediment balance. Therefore, poor rural coastal communities face migration landwards or being trapped in increasingly frequent flooded villages where floors and roofs need frequent raising.

In conclusion, using the coast of Demak, Indonesia, as example of a heavily impacted coastline, this dissertation shows that seaward mangrove restoration is challenging along coastlines that are subjected to rapid relative sea level rise. Under RSLR and relatively-low sediment input, lateral erosion is eminent. However, foreshore management aimed at creating artificial cheniers or wide mudflats (e.g. with the help of sediment nourishments or permeable structures) might reduce lateral erosion. In less rapidly subsiding areas than Demak (i.e., subsidence rate of < 8-20 cm / year) these measures may even lead to a limited extend of seaward mangrove expansion. However, in all cases, it is also important to consider the landward edge of a desired greenbelt when attempting mangrove restoration, as this is where in many cases hectares of potential mangrove habitat still exist in the form of other land-uses. From an ecological and biogeomorphic perspective, restoration will be much easier there, but it does require effective socio-economic measures in addition to biophysical ones, to support mangrove expansion; This may be challenge in its own right, although as shown in this dissertation, not impossible. Ultimately, interventions with the largest impact and the most long-term effect on coastal protection, will be interventions that halt regional and global stressors of mangrove establishment and survival: land conversion, pollution, subsurface fluid extraction, and climate change.

Samenvatting (Dutch summary)

Oceanen stijgen, onze wereldbevolking neemt toe en de natuur staat onder druk. Dit is problematisch aangezien bijna tweederde van de wereldbevolking binnen een straal van 50 km van de kust woont. Mangrovebossen hebben de eigenschap om golven af te zwakken en sediment in te vangen. Mangroven zouden daarom de overstromingsbestendigheid van laaggelegen landelijke kustgebieden kunnen vergroten, hetzij door de eroderende kusten te stabiliseren of door de conventionele infrastructuur voor waterkeringen te versterken, waardoor deze betaalbaarder worden. Mangroveherstel wordt van oudsher benaderd vanaf de zeewaartse rand, meestal door grootschalige aanplant. Het succes van deze aanpak was vaak beperkt. Op veel van deze locaties worden mangroven hersteld omdat de kust onderhevig is aan erosie. Dit bemoeilijkt mangroveherstel in zeewaartse richting. Door natuurlijke en door de mens veroorzaakte processen te begrijpen die de dynamiek van mangroven langs een eroderende, verzakkende en vervuilde aquacultuurkust stimuleren, probeer ik inzicht te krijgen in hoe mangroven het beste kunnen worden hersteld en onderhouden voor kustbescherming langs kwetsbare kustlijnen. In dit proefschrift bestudeer ik dit aan de hand van de kust van Demak, Indonesië. Deze kust is een voorbeeld van een zwaar getroffen kustlijn die zowel lijdt aan kusterosie na mangroveconversie naar aquacultuur als aan snelle bodemdaling. De locatie heeft echter een grote sedimentaanvoer, wat het herstel van de mangrove zou kunnen ondersteunen.

Het doel van hoofdstuk 2 en 3 was om inzicht te krijgen in welke natuurlijke factoren de mangrovedynamiek aandrijven langs de landwaartse en zeewaartse rand van een mangrove gordel langs aquacultuurkusten. Hoofdstuk 2 laat zien dat landwaartse uitbreiding van mangroves in ongebruikte aquacultuurvijvers wordt belemmerd door lange overstromingstijden en zacht sediment. Deze factoren houden verband met de diepte van de vijver en een slechte drainage. Zeewaartse uitbreiding van mangroven was sterk geassocieerd met vooroevermorfologie: mangroven breidden zich alleen uit in de aanwezigheid van een verhoogd slik en langs glooiende hellingen. In hoofdstuk 3 is het effect van verschillende vooroevermorfologieën verder onderzocht om vooroeverparameters te verkrijgen die kunnen worden gebruikt in kustzonebeheer om de kans op mangrove weerbaarheid en zelfs (her)vestiging te vergroten. Door twee transecten haaks op de kust te vergelijken tussen een zich terugtrekkende en een zich uitbreidende mangroverand, ontdekten we dat cheniers (d.w.z. verhoogde zandlenzen bovenop de modderige vooroever) de golven die de mangroverand binnendringen aanzienlijk verminderen. Een satelliet-tijdreeks bevestigde op grotere schaal dat het golfbeschermende effect van cheniers in combinatie met een breed slik resulteerde in een lagere kans op mangrove mortaliteit en een grotere kans op mangrove vestiging.

Een stabiele chenier (minimaal twee jaar op rij aanwezig) bleek de mangrove-uitbreiding te bevorderen, zelfs in combinatie met een verwaarloosbaar klein slik van 16 m breed (95%CI: 0 - 43 m). Zonder een chenier was een minimale slikbreedte van 110 m (95% CI: 76 – 183 m) vereist om mangrove-uitbreiding waarschijnlijker te maken dan mangrove-terugtrekking. Hoewel de uitbreiding van de mangrove door gebeurtenissen wordt aangedreven en afhankelijk blijft van een rustig voortplantingsseizoen.

Het doel van hoofdstuk 4 was om eerst het effect te onderzoeken van ecologische herstelmaatregelen (EMR) die de stressoren van mangrove-uitbreiding, gevonden in hoofdstuk 2 en 3, tegengingen, en vervolgens om die EMR-maatregelen te vergelijken met traditionele mangroveaanplant. Satellietgegevens lieten zien dat mangroveaanplant de afgelopen decennia heeft geleid tot landwaartse mangroveuitbreiding naar het gebied waar actieve aquacultuur wordt bedreven, hoewel de aanplant alleen op vijverdijken heeft plaatsgevonden en daarom niet heeft geleid tot een functioneel mangrovebos. EMR-maatregelen bleken effectiever in het creëren van functionele bosuitbreiding in landwaartse richting. Gegevens verzameld met een propagule-val uit het veld onthulden dat propagules van pioniersoorten tot 21 keer meer voorkomen in kreken van de aquacultuur zone dan in de buurt van hun bron in de kustzone, vooral tijdens het natte seizoen. Dit suggereert dat EMR in landwaartse richting kan worden bereikt door slim visvijver-sluisbeheer in het juiste seizoen om zaden te vangen in doelvijvers, zoals gemixte mangrove-aquacultuurvijvers (AMA) waarvan een deel wordt opgeofferd voor het herstel van mangrovegordels langs kreken. Remote sensing-gegevens die in de synthese (hoofdstuk 7) worden besproken, lieten inderdaad zien dat de mangrovebedekking in AMA-vijvers aanzienlijk toenam gedurende de loop van het project. Dit suggereert dat sociaaleconomische maatregelen gericht op het bevorderen van alternatieve middelen van bestaan en duurzame mangrove-aquacultuur succesvol zijn geweest bij het creëren van mangrovehabitat langs de landwaartse rand. Langs de zeewaartse rand bleek uit veldexperimenten dat EMR met doorlatende kreupelhoutdammen, gericht op het uitbreiden van mangrovehabitat, de bodemdaling niet voldoende kon overwinnen om de natuurlijke mangrove-expansie in zeewaartse richting te bevorderen. Echter, de overleving van reeds gevestigde (geplante) zaailingen werd wel aanzienlijk verhoogd achter de kreupelhout dammen ten opzicht van aan golven blootgestelde locaties. De structuren boden dus beschutting, maar creëerden geen duurzame mangrovehabitat. Dit suggereert dat mangroveherstel in zeewaartse richting steeds moeilijker zal worden bij relatieve zeespiegelstijging.

In hoofdstuk 5 onderzochten we een potentiële bedreiging van plastic afval voor mangroven als ze eenmaal zijn hersteld langs dichtbevolkte en kwetsbare kusten. Zuidoost-Azië herbergt zowel de meest uitgestrekte (snel afnemende) mangrovebossen als de grootste hoeveelheid slecht beheerd plastic afval. Toch is over de impact van plastic afval op mangroven nog weinig bekend. Door plastic op de bosbodem te monitoren en een veldexperiment uit te voeren, ontdekten we dat luchtwortelverstikking door plastic onmiddellijke pneumatofoorgroei en bladverlies veroorzaakte. Dit gebeurde zowel met bomen in het veld die onderhevig waren aan natuurlijk opgehoopt plastic, als met experimenteel behandelde bomen waarbij 50% van hun wortelzone bedekt was met plastic zakken. Bomen in de behandeling met 100%-plastic bedekking hadden aan het einde van het experiment een significant verminderde bladoppervlak-index en overleving. Onze bevindingen tonen aan dat mangrovebomen relatief veerkrachtig zijn tegen gedeeltelijke begraving door plastic afval. Echter, mangroven zullen waarschijnlijk uiteindelijk verslechteren als plastic zich blijft ophopen.

Hoofdstuk 6 gaat over het begrijpen van de veerkracht van de mangrovegordels op de lange termijn in het licht van de aanhoudende snelle relatieve zeespiegelstijging. Om de mate van bodemdaling van het land en de vooroevers in het onderzoeksgebied te kwantificeren, hebben we gegevens verzameld over vloerophogingen in huizen en de waterstanden in aangrenzende mangroveranden gemonitord. Deze twee datasets lieten zien dat terwijl de dorpen die het dichtst bij het verzakkingsepicentrum lagen met 8,2 (95%CI: 6,0-10,4) cm per jaar zonken, hun mangrovengordel bijna twee keer zo langzaam zonk met gemiddeld 4,3 (95%CI: 2,3-6,3) cm per jaar. Toch is deze relatieve zeespiegelstijging vele malen sneller dan de huidige mondiale zeespiegelstijging. Bomen konden zich echter aanpassen aan het snel stijgende waterpeil door pneumatofoorverlenging en het aanleggen van nieuwe wortelmatten. Een sedimentatie-experiment bracht vervolgens aan het licht dat bomen door deze aanpassingen waarschijnlijk boven een bodemdaling van 20 cm per jaar uit kunnen groeien, mits er sediment beschikbaar is om de bodemdaling te compenseren. Mangroveranden bleken echter bedreigd door verdieping van de kale vooroevers onder hoge bodemdaling en lage sedimentaanvoer, aangezien diepe vooroevers grotere golven toelaten aan de bosrand. De aanwezigheid van voldoende sediment om ondiepe vooroevers in stand te houden bepaalt dus het lot van de de mangrovengordel en dus de mate van bescherming die het bos kan bieden tegen stormschade aan het achterland. Helaas hebben de meeste verzakkende delta’s een negatieve sedimentbalans. Daarom worden arme landelijke kustgemeenschappen geconfronteerd met migratie of vastzitten in steeds vaker overstroomde dorpen waar vloeren en daken regelmatig moeten worden verhoogd.

Concluderend, met behulp van de kust van Demak, Indonesië, als voorbeeld van een zwaar getroffen kustlijn, laat dit proefschrift zien dat zeewaartse mangroveherstel een uitdaging is langs kustlijnen die onderhevig zijn aan snelle relatieve zeespiegelstijging. Onder RSLR en relatief lage sedimentinvoer is laterale erosie onvermijdelijk.

Vooroeverbeheer gericht op het creëren van kunstmatige cheniers of brede slikplaten (bijvoorbeeld met behulp van sedimentsuppleties of kreupelhout dammen) kan laterale erosie echter verminderen. In minder snel dalende gebieden dan Demak (d.w.z. een bodemdaling van < 8-20 cm/jr) kunnen deze maatregelen zelfs leiden tot een beperkte uitbreiding van mangroven in zeewaardse richting. In alle gevallen is het echter ook van belang om de landwaartse rand van een gewenste mangrovengordel in ogenschouw te nemen bij pogingen tot mangroveherstel, omdat hier in veel gevallen nog hectaren aan potentiële mangrovehabitat beschikbaar zijn in de vorm van ander landgebruik. Vanuit ecologisch en biogeomorfisch perspectief zal herstel daar veel gemakkelijker zijn, maar het vereist naast biofysische ook effectieve sociaal-economische maatregelen om de uitbreiding van de mangrove te ondersteunen; Dit kan een uitdaging op zich zijn, hoewel, zoals in dit proefschrift wordt aangetoond, het niet onmogelijk is. Uiteindelijk zullen interventies met de grootste impact en het meest langetermijneffect op kustbescherming interventies zijn die regionale en wereldwijde stressoren van de vestiging en overleving van mangroves een halt toeroepen: landconversie, vervuiling, ondergrondse vloeistofextractie en klimaatverandering.

Ringkasan (Indonesian summary)

Kenaikan muka air laut, meningkatnya populasi manusia, dan lingkungan saat ini berada di bawah tekanan. Permasalahan tersebut sering dialami hampir 2/3 populasi dunia yang hidup di wilayah pesisir yang berjarak kurang lebih 50 km dari garis pantai. Hutan mangrove dipercaya memiliki kemampuan untuk meredam gelombang dan menangkap sedimen. Oleh karena itu mangrove berpotensi untuk meningkatkan resiliensi terhadap banjir di kawasan pemukiman pesisir baik dengan menstabilkan erosi di garis pantai atau dengan menguatkan infrastruktur penahan gelombang konvensional sehingga biayanya akan lebih terjangkau. Restorasi mangrove semenjak dulu telah dilakukan secara tradisional dari batas darat kearah laut, mayoritas menggunakan cara penanaman berskala besar. Seringkali, tingkat keberhasilan dengan menggunakan pendekatan ini masih rendah. Di banyak lokasi, alasan utama dilakukannya restorasi adalah karena daerah tersebut mengalami abrasi. Hal ini menyebabkan restorasi mangrove di lokasi yang langsung menghadap laut menjadi suatu tantangan tersendiri. Dengan memahami proses-proses baik yang alami maupun akibat aktivitas manusia yang menyebabkan dinamisasi vegetasi mangrove sepanjang pesisir yang abrasi, permukaan tanahnya turun dan kawasan budidayanya tercemar, penelitian ini bertujuan untuk memahami bagaimana ekosistem mangrove dapat direstorasi dan dijaga dengan benar untuk dapat berfungsi sebagai pelindung daerah pesisir sepanjang garis pantai yang rentan. Dalam disertasi ini, pantai di wilayah Demak digunakan sebagai lokasi penelitian karena merupakan daerah dengan garis pantai yang yang sangat terdampak baik karena erosi pesisir setelah mangrove dikonversi menjadi area budidaya perikanan maupun karena laju penurunan muka tanah yang sangat tinggi. Akan tetapi, lokasi tersebut, memiliki suplai sedimen yang banyak, sehingga cukup memungkinkan untuk mendukung restorasi mangrove.

Tujuan dari Bab 2 dan 3 adalah untuk menjelaskan faktor-faktor alami apa saja yang menyebabkan dinamika mangrove baik kearah batas darat maupun ke batas laut dari suatu kawasan mangrove sepanjang pesisir-pesisir tambak. Bab 2 mengungkap bahwa perluasan mangrove kearah daratan menjadi tambak-tambak yang terbengkalai terhambat oleh waktu penggenangan yang lama dan sedimen yang lunak. Faktor-faktor tersebut terkait dengan kedalaman tambak dan minimnya proses pergantian air.Perluasan mangrove kearah laut sangat dipengaruhi oleh morfologi pantai: Mangrove hanya akan meluas pada area dengan dataran lumpur yang ditinggikan dan sepanjang lereng dengan kemiringan yang kecil. Di Bab 3 efek dari perbedaan morfologi pantai dikaji lebih dalam untuk mendapatkan parameter-parameter pantai yang dapat ditargetkan dalam pengelolaan wilayah pesisir untuk meningkatkan persistensi mangrove and pembentukan Kembali. Dengan membandingkan dua lintasan jalur transek pantai, maju mundurnya vegetasi mangrove, kami menemukan bahwa gosong pasir (sering disebut sebagai cheniers, lapisan pasir yang terangkat menutupi pantai berlumpur) dapat mengurangi gelombang yang masuk ke mangrove secara signifikan. Serial citra satelit dapat mengkonfirmasi pada skala yg lebih besar efek peredaman gelombang oleh gosong pasir dikombinasikan dengan dataran lumpur yang luas memperkecil kemungkinan mundurnya mangrove dan memperbesar kemungkinan perluasan area mangrove. Gosong pasir yang stabil setidaknya berada selama dua tahun berturut-turut) ditemukan mampu mendukung perluasan mangrove, sekalipun hanya dengan kombinasi luasan dataran lumpur yang sempit sepanjang 16 m (95%CI: 0 - 43 m). Tanpa adanya gosong pasir, paling tidak dibutuhkan dataran lumpur sepanjang 110 m (95%CI: 76 – 183 m) untuk dapat memperbesar kemungkinan perluasan mangrove dibandingkan dengan mundurnya mangrove. Namun demikian, perluasan mangrove tetap dapat dinamik dalam masa dan tergantung pada musim reproduksi yang tenang.

Tujuan dari Bab 4 yang pertama adalah untuk menginvestigasi efek dari upaya restorasi ekologi (EMR) terhadap faktor-faktor penghambat perluasan mangrove seperti yang telah dijelaskan di Bab 2 dan Bab 3, dan kemudian membandingkan upaya restorasi ekologi (EMR) tersebut dengan upaya penanaman mangrove secara tradisional. Data satelit mengungkapkan bahwa setelah beberapa dekade terakhir penanaman mangrove menunjukkan ekspansi mangrove kearah daratan menuju kawasan zona tambak produktif, meskipun penanaman hanya dilakukan di pematang-pematang tambak sehingga tidak menghasilkan suatu hutan mangrove yang fungsional. Upaya restorasi ekologi (EMR) terbukti lebih efektif dalam menciptakan perluasan mangrove yang lebih terhubung kearah daratan. Data perangkap propagul di lapangan menunjukkan propagul dari jenis pionir 21 kali lebih banyak di saluran-saluran air di kawasan pertambakan daripada di sekitar sumber benihnya di zona pantai, khususnya di musim-musim penghujan. Hal ini menyarankan bahwa upaya restorasi ekologi (EMR) ke arah daratan dapat dicapai melalui pengaturan pintu air pada musim yang tepat untuk memerangkap benih di tambak-tambak target, seperti halnya tambak budidaya asosiasi mangrove (AMA, associated mangrove aquaculture) dimana sebagian areanya disediakan untuk rehabilitasi mangrove bantaran sungai. Data penginderaan jauh sebagaimana didiskusikan di sintese (Bab 7) memang mengungkapkan bahwa tutupan mangrove di tambak-tambak asosiasi mangrove (AMA, associated mangrove aquaculture) meningkat secara signifikan selama proyek berlangsung. Ini menunjukkan upaya-upaya sosio-ekonomi yang ditujukan untuk mempromosikan mata pencaharian alternative, dan mangrove-budidaya tambak yang lestari sudah cukup berhasil menciptakan habitat mangrove di sepanjang batas daratan. Sepanjang batas laut, percobaan di lapangan menunjukkan bahwa upaya restorasi ekologi (EMR) menggunakan bendungan-bendungan permeable dari kayu dan ranting, bertujuan untuk memperluas habitat mangrove, belum cukup mampu mengatasi laju penurunan tanah untuk meningkatkan perluasan mangrove secara alami ke arah laut, tapi secara nyata meningkatkan ketahanan hidup dari anakan mangrove jika dibandingkan dengan lokasi-lokasi yang terpapar gelombang secara langsung. Struktur yang ada mampu menyediakan perlindungan tapi belum menciptakan habitat mangrove yang lestari. Hal ini membuktikan bahwa restorasi mangrove kearah laut menjadi semakin sulit pada kondisi muka air laut yang relatif meningkat.

Bab 5 menggambarkan investigasi ancaman potensial dari limbah plastik terhadap mangrove hasil rehabilitasi di sepanjang garis pantai yang rentan dan tinggi populasi penduduknya. Asia Tenggara merupakan kawasan dengan mangrove terluas (tapi laju penurunan juga cepat) di dunia, dan juga memiliki jumlah terbesar sampah plastik yang tak terkelola dengan baik. Demikian pula dampak dari sampah plastik ini terhadap ekosistem mangrove belum banyak dikaji. Dengan melakukan pengamatan terhadap sampah plastik di lantai hutan dan melakukan eksperimen lapangan dapat diketahui bahwa penutupan akar nafas mangrove dengan plastik menyebabkan pertumbuhan akar nafas secara mendadak dan pengguguran daun. Hal tersebut terjadi baik pada pohon di lapangan yang tertutup perakarannya secara alami oleh plastik, maupun pada pohon eksperimen yang 50% dari area perakarannya ditutup dengan kantong plastik. Pohon dengan perlakuan penutupan 100% secara signifikan menurunkan indeks permukaan daun (LAI, Leaf Area Index) dan ketahanan hidupnya di akhir masa percobaan. Temuan ini menunjukkan bahwa pohon mangrove cukup resilien terhadap penutupan parsial oleh sampah plastik. Namun demikian tegakan mangrove pada akhirnya akan mengalami kerusakan jika akumulasi sampah plastik terus bertambah.

Bab 6 menceritakan pemahaman tentang resiliensi jangka panjang dari sabuk hijau mangrove yang dihadapkan pada peningkatan muka air laut secara cepat dan terus menerus. Untuk menghitung laju penurunan tanah dan pantai sepanjang area penilitian, kami mengumpulkan data peningkatan lantai rumah penduduk dan tinggi muka air di area mangrove terdekat. Dua data tersebut mengungkapkan bahwa sementara laju penurunan tanah desa di titik terdekat pusat penurunan rata-rata 8.2 (95%CI: 6.0-10.4) cm per tahun, vegetasi mangrove turun lebih lambat sekitar setengahnya yaitu rata-rata 4.3 (95%CI: 2.3-6.3) cm per tahun. Akan tetapi, perlu jadi catatan bahwa nilai penurunan tersebut, diasumsikanjuga sebagai laju kenaikan muka air laut, jauh lebih tinggi dibanding laju kenaikan muka air laut secara global. Meskipun begitu, pohon mampu menyesuaikan laju peningkatan air yang cepat dengan peninggian akar nafas dan pembentukan jaringan akar baru. Eksperimen di bab ini berupa penutupan dengan sedimen telah membuktikan bahwa adaptasi ini memungkinkan tanaman untuk tumbuh melebihi laju pertumbuhan 20 cm per tahun dengan asumsi bahwa ada cukup sedimentasi yang bisa mengkompensasi penurunan muka tanah. Namun, vegetasi mangrove terbukti terancam oleh penurunan area tepi pantai terbuka dikarenakan laju penurunan yang tinggi dan kurangnya suplai sedimen, karena semakin dalam pantai maka akan menyebabkan makin besar gelombang datang menghantam vegetasi mangrove yang ada. Adanya sedimen yang cukup untuk menjaga tepi pantai dangkal pada akhirnya menentukan nasib jangka panjang keamanan dari banjir pada pantai yang bervegetasi. Sayangnya sebagian besar delta yang mengalami penurunan muka tanah memiliki keseimbangan sedimen yang negatif. Dengan demikian, komunitas pedesaan pesisir yang miskin harus bermigrasi kearah daratan, atau terperangkap di desanya yang makin sering kebanjiran dimana lantai dan atap rumahnya harus lebih sering ditinggikan.

Kesimpulannya, dengan menggunakan pesisir Demak, Indonesia, sebagai contoh garis pantai yang sangat terdampak, disertasi ini menunjukkan bahwa restorasi mangrove kearah laut sangat menantang khususnya di sepanjang pantai yang mengalami kenaikan muka air laut relatif cepat. Dibawah kenaikan muka air laut relatif dan input sedimen yang rendah, erosi lateral menjadi sangat menonjol. Namun demikian, pengelolaan pantai yang bertujuan menciptakan gosong pasir artifisial atau dataran rumpur yang luas (misalnya dengan pengkayaan sedimen atau struktur permeable) mungkin bisa mengurangi erosi lateral. Di area yang laju penurunan tanahnya lebih kecil dari Demak (laju penurunan rata-rata < 8-20 cm/tahun) upaya-upaya tersebut mungkin akan bisa membantu perluasan mangrove kearah laut secara terbatas. Meski demikian, di kebanyakan kasus, sangat penting untuk mempertimbangkan sabuk hijau kearah darat dalam restorasi mangrove, hal ini karena sebagian dari lahan yang potential untuk habitat mangrove secara eksistingnya masih dalam bentuk penggunaan yang lain. Dari perspektif ekologi dan biogeomorfi, restorasi akan lebih mudah di sana, tetapi memang membutuhkan langkah-langkah sosial-ekonomi yang efektif selain yang biofisik, untuk mendukung ekspansi mangrove. Ini merupakan tantangan tersendiri, meskipun tampak dari disertasi ini, bukan sesuatu yang tidak mungkin untuk dilakukan. Akhirnya, intervensi dengan dampak yang sangat besar dan efek jangka panjang dalam perlindungan pesisir adalah merupakan intervensi yang menghentikan stresor regional dan global terhadap pertumbuhan dan kelangsungan hidup mangrove yaitu: konversi lahan, polusi, ekstraksi air bawah tanah, dan perubahan iklim.

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Food for thought

The start of the acknowledgement section is the start of a celebration. For me, my PhD was a celebration in itself; For me, it was a dream come true to work on a topic that I am so passionate about and to work with such wonderful people who are all devoted to make this world a better place. I loved every facet of it, the design of experiments, the collaboration across borders and disciplines, the field work, the analysis, the writing. Everything. Unfortunately, every high has its low. I went through a kind of mourning process when my contract ended. I did not want it to end. A difficult year followed. Amidst breastfeeding, flu outbreaks, broken nights, and a new job, I spend all my Saturdays on writing the synthesis of my dissertation and revisions of manuscripts that had come back from review. The last mile truly is the longest.

This is probably the point where I should turn to the upside of it all and point out that all these efforts did result in four published papers, a paper that is under consideration at a high impact journal, and two wonderful children. While these are definitely reasons for celebration and gratitude, the road there was bumpy and could have been easier if I had made different decisions. I think it is important to acknowledge these "bumps" because I hope that this may prevent them from happening to others. My bump in the road was a burn-out, which originated in the love for my work, and my inability to say "no" to opportunities. All my days turned grey and I felt detached from what happened around me. I could no longer enjoy the smiles of my children or the sun in the sky. Luckily, I was able to turn my life around with a year of hard work in therapy, mindfulness, yoga and the support of family, friends and colleagues. However, the many talks with friends, former and new colleagues made it painfully clear that I was not alone in this struggle. Can we not do better?

Burn-out and past-contract struggles to finish a PhD trajectory seem to be common across institutions. While part of the responsibility of finishing a PhD on time lies with the PhD candidate, I think there are important lessons that can be drawn for both PhD candidates and their supervisors to prevent past-contract work on a dissertation and the associated mental health issues. A common denominator in these problems seems to be the joint interest of PhDs and their supervisors to publish papers. This is beneficial for both PhD students and their supervisors, but the timing of when this is attempted, can make a great difference for the wellbeing of the PhD candidate. When a PhD candidate is aiming for a career in science, they are motivated to get papers published, which naturally puts the focus on submitting papers in their PhD trajectory, and less so on the actual dissertation. This does not have to be a problem, but it does increase the risk of having to write the actual dissertation past contract, especially when delays occur in the review or publication process of existing chapters. What if, we would decide to devote the last half year of a PhD contract to finishing the actual dissertation (i.e., writing an introduction and a discussion around the existing chapters). Any chapters that may remain unpublished can be submitted after the defence, but then at least, it is truly the choice of the PhD candidate if they decide to go for it. Dear PhD students, be mindful of your planning and do not fall into the trap of post-contract writing. Dear PhD supervisors, please help your PhD students to finish their dissertation in time. Ultimately, this is likely in the best interest of all parties involved.

Actual Acknowledgements

With this in mind I would like to turn to acknowledging the positive side of PhD life, and to my gratitude for all the wonderful people who have supported me over the years, both scientifically and not unimportant morally. Acknowledgements for specific chapters are given at the end of each chapter, that is why I would like to focus here on the people that have helped me throughout the process of my entire PhD.

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Dear Bregje, we go way back. The BwN project had just started when you took me on as one of the first MSc students on the project to collect data and understand the prerequisites for natural mangrove establishment a little better. We had a great and extremely muddy time in the field, swimming around waste-deep in fluid mud to transplant seedlings and mud between aquaculture ponds to find out if soil properties of bare ponds hampered seedling survival. It was a labor intensive process that proved pointless, when a local fishermen kindly asked us when we would finish our experiment so that he could resume clearing and excavating the pond, the main reason that we had not observed mangrove recruitment at that site. Bregje, thank you for allowing me to do my MSc internship with you and providing me with the opportunity to further expand that great experience in my PhD. You are a great supervisor. You always made time for me, and made me feel like a peer when we discussed experimental set-ups and analyses. I also admire your outspoken opinion about injustices in science, organizations or intercultural matters. You are a great example for me in how I now try to supervise my own students. I am looking forward to working together again on mangroves resilience in the upcoming student projects.

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About the author

Celine van Bijsterveldt was born near Wageningen, the Netherlands in December 1989. She has been fascinated by marine biology since she was eight years old and put on some goggles and a snorkel for the first time. Her love for swimming and her interest in underwater life was fuelled by the enthusiasm of the various biologists and nature enthusiasts at the local diving school. At 16 years old she started teaching scuba diving herself, and after secondary school and a year of travelling she decided to pursue two of her dreams in her studies. In 2009 she started her bachelors at Utrecht University, where she combined medical school with a biology, and even found time to (co)author and edit a book on sustainability with her fellow honours students. After successful completion of both bachelor programmes, she chose to pursue a career in environmental biology. During her masters (2013-2015), she did various marine related research projects, ranging from electrifying cable bacteria in marine sediments and parasite infection in mussels, to effects of mangrove forests on coastal salt intrusion. She was fascinated by mangrove ecosystems due to their applicability in nature-based solutions, which is why she chose another mangrove related topic for her final MSc research project at Deltares: Identifying minimal requirements for mangrove establishment along eroding aquaculture coastlines. This project landed her the PhD position that resulted in this dissertation. After her MSc, Celine worked for one year as a researcher on soil ecology in agriculture (2015-2016), before starting her PhD on mangrove ecology at the Royal Netherlands Institute for Sea Research (2016-2022). In the final year of her PhD she started working part-time for the NGO Wetlands International. During this time, she co-coordinated the writing of a widely endorsed manual for mangrove restoration, which was not yet published at the time of printing of this dissertation. Since December 2022, Celine has worked in her current position as a lecturer Marine Ecology at Wageningen University & Research.

List of publications

First author publications

van Bijsterveldt, C.E.J., *van Wesenbeeck, B.K., van der Wal, D., Afiati, N., Pribadi, R., Brown, B., Bouma, T.J., 2020. How to restore mangroves for greenbelt creation along eroding coasts with abandoned aquaculture ponds. Estuar. Coast. Shelf Sci. 235, 106576. https://doi.org/10.1016/j.ecss.2019.106576*

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