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ARTICLE

Defining Eco-Morphodynamic Requirements for Rehabilitating Eroding Mangrove-Mud Coasts

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Abstract We present an eco-morphodynamic analysis on causes of erosion along degraded mangrove-mud coasts, and the causes of failure to rehabilitate these coasts. Our analyses are based on studies in Thailand, British Guyana and Suriname, and observations in Indonesia, the Philippines and southern China, where degradation is directly attributed to the erection of fish/shrimp ponds too close to the waterline. Ecological determinants of low success rates of rehabilitation efforts are well explained in literature, and are briefly summarized in this paper. Our analyses show that ongoing erosion of degraded mangrove-mud coasts are also the result of a disturbed balance in fine sediment dynamics, and the subsequent change in mudflat morphology. The initial drivers for this degradation are a decrease in on-shore sediment flux and a local increase in wave height near the fish/shrimp ponds. A positive feed-back loop is initiated by which stable, convex-up cross sectional profiles of mud flats evolve towards unstable, concave-up profiles. This loop is induced by an unfavorable feedback between tide-induced sedimentation and wave-induced erosion, deteriorating habitat conditions for mangrove recruitment. Based on the current analysis, we present ingredients for a more sustainable use of mangrove-mud coastal systems, and a more successful rehabilitation of eroding mangrove-mud coasts.

Keywords Mangrove rehabilitation · Fish/shrimp ponds · Coastal erosion · Eco-morphodynamic conditions · Sediment dynamics · Waves

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Introduction

Throughout the tropics, mangrove forests provide tens of millions of people with a reasonable living. For many centuries, they have provided coastal communities with rich fisheries, timber and fuel wood resources (Rönnbäck 1999; Manson et al. 2005; Giesen et al. 2007). Tropical mangroves play a key role in the regulation of atmospheric carbon dioxide levels; containing a carbon stock that averages around 1,000 tonnes per hectare, they rank among the most carbon rich forests in the world (Donato et al. 2011). Mangroves also offer protection to extreme events, e.g. storms, and more gradual processes such as salt water intrusion and shoreline erosion (Gedan et al. 2011; Zhang et al. 2012; Thampanya et al. 2006). These regulating services become increasingly important in the face of projected climate change, specifically with regards to sea level rise. The total economic value represented by mangrove forests is substantial, ranging typically between 2,000 and 9,000 USD per hectare per year (UNEP-WCMC 2006; Spalding et al. 2010).

Despite these important ecosystem services, mangrove systems worldwide are under severe pressure. A global decline has been recorded for mangroves from a total of 18. 8 million hectares of mangrove-covered shorelines in 1980 to 15.2 million hectares in 2005 (FAO 2007). Similar figures are given by Primavera and Esteban (2008) who reported a decline from 450,000 ha of mangrove forest in 1920 to 150,000 ha in 1995 in the Philippines. Though annual deforestation rates have decreased from 1 to 2 % in 1980–2000 to 0. 7 % in 2000–2005, losses remain substantial (FAO 2007).

Mangrove degradation has had a far-reaching impact on the stability of coastlines. While mud-coasts are typically dynamic and naturally subject to progradation and retreat (e.g. Healy et al. 2002), it was found that anthropogenic disturbances of mangroves may inflict major losses of land through erosion (Thampanya et al. 2006). This erosion is caused by disturbance of the fine sediment balance, which occurs once mangroves are converted and tidal flows are confined, following near-shore infrastructure development (Winterwerp et al. 2005). Worldwide, numerous mangrove rehabilitation efforts have been made to halt these losses. However, many of these initiatives met with little success (Ellison 2000; Lewis 2005; Samson and Rollon 2008). In the Philippines for example, 44,000 ha of land was planted with no less then 440 million mangrove propagules at a cost of USD 17.6 million, but survival rates were found to average a meager 10–20 % only (Primavera and Esteban 2008).

Following research in Thailand, Winterwerp et al. (2005) concluded that disturbance of the fine sediment balance is a major, yet overlooked root cause of the poor success of rehabilitation efforts. The changes in sediment dynamics and bathymetry that cause erosion were found to also prevent mangroves to re-establish and resume their coastal protection services. Hence, mangrove degradation may cause accreting coastlines to flip towards an alternate state where net erosion takes place (see also Stanley and Lewis 2009).

This study focuses on the fine sediment dynamics in mangrove-mud areas, and how these dynamics are disturbed by thoughtless land-use, such as the erection of too many fish/shrimp ponds too close to the waterline. This focus is the major difference with earlier studies, such as by Lewis (2005), who emphasized the importance of understanding tidal hydrology to achieve successful and cost-effective mangrove restoration; Winterwerp et al. (2005), who described a pilot study of the erosion processes along a degraded mangrove mud coast in the Gulf of Thailand; and Anthony and Gratiot (2012), who addressed the role of increased wave exposure following destruction of mangrove vegetation in causing mud bank erosion in Guyana.

Building on these previous studies, the aim of this paper is to identify how mangrove rehabilitation efforts along eroding mud-coasts can be made more successful and sustainable. Based on a morphodynamic analysis of the mangrove-mudflat continuum and illustrated through case studies in Thailand, (British) Guyana, Suriname, China, and Indonesia we describe how geomorphological and ecological requirements for rehabilitation can be integrated. Emerging insights could help to reverse human-induced erosion along degraded mud coasts through mangrove-based coastal defense applications. Such applications may prove to be more cost-effective compared to conventional hard infrastructure solutions, while providing a swath of co-benefits derived from the reinstated mangrove services.

Eroding Mangrove-Mud Coasts – Impacts of Mangrove Conversion Illustrated

To illustrate the impact of mangrove conversion on erosion processes along mud coasts, we summarize the results of various multi-disciplinary studies along mangrove-mud coasts in Thailand, (British) Guyana and Suriname and provide anecdotal accounts derived from rapid inventories in Indonesia and southern China.

The erosion processes that are inflicted by mangrove conversion and ill-informed infrastructure development are well visible along Bang Khun Thien coast, just south of Bangkok along the Bay of Bangkok (**Thailand**). The western and northern coastline of this 100×100 km-wide bay have been eroding by 25–30 m per year since the 1960s. Figure 1 is typical for this coastline, showing extensive fish ponds, separated from the bay by small earthen bunds and narrow fringes of mangroves. The coastline is very irregular, owing to local set-backs where bund and mangrove protection have failed.

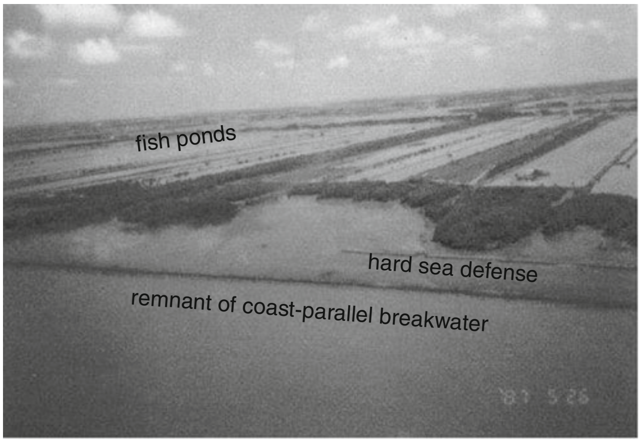

Fig. 1 Coastline of Bang Khun Thien, Gulf of Bangkok, Thailand, showing extensive shrimp ponds protected from the sea through earthen bunds and fringes of mangroves. The aerial photograph also shows remnants of a coast-parallel breakwater and hard sea defence

Then, a new bund is erected several tens of meters further inland, aiming to withstand erosive forces for some time, being encapsulated by surrounding fishponds. Similar erosion patterns can be observed along degraded mangrove-mud coasts elsewhere in the world (e.g. Stanley and Lewis 2009). The coastline in Demak district, mid Java (**Indonesia**) for example (Fig. 2) depicts the remnants of the fish farms erected in former mangrove areas, revealing erosion rates up to 900 m in 10 years. Erosion patterns follow the lay-out of these fish farms. Zooming out with Google Earth reveals similar patterns along the entire northern coast of Java, once renowned for its extensive mangrove forests.

Both in Thailand and Indonesia 'hard' engineering structures, such as coast-parallel breakwaters and seawalls were applied. These structures often failed to protect the coastline from further erosion. In fact, observations suggest that the construction of a breakwater may increase the net erosion rate along a coast; the remains of such a coast-parallel breakwater is shown in Fig. 1. Further details of this study can be found in Winterwerp et al. (2005), who argued that the large erosion rate of Bang Khun Thien's coastline are to be attributed primarily to poor land use planning, exploiting fish ponds too close to the waterline and thereby compromising the protective function of the mangroves. By erecting bunds too close to the waterline, coastline-restoring processes, e.g. sedimentation, were diminished, thereby increasing the eroding forces by wave attack.

The (im)balance between sedimentation and erosion processes is well-illustrated in the alternating accretion and retreat of the mangrove-fringed coastline of **Suriname** (Augustinus 2004; Erftemeijer and Teunissen 2009). Figure 3 shows spatial and temporal variations that synchronize with large, 30 – 50 km long mud banks, migrating along Suriname's coast at a rate of about 1 – 3 km/year with a periodicity of about 30 years (Augustinus 1978; Plaziat and Augustinus 2004). In between mudbanks, the coast erodes, otherwise, accretion occurs, with a general net accretion over longer periods of time (multiple decades). The coastal stretch around Coronie (Fig. 3) however is characterized by long-term erosion since the second half of the 19th century, as further illustrated in more detail in Fig. 4. That erosion is attributed to the development of plantations in former mangrove areas, (too) close to the shoreline by settlers around the 1840s. Authorities have tried to safeguard Coronie's hinterland by the construction of a solid seawall, and less solid, but nevertheless 'hard' engineering constructions in earlier times. Again, inappropriate spatial planning of coastal land use, and subsequent 'hard' engineering coastal protection works seem to have induced and aggravated the erosion problems along this coast. The unfavorable effects of inappropriate land use and 'hard' engineering structures are further illustrated by the dramatic developments along the coast of (British) **Guyana**, notably around its capital Georgetown. Figure 5 illustrates how all original mangrove vegetation along this coast is gone. It has become evident that solid concrete structures do not constitute an appropriate and effective form of coastal protection along such soft muddy coastlines, not in the least because of the poor subsoil conditions and stability problems related to compaction and soil subsidence. In the next sections, we elaborate further on the unfavorable effects of (hard) coastal structures on mangrove-mud coasts, e.g. hard from a hydrodynamic point of view.

Fig. 2 Coastline changes in Demak District (Timbul Sloko village, Central Java, just east of Semarang) between 2003 and 2012 estimated land loss 200 – 900 m (source: *Google Earth*)

Rehabilitation Attempts – Reasons for Failure

Major investments have been made to rehabilitate mangroves across the tropics, amongst others to stabilize eroding coastlines. The large-scale rehabilitation attempts in Bangladesh have received much attention in scientific literature. Here no less than 120,000 ha of mangroves have been planted since 1966, mainly on newly accreted land (Saenger and Siddiqi 1993). Two species of mangrove, *Sonneratia apetala* and *Avicenna officinalis*, dominate the mangrove plantations, usually as monospecific stands. According to Field (1998), the planting of mangroves has been successful in protecting and stabilizing coastal areas and in providing substantial timber production. However, Lewis (2005) states that 'in spite of the success in Bangladesh, most attempts to restore mangroves often fail completely, or fail to achieve the stated goals'. For example, between 1989 and 1995, 9,050 ha of mangroves were planted in West-Bengal, India with only a 1.5 % success rate (observations by Sanyal, Lewis 2005).

In the Philippines, over the past two decades, more than 44,000 ha, mostly non-mangrove mudflats, sandflats and seagrass beds (targeted for restoration because they constituted open access areas with no ownership problems) had been planted with mangroves, using almost exclusively the genus *Rhizophora* (Primavera and Esteban 2008). In these areas, seedlings experienced high levels of mortality and the few that survived (apparently through stubborn, expensive replanting) had displayed dismally stunted growth relative to the corresponding growth performance of individuals

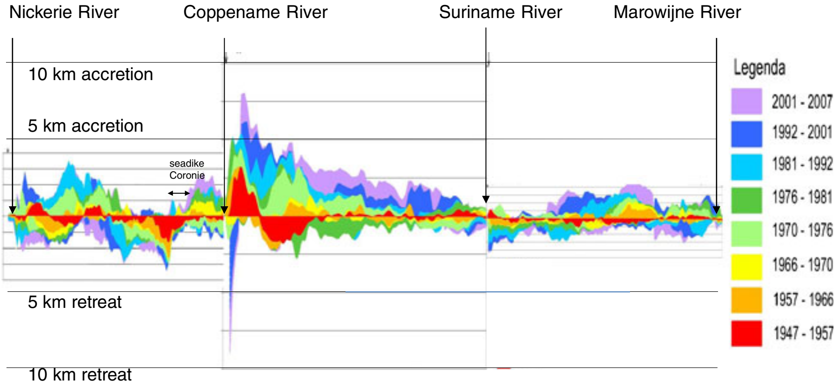

Fig. 3 Alternating retreat and accretion of Suriname's 300 km coastline since 1947, over eight periods of time (redrawn from Augustinus 2004). Note cyclic variations both in space and time. Most of the coast accretes over longer periods of time, except near Paramaribo (located along the Suriname River, and Coronie coast where more than 5 km erosion has been observed since 1900

thriving at the high intertidal position and natural mangrove sites (Samson and Rollon 2008). Despite heavy funds for these massive rehabilitation efforts of mangroves in the Philippines over the last two decades, the long-term survival rates are generally low at 10–20 % (Primavera and Esteban 2008). While the low survival rates are primarily due to wrong site selection, land tenure issues have frustrated efforts to target former fish and shrimp pond areas for restoration, further aggravated by weak law enforcement and a lack of political will (Primavera and Esteban 2008).

In Thailand, a massive 5-year mangrove-replanting program was launched by the Thai Government during 1991–1996, targeting to replant 40,000 ha. According to Suwannodom et al. (1998) this programme cannot be evaluated as successful, except in a few cases in Southern Thailand where a community-based management approach was followed (Erftemeijer and Lewis 2000; Erftemeijer and Bualuang 2002).

Mangroves in (southern) China formerly covered an estimated 50,000 ha in the 1950s, but were greatly reduced to less than 23,000 ha by 2007 (Chen et al. 2009). Large stretches of intertidal area in the southern part of China were considered suitable for rehabilitation of mangrove forest. Many efforts have been made to such rehabilitations, totaling some 2,678 ha of replanted mangroves by the year 2002. However, only 57 % of these rehabilitation efforts were evaluated as successful (Chen et al. 2009). In Fujian and Zhejiang provinces, survival rates of rehabilitated mangrove stands were reported as low as 1.3 % – 31 % (Peng et al. 2008). The unsuccessful rehabilitation of mangroves was mainly attributed to wrong selection of species, unfavorable climate and site conditions such as winter temperature, sediment properties and morpho-hydrological factors, as well as a lack of post-management and monitoring (Peng et al. 2008). Among these, the erosion of coasts has been attributed to be one of the main factors for the unsuccessful rehabilitation of mangroves in the southern part of China. However, a comparative analysis of eroding mangrove-mud coasts and rehabilitation attempts in China has yet to be explored.

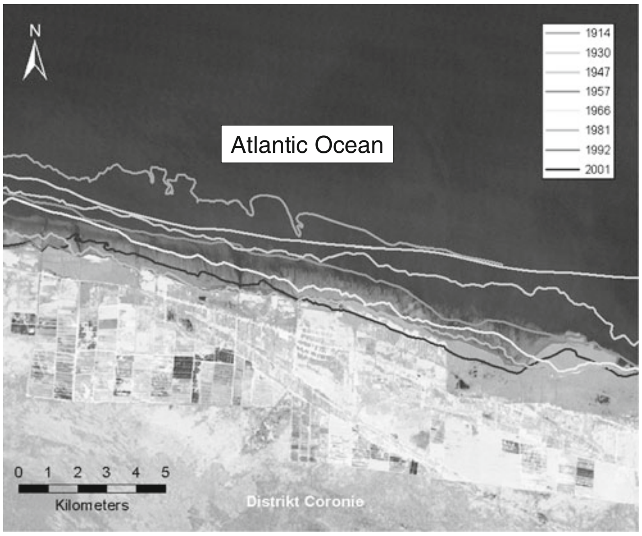

Fig. 4 Eroding coastline at Coronie, Suriname; colored lines indicate the location of the coastline at different years

The above findings underline the importance of carefully studying and evaluating the conditions that determine successes and failures in similar efforts elsewhere in the world. Samson and Rollon (2008) indicated a large number of reasons for failure of mangrove restoration projects in the Philippines, many of which also apply to mangrove restoration initiatives in other countries. Some are of institutional or socio-economic nature (also see Erftemeijer and Bualuang 2002). Others are ecological factors. In most cases, the lack of baseline assessment of the target areas prior to mangrove rehabilitation - often as a consequence of opportunistic site selection and lack of knowledge, funds or equipment - and absence of clearly defined goals and lack of a proper monitoring and evaluation system, have contributed to this failure.

Various authors have emphasized the importance of ensuring appropriate hydrological regimes (depth, duration and frequency, and of tidal flooding) in restoration sites (Kjerfve 1990; Lewis 2005). Failure for these factors to be considered was found to be one of the most important factors that caused low success rates, along with insisting on planting of non-pioneer species that lack specific biological traits related to inundation

Fig. 5 Guyana mud coast around Georgetown. All mangroves have disappeared by 'hard' sea defences, resulting in severe coastal erosion, as shown from the abandoned sluice systems

tolerance and rapid rooting (Balke et al. 2011; Friess et al. 2012). Erftemeijer and Lewis (2000) for example report: '[…] mangrove reforestation on mudflats is not easy, it is often characterized by high mortality rates caused by factors such as barnacle infestation, smothering or burial from excessive sedimentation, wave action and so forth […]. In areas where sedimentation is substantial and mudflats are accreting, such as in the case of Bangladesh and some localized estuaries, success rates are likely to be higher'. On this basis, Erftemeijer and Lewis (2000) and Samson and Rollon (2008) recommended mangrove planting on lower intertidal mudflats to be discouraged or at least reconsidered.

The following five ecological principles, considerations and practical suggestions are based on a well-established process called "Ecological Mangrove Restoration" (Stevenson et al. 1999; Lewis 2005), building on lessons learnt from rehabilitation attempts worldwide (Erftemeijer and Lewis 2000; Lewis 2001; Primavera and Esteban 2008):

1. Understand the ecology of the mangrove species at the site, in particular the patterns of reproduction, propagule distribution, and successful seedling establishment.
2. Understand the hydrologic patterns (in particular the depth, duration and frequency of tidal inundation) that control the distribution and successful establishment and growth of (targeted) mangrove species.
3. Assess modifications of the original mangrove environment that currently prevent natural regeneration (recovery after damage).
4. Restore hydrology and other environmental conditions that encourage natural recruitment of mangrove propagules and successful plant establishment.
5. Only consider actual planting of propagules, collected seedlings, or cultivated seedlings after determining (through steps 1–4) that natural recruitment will not provide the quantity of successfully established seedlings, rate of stabilization, or rate of growth of saplings established as objectives for the restoration project.

This was recently updated with a sixth step, incorporating socio-economic aspects and monitoring requirements (Lewis 2009). While ecological, hydrological (and socio-economic) issues have been adequately covered in the literature, morphodynamic requirements for successful mangrove rehabilitation have received little attention to date. Particularly along eroding coastlines, consideration of these factors is deemed critical to ensure a successful mangrove restoration and halt further losses of land. From the assessment below we derive recommendations for these to be considered in future mangrove replanting schemes.

Morphodynamic Requirements for Succesfull Rehabilitatiion of Mangrove-Mud Coasts

Morphodynamic Processes: Waves, Tides and Sediments

The data and observations discussed in this article illustrate that many mangrove-mud coasts do not form rigid, stable coastlines. Instead, they are often highly dynamic, accreting and retreating cyclically, in particular along coasts facing the open ocean. To understand how mangrove degradation and other anthropogenic disturbances affecing these natural processes, and to appreciate implications for mangrove rehabilitation programs, thorough insight is required into the sediment dynamics across muddy coastlines and intertidal mudflats. The literature contains a wealth of publications on morphodynamic processes along muddy shorelines; the European INTRMUD project for instance, explored the basic characteristics (such as tidal range, wind and wave climate, sediment availability) of these systems (Dyer et al. 2000; Le Hir et al. 2000; Roberts et al. 2000; Anthony et al. 2010). Here we follow Friedrichs and Aubrey (1996) and Lee and Mehta (1997) to explain the interactions between waves, tides and fine sediments. Continuing on the work by Dean (1977), Lee and Mehta (1997) predicted cross-shore profiles to have a parabolic shape, modulated by the effect of wind waves and swell, assuming that the wave dissipation is uniformly distributed across the mudflat. The bed level Z_b then attains a parabolic-exponential profile:

$$\frac{Z_b(x)}{Z_{b,0}} = \left(\frac{x}{X_0}\right)^2 \exp\{-4k_i(x - X_0)\} \tag{1}$$

where $Z_{b,0}$=bed level at offshore location X_0, where the bed starts to feel the waves, and k_i is the imaginary wave number. Lee and Mehta (1997; see also Lee 1995) favorably compared profile (1) with numerous mudflat profiles along the coast of Louisiana and in San Francisco Bay. In the absence of waves, Eq. (1) attains a convex-up profile, whereas this profile becomes progressively more concave-up with increasing wave effects. Other literature sources confirm these observations (e.g. Lee 1995; Friedrichs 2011). Note that Lee and Mehta (1997) added a so-called “near-shore correction” to account for the effects of breaking waves near the waterline, yielding a local increase in steepness of the mudflat profile.

Friedrichs and Aubrey (1996) established an equilibrium cross shore profile of intertidal mudflats under tidal conditions, assuming that the amplitude of the bed shear stress, induced by tidal filling and emptying of the mudflat, is uniformly distributed across the flat. The bed level Z_b then attains a linear-sinusoidal profile in case of a straight coastline (e.g. with respect to plan view):

$$Z_b(x) = \begin{cases} a_0(x/L_r - 1) & \text{for } \mathrm{x} \leq \mathrm{L_r} \\ a_0 \sin\{x/L_r - 1\} & \text{for } \mathrm{x} \geq \mathrm{L_r} \end{cases} \tag{2}$$

where x=coordinate along mudflat, a_0=tidal amplitude and L_r is a reference length of the flat, here set as the length of the intertidal mudflat below mean sea level (MSL). Note that Friedrichs and Aubrey (1996) also give cross-section mudflat profiles for non-straight coastlines, such as in bays.

An arbitrary, but characteristic example of a cross-sectional mudflat profile is shown in Fig. 6, assuming a 1 m tidal range at spring tide (a_0), a 0.8 m tidal range at mean tide, and a mudflat slope of 1:1000 below MSL. The mangrove habitat is found above the mean high water (MHW) level, as sketched in Fig. 6. The variation of the depth-mean flow velocity over the tidal cycle at x=1,000 m, i.e. at mean sea level (MSL) can be obtained from integration of the continuity equation for water, assuming tidal filling only:

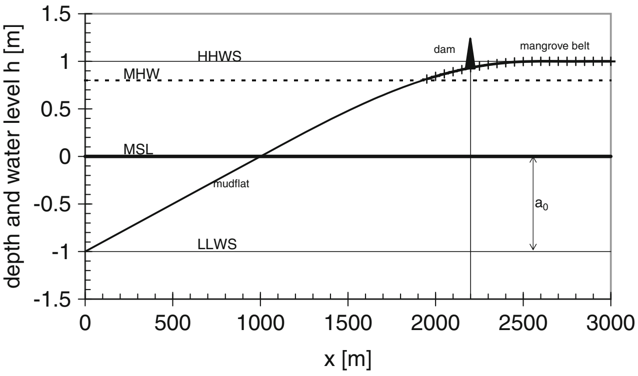

Fig. 6 Cross-sectional mudflat profile (after Friedrichs and Aubrey 1996) with mangrove habitat between mean high water (MHW) and high high water spring (HHWS). Spring tidal amplitude a_0 amounts to 1 m. A dam has been sketched within the mangrove forest at x= 2,200 m, whereas the top of the mudflat is found at x=2,500 m (dam at high-waterline is not drawn)

$$\frac{\partial h}{\partial t} + \frac{\partial hu}{\partial x} = 0, \qquad \text{hence } u(x) = \frac{1}{h}\int_x \frac{\partial h}{\partial t}\,\mathrm{d}x \tag{3}$$

where h=water depth, u=flow velocity, t=time and x=coordinate along the mudflat. The resulting flow velocity at mean sea level (MSL) is presented in Fig. 7, showing flow velocities slightly exceeding 0.1 m/s. Indeed, these are typical velocities over tidal flats.

Integration of this flow velocity over depth and flood period yields the tidal prism P_T per unit width [m^3/m′], i.e. the amount of water flowing towards the coast at location x. This can be done for all locations along the mudflat. The resulting $P_T - x$ graph is given in Fig. 8, showing $P_T \approx$ 560 m^3/m′ at x=1,000 m (MSL) and P_T=0 at the top of the mudflat (x=2,500 m).

Impacts of Mangrove Conversion Explained

So what happens if the processes that determine sediment dynamics and soil profile are disturbed as a result of mangrove conversion and subsequent infrastructure development? Let us analyze the effect of the erection of fish/shrimp ponds at 200 m from the high-waterline (i.e. the edge of mangrove forest) and at the high-waterline itself by studying the effect of dams on the mudflat at these locations, e.g. at x=2,200 and 1,950 m (Fig. 6). The resulting changes in tidal prism are also presented in Fig. 8, showing a reduction by 10 % (dam at 2,200 m) to 20 % (dam at 1,950 m) close to the location of the dam. Further seaward, this reduction is still about 5 to 10 %, depending on the location of the dam within the mangrove forest. This

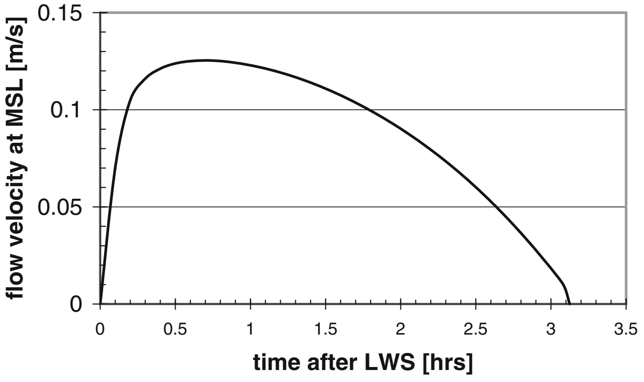

Fig. 7 Flow velocity over undisturbed mudflat computed at x=1,000 m (MSL, e.g. Fig. 6) as a result of tidal filling

may not seem a large reduction in onshore sediment flux, but it should be realized that this reduction is experienced every rising tide.

This reduction in the amount of water flowing towards the coast, cannot directly be translated into a reduction in net sediment flux towards the coastline, as this flux depends on a number of asymmetries in the water movement (tidal asymmetry) and sediment behavior (in particular settling and scour lag), e.g. Friedrichs (2011). However, when such asymmetries are taken into account, a reduction in tidal prism does result in a reduction in sediment flux, even while the amount of water flowing towards the coast during rising tide equals the amount of water flowing off the coast during falling tide.

Erection of a dam not only causes the on-shore sediment flux to decrease, but also induces reflection of incoming waves. In principle, standing wave patterns are formed with wave heights between 1.5 and 2 times the height of the incoming waves, depending on the substrate of the dam (porosity), e.g. Dean and Dalrymple (1984); Svendsen (2006); and Holthuijsen (2007). As bed shear stress scale with the wave height squared, eroding forces would increase by a factor 2 – 4. Because the incoming waves are irregular in general, they become uncorrelated rapidly, and the area of influence, i.e. the area with enhanced wave action is limited to 1 – 2 times the wave length. For the shorter waves of 2 – 4 s, this implies a length scale of 10 – 25 m seaward of the dam, whereas for the longer swell, this may be of the order of a few 100 m. These higher waves erode the soil at the foot of the dam, scouring sediment locally, inducing the unfavorable concave-up profile. When the waves are short and steep, cliff erosion may occur at the foot of the structure, rapidly destabilizing this structure. Figure 9 presents the results of a study on wave reflection by Klopman and van der Meer (1999) for a so-called Jonswap irregular wave spectrum, showing that close to the construction, wave heights increase by (1+R), whereas farther away this factor reduces to $\sqrt{1+R^2}$. The reflection coefficient R varies between 0 and 1, for full dissipation and full reflection, respectively.

Based on the reasoning above, Winterwerp et al. (2005) presented a qualitative model of the dynamical behavior of mangrove-mud coasts, following a conceptual description of the variation in location of the (mean) coastline cl:

$$\frac{\mathrm{d}cl}{\mathrm{d}t} = \text{sedimentation rate} - \text{erosion rate} \tag{4}$$

While sedimentation in mangrove-mudflat systems is primarily governed by tidal processes, sediment trapping by the mangroves, and consolidation of freshly deposited sediments, erosion is largely governed by surface waves. Note that mangrove-mud coasts are subject to erosive processes throughout the year, as even small, capillary waves can stir up the bed, owing to the fineness of the sediments. Larger waves are expected to have larger effect, of course, and may erode large amounts of sediment in a short period of time. As mangroves roots anchor only about 0.5 m deep within the soil (Tomlinson 1986), a few decimeters of erosion is sufficient to destabilize the mangrove trees. However, as argued in Winterwerp et al. (2005), the bigger

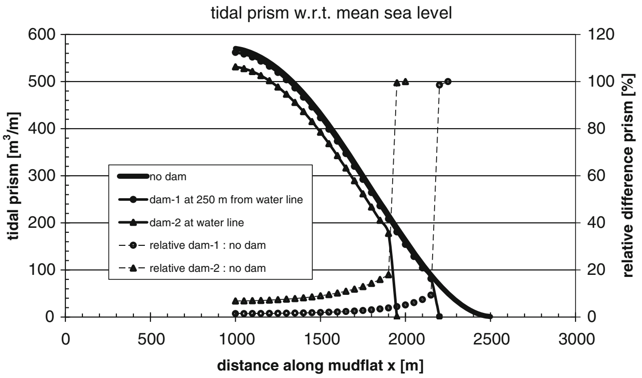

Fig. 8 Computed tidal prism P_T starting at mean sea level, along mudflat with and without dam at x=2,200 m (250 m from waterline) and at x=1,950 m (at high-waterline)

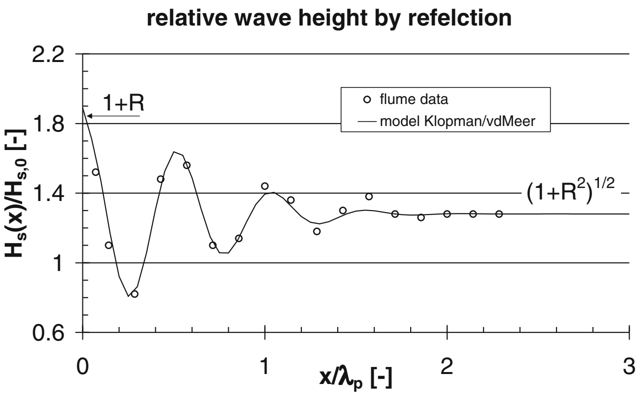

Fig. 9 Wave reflection against hard, impermeable structure (redrawn from Klopman and van der Meer 1999); R=reflection coefficient

waves do not only erode the muddy mangrove-mud coast, they also stir up fine sediments from the foreshore. These sediments are transported towards the mangrove-mud coast during rising water in response to the tidal filling of the upper parts of the mudflats, as discussed below. Hence, one can argue that larger waves give and take, whereas smaller waves, which do not stir up the foreshore, only take. This is the reason that coast-parallel breakwaters work contra-productive along mud coasts, as the off-shore sediment source is reduced, e.g. Winterwerp et al. (2005).

The simple coastline Eq. (4) allows for a qualitative assessment of a feedback loop that occurs following conversion of mangroves and subsequent infrastructure development along mud-coasts. The initial response to this land-use change is a reduction in sediment flux towards the coast, and a local increase in wave height, as discussed above. Then, Eq. (4) implies that the eroding wave-induced forces are no longer compensated by onshore sediment transport. The coastline starts to retreat. Fringes of mangroves that remain in the disturbed area are slowly lost to the sea. This reduces sedimentation rates substantially, as less mangrove trunks and roots remain to capture sediments coming in with the tides. As the retreating coast then develops a large-scale concave-up cross-sectional profile, wave effects start to dominate the sediment dynamics (see Fig. 10; Friedrichs 2011; and many other publications). Then, water depths in front of the coastline increase further, and waves can penetrate further towards the coast, enhancing erosion further. Figure 11 shows a schematization of the interactions leading to this snowball effect.

Recently, Anthony and Gratiot (2012) presented an analysis on the impact of land-use on the mangrove-mud coastal system of Guyana. Their major finding is that the loss of mangroves reduces wave dissipation within the mangrove area, promoting net erosion of the sediments in between the mangroves. This mechanism of reducing wave dissipation due to the loss of vegetation follows on our feed-back loop as described in Fig. 11, and can as such be interpreted as the second phase of coastal degradation, following the initial effects described above. Upon interventions, e.g. by placing seawalls or pond dikes in the mangrove area, it is the sediment dynamics that are affected first, and only later the vegetation (apart, of course, from the vegetation that was removed). The response of the coastal system to such interventions (or to coastal squeeze) includes a disturbance of the supply of sediment, i.e. the mechanism responsible for accretion, and a local increase in wave height by reflection against the sea wall. As the waves continue to do their job, the profile of the mudflat (within the mangrove area, and in front) changes, and then the wave-dominated processes described by Anthony and Gratiot (2012) start to play a role.

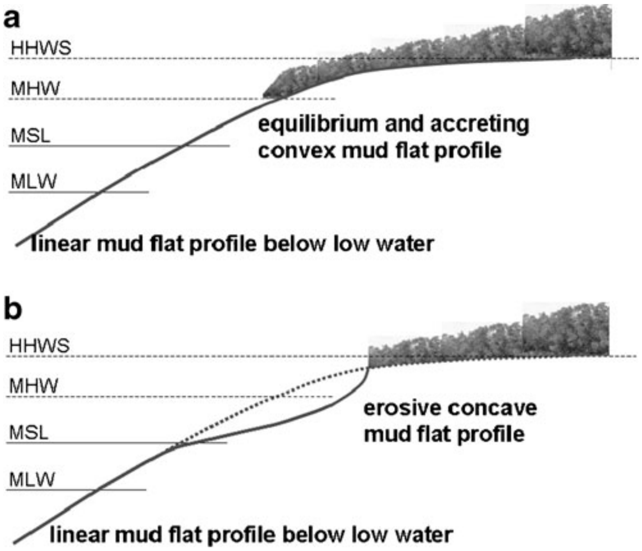

Fig. 10 Cartoon of equilibrium/accreting convex-up mudflat profile (left panel) and eroding convex-up mudflat profile (right panel). The latter profile allows for larger waves near the mangrove fringe

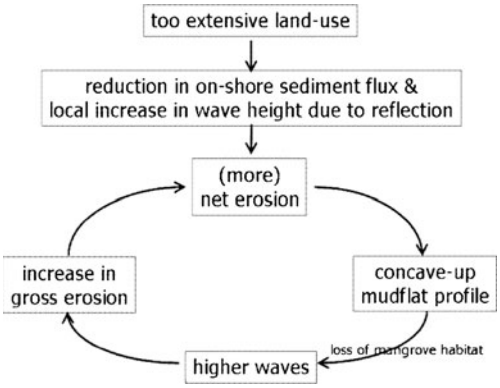

Fig. 11 Snowball effect for coastal erosion of mangrove-mud coasts induced by thoughtless land-use, such as extensive erection of fish/shrimp ponds

These interactions explain why mangrove rehabilitation along eroding mangrove-mud coasts has proven so unsuccessful; the erosive forces prevented natural recruitment of mangroves, while planted seedlings literally washed to sea soon after they were put in place. This does not mean it is impossible to restore eroding shorelines. Through a number of relatively simple measures, one may well be able to restore the morphodynamic requirements to facilitate mangrove growth and ensure that mangroves can resume their important role in protecting vulnerable coastlines. These options are presented in the discussion section below.

Discussion and Conclusions

In this paper we present an eco-morphodynamic analysis on the causes of erosion along degraded mangrove-mud coasts, and the reasons for the failure of attempts to rehabilitate these coasts. Our analyses are based on the results of studies in Thailand, British Guyana and Suriname, and observations in Indonesia, the Philippines and southern China, whereas the study by Winterwerp et al. (2005) is based on incomplete observations in Thailand only.

Besides socio-economic factors, which are beyond the scope of the present paper (but see Spurgeon 1998), ecological determinants of low success rates of rehabilitation efforts have been well explained in literature. In particular, various authors highlight the need to i) adopt improved techniques for mangrove rehabilitation, and ii) ensure reinstatement of appropriate hydrological conditions (e.g. in the case of abandoned shrimp pond areas). Here we add a third argument on the basis of a morphodynamic analysis of eroding mudflats: iii) restore morphodynamic conditions, e.g. the fine sediment balance, required for mangrove growth – the latter is our novel contribution to earlier work.

We conclude that the erection of dams/structures/bunds within or at the edge of mangrove forests for e.g. fish/shrimp farming or other land-use, initially affects the fine sediment balance in mangrove-mud areas in two ways:

- the on-shore fine sediment flux decreases as a result of a decrease in on-shore water flux (tidal prism),
- the wave height close to such structures increases due to reflection against that structure, inducing local scour in front of the structure.

This in-balance in fine sediment dynamics sets in motion an erosion process at a larger scale through which the shape of the mudflat becomes progressively concave-up, enhancing the wave effects further.

From our analyses and building on recommendations by Winterwerp et al. (2005), a strategy emerges for rehabilitating eroding mangrove-mud coasts, in which morphodynamic requirements are incorporated. Such integration requires an innovative joint application of mangrove rehabilitation with conventional engineering measures, also referred to as 'hybrid-engineering' or 'building with nature' (a new approach developed within the Dutch water sector that integrates ecological processes and engineering measures; see www.ecoshape.nl) .

This strategy contains the following elements:

1. Restore the onshore flux of fine sediment by restoring the intertidal area. This could be accomplished by creating a well managed buffer zone (green belt) in which incoming tides can freely flow. Any obstacles in this buffer zone (dykes, artificial mounds etc.) should be removed so as to provide this much-needed 'room for the sea'. A first assessment of the width of such a buffer zone can be obtained from the work by Friedrichs and Aubrey (1996), applying equation (2), and would measure several hundreds of meters, depending on the tidal range;
2. Enhance trapping of fine sediments on the mudflat in a natural way, for instance through the use of 'salt marsh works' – i.e. erection of small, permeable groins (made of locally available materials such as brushwood, poles, bamboo and/or flexible twigs) in the intertidal zone to enhance sediment accretion, followed by the excavation of ditches to improve drainage and accelerate plant establishment (Bakker et al. 2002). An additional advantage of such artificial reclamation works is the local reduction in long-shore currents, reducing long-shore transport away from the rehabilitation site of fine sediments by these long-shore currents; an example is shown in Fig. 12 for temperate salt marshes. It is noted that the authors are not aware of any 'salt marsh works' in mangrove areas, but there is no reason to doubt their efficiency;
3. Reduce wave heights by reducing reflections against the structures in the mangrove forest (or at the waterline) and/or with the permeable groins described above. This also follows from recent findings by Anthony and Gratiot (2012), who demonstrated the deleterious feedback effects of reduced wave dissipation following large-scale mangrove removal and construction of seawalls on the interaction between mudbanks and mangroves in Guyana;
4. Restore hydrological conditions, if disturbed, for example through rehabilitation of creeks or removal of small dams further inland, to ensure appropriate tidal inundation characteristics and sufficient freshwater flow towards the mangrove stands (sensu Lewis 2005). Where human modifications of the coast include large-scale conversion of mangroves to aquaculture ponds, restoration of these ponds back to mangroves – through hydrological

Fig. 12 Example of salt marsh works along the Wadden Sea (The Netherlands) – such salt marsh works, consisting of permeable groins made of willow twigs, have been deployed for many centuries, reclaiming substantial areas from the sea

restoration, e.g. by partial removal of pond dikes - should be considered, which would also assist with coastal stabilization, providing erosion protection from storm waves (Stevenson et al. 1999; Lewis et al. 2006);

5. Plant the proper mangrove species at suitable locations (above MHW) in areas where natural propagule supply is considered a constraint, and at the proper time, i.e. within "the window of opportunity" suggested by Balke et al. (2011). Note that in many cases, replanting is not necessary. As long as there is an abundant supply of propagules, mangroves will emerge autonomously as long as the environmental conditions are favorable.

Note that the last two issues have been addressed at length in many publications, but are repeated here as it is stressed that restoring the morphodynamic conditions alone is a necessary, but not sufficient condition to restore the eroding mangrove-mud coasts.

Further to this lists of critical steps, our analyses also explain why certain types of well-intended interventions should be prevented along eroding mangrove-mud coasts:

1. Do not erect sea defenses too close to the waterline (in between the mangrove stands). Such sea defenses will reduce onshore sediment transport further and aggravate erosion as discussed in the previous section, and inevitably lead to the destruction of the remaining mangrove stands in front of that sea defense. They might also cause severe damage to mangroves on either side of the sea defense as a result of disturbance of hydrology;
2. Do not deploy revetments to enforce the muddy coastline, as all hard structures in the end will initiate an unstable concave-upward mudflat profile preventing successful (natural) re-establishment of mangroves;
3. Do not erect coast-parallel breakwaters. Such breakwaters attenuate/eliminate the larger waves necessary to stir up fine sediment from the seabed, depleting the mudflat from the sediments to survive. Remember that even the smallest waves (capillary waves), generated beyond the breakwater by local winds, can erode the mud in between the mangrove roots;
4. Do not plant the wrong mangrove species (e.g. *Rhizophora* spp. at the sea front; see Primavera and Esteban 2008) and certainly not at inappropriate locations (too wet/low, wrong soil, wrong hydrology, wrong hydrodynamics) or wrong conditions (eroding mudflat) (Lewis 2005). Where natural propagule supply is not compromised and conditions are favorable, there is often no need for planting (Lewis 2005), as there is a good chance that mangroves will come autonomously. Such facilitation would allow seedlings to persist in what otherwise would be morphodynamically stressful locations, and encourage natural patterns of primary and secondary succession, which - in some parts of the world - may include a tidal marsh intervening stage with native marine emergent grasses such as *Spartina* (new world) or *Porteresia* (old world) acting as nurse plants to facilitate mangrove establishment (see Connell and Slatyer 1977; Ball 1980; Johnstone 1983).

Finally, we note that our analysis adds a further explanation why 'coastal squeeze' can be so devastating for the health of coastal wetlands such as mangroves and salt marshes. Coastal squeeze reduces the intertidal area ('room for the sea'), depleting these wetlands from the ready supply of sediments upon which their survival depends. The feedback between mudflat profile, onshore sediment transport and sedimentation room may induce a rapid decline of the wetlands.

Our analysis is based on a series of descriptive cases and a schematized analysis of the water movement and wave dynamics over tidal flats, and the numbers given herein are only approximate and qualitative. However, they comply well with actual field observations and explain many features of eroding mangrove-mud coasts as well as the many failed attempts to rehabilitate these. Our analysis may prove valid for salt marsh environments as well, as here the building tidal forces also compete with eroding waves, and the overall sediment balance may well prove to be playing a major role in the sustainability of salt marshes (Friedrichs 2011). We feel that a dedicated strategy combining ecological, hydrological and morphodynamic elements, accounting for local socio-economic aspects, will provide the best means to stop and even reverse the erosion of mangrove-mud coasts. Such a strategy should be curtailed to the conditions at hand, accounting for among others the local tidal conditions, wave climate, suspended sediment concentrations, freshwater supply, overall bathymetry, soil composition and autochthonous mangrove ecology.

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