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# Increasing coastal resilience through mangrove afforestation in the Mekong Delta

## International examples and lessons learnt



Technical Report

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Cover photo: Mangrove rehabilitation at Demak, Central Java, Indonesia (Courtesy: Wetlands International)

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# 1 INTRODUCTION

The Asian Development Bank (ADB) and the Government of the Netherlands have committed to support the Government of Vietnam in strengthening the coastal zones of the Mekong Delta and the resilience of its inhabitants. More specifically, this will be done by contributing to the early stages of project development for a mangrove afforestation project, contributing to the natural defence system for coastal protection. The ADB-financed project development will be implemented jointly with the Central Program Office of the Department of Forestry at the Ministry of Agriculture and Rural Development (MARD). The large-scale ADB-financed project is scheduled to be approved in 2023 and cover five coastal provinces (Bac Lieu, Soc Trang, Tra Vinh, Ben Tre and Tien Giang). The primary goal of the ADB-project is to restore the natural environment and strengthen coastal protection to increase resilience to the impacts of climate change.

A consortium of Royal Haskoning-DHV and Wetlands International was commissioned by the Netherlands Enterprise Agency (RVO) to assist the ADB by contributing to the early stages of project development for the mangrove afforestation project.

The present report addresses Task 1 of that assignment, which aims to compile international examples of mangrove ecosystem restoration / reforestation as 'green infrastructure' for coastal protection, and draw lessons learnt from those examples. The report is based on a comprehensive literature review, internet search, expert experiences and interviews to investigate and compile examples of mangrove ecosystem restoration for coastal protection and apply the findings to the Vietnamese situation.

The present report provides a meta-analysis of international examples of mangrove afforestation/restoration, an overview of the lessons learnt for nature-based solutions for coastal protection that have potential for the Mekong Delta, an analysis of the environmental and socio-economic perspective of the Mekong Delta to increase coastal resilience, and ends with conclusions of our findings.

## 2 INTERNATIONAL BEST PRACTICE EXAMPLES OF MANGROVE RESTORATION AND ASSOCIATED NATURE-BASED SOLUTIONS FOR COASTAL RESILIENCE

### 2.1 Overview of worldwide mangrove restoration efforts

#### Importance of mangroves

The evidence for the important values and functions of mangroves for humans and society is overwhelming. Mangroves are the most valuable and most productive coastal ecosystems on earth, and are the best carbon-capture-and-storage systems on this planet (McLeod et al., 2011; Donato et al., 2011). Through their irreplaceable role as nursery for fish and shrimp, among others, they support and sustain tropical fisheries (Zavalloni et al., 2014). They protect shorelines from waves, storm surge and coastal erosion, acting as self-repairing sea defences that can naturally adapt to the effects of climate change (Costanza et al., 2021). The awareness of the important benefits provided by mangroves and their economic value, has increased substantially over the last few decades (Barbier et al., 2011; Spalding and Leal, 2021). There has especially been a rapid rise in appreciation of the coastal protection offered by mangroves following the Indonesia tsunami in 2004 (Barbier, 2006, 2008).

#### Causes and scale of mangrove loss

High population pressure in coastal areas has led to the conversion of many mangrove areas to other uses, including aquaculture, salt production, rice farming and infrastructure. Some 35% of the total area of mangroves worldwide were lost by the turn of the century (Valiela et al., 2001). A first comprehensive global analysis of the world's mangroves by FAO in 2005 concluded that an estimated 3.6 million hectares of mangroves were lost worldwide between 1980 and 2005 (FAO, 2007a). A more recent study documented a total loss of approximately 15,262 km<sup>2</sup> of mangroves between 1996 and 2016 (globally), but also recorded some remarkable (mostly natural) gains in certain areas over that same period (Spalding and Leal, 2021). Some of the most extensive global losses have been in Southeast Asia, with dramatic changes occurring in Kalimantan, Riau and West Papua (Indonesia), the Sundarbans (Bangladesh and India), the Irrawaddy Delta (Myanmar) and the Mekong Delta (Vietnam). The causes of these losses are manyfold, but direct human impacts are responsible for over 60% of the total global mangrove loss. Primary causes include conversion to farmland (esp. for oil palm and rice), aquaculture (fish & shrimp ponds) and urbanisation. Natural or indirect human causes make up most of the remainder, including shoreline erosion, soil subsidence and storms, further exacerbated by sea level rise due to climate change (Spalding and Leal, 2021).

#### Mangrove restoration

Restoration or rehabilitation<sup>1</sup> may be recommended when a system has been altered to such an extent that it cannot self-correct or self-renew. Successful mangrove forest restoration

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<sup>1</sup> The terms 'afforestation', 'restoration' and 'rehabilitation' of mangroves are often used interchangeably, although they actually refer to different things. 'Restoration' refers to returning an ecosystem back to its original state or condition (as much as possible), 'rehabilitation' refers to improving, augmenting or enhancing

requires careful analysis of a number of factors in advance of attempting actual restoration, such as defining the existing watershed (its size and extent), understanding the general hydrology and sedimentology of the coastal plain or delta, and taking into account the history of the site to understand the human-induced changes to environmental conditions that may have contributed to the degradation and/or loss of the mangroves.

There is a growing body of literature and documentation available on mangrove restoration and/or rehabilitation. This includes scientific publications, workshop proceedings, restoration handbooks, nursery and planting guidelines, training manuals, individual restoration project reports, documentation by NGO's and community-based organisations, position papers and rehabilitation guidelines by international organisations. Some key references used for this review include Field (1996, 1998), Ellison (2000), Erftemeijer and Lewis (2000), Primavera and Esteban (2008), Lewis (2009), Dale et al. (2014), and Kairo and Mangora (2020).

### Learning from failures and successes

Worldwide, many attempts to restore or rehabilitate mangroves have failed completely, or failed to achieve their stated goals (Lewis, 2005). Despite heavy funds for massive rehabilitation of mangroves in the Philippines over the last two decades, during which over 44,000 hectares were planted, the long-term survival rates are generally low: 10-20% (Primavera & Esteban 2008; Samson & Rollon 2008). A similarly ambitious 5-year mangrove replanting program in Thailand targeting to replant 40,000 ha between 1991 and 1996, was evaluated as unsuccessful (Suwannodom et al., 1998). Many programs in South East Asia have targeted intertidal mudflats for mangrove replanting, but survival in such areas is typically low (Erftemeijer and Lewis, 2000).

**Table 1.** Common reasons for failure of mangrove restoration projects and lessons learnt (from: Kairo and Mangora, 2020)

| REASONS AS TO WHY RESTORATION EFFORTS FAIL                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                | LESSON LEARNT                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                             |
|-----------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|
| <ul style="list-style-type: none"> <li>- Use of inappropriate methods</li> <li>- Insufficient information</li> <li>- Failure to involve communities</li> <li>- Inadequate monitoring of seedlings after planting</li> <li>- Poor habitat selection without adequate site assessment</li> <li>- Faulty selection of mangrove species for replanting.</li> <li>- Improper planting of mangrove seedlings at a substrate depth beyond the natural range for mangroves in the area</li> <li>- Poor coordination among the institutions involved</li> <li>- Poor understanding of the ecological role of mangrove forests among policy and decision makers at different levels of government that result in weak support of restoration initiatives</li> </ul> | <ul style="list-style-type: none"> <li>- Diagnostic studies on socio-ecological aspects of mangroves</li> <li>- Embracing a more holistic approach</li> <li>- Engaging multiple experts and stakeholders</li> <li>- Linking local language to expertise of the scientific community</li> <li>- Monitoring and assessing success</li> <li>- Early identification of problems and taking corrective actions</li> <li>- Sharing knowledge and experience</li> <li>- Importance of mangrove valuation studies</li> <li>- Continued awareness and advocacy on mangrove conservation</li> </ul> |

It is now well-understood that many mangrove restoration projects have been unsuccessful due to an emphasis on planting of seedlings or propagules while neglecting the importance of hydrology and sediment dynamics. Lewis (2005) cites poor site selection, choice of wrong species, unsuitable hydrological regime, absence of clearly defined goals and a lack of follow-

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a degraded area with the expectation of a return of some improved level of ecosystem functioning, while 'afforestation' refers to planting of mangrove vegetation in areas where it previously didn't exist.

up monitoring and maintenance, as some of the main reasons for failure. Often, projects move immediately into planting of mangroves without determining first why natural recovery has not occurred. This typically results in major failures of planting efforts. Table 1 provides a summary of some common reasons for failure and lessons learnt.

On the other hand, there are an increasing number of examples where mangrove rehabilitation has been successful or mangroves recovered naturally after a period of degradation. These successes and failures offer an opportunity to learn from and derive key principles for successful mangrove rehabilitation (Field, 1996, 1998; Lewis, 2005, 2009).

### **Important general ‘common sense’ considerations**

If mangroves are not growing somewhere, there are two possibilities: [1] mangroves have never grown there because the conditions at the site are unsuitable for mangroves, or [2] mangroves used to grow there in the past, but they disappeared due to an adverse (human or natural) impact. In both cases, the environmental conditions are apparently not suitable for mangroves at present. As such, it would not make much sense to just start planting mangroves at such sites and expect any of the transplants to survive. Instead, the underlying cause of the mangrove loss (and lack of natural recovery) needs to be addressed first by improving the environmental conditions. Once conditions have been improved or returned back to what they were before the impact/disturbance, the mangrove vegetation may come back by itself, gradually recovering its former cover and ecological functioning with time. Restoration may help to overcome recruitment limitation (where there is limited propagule supply) and can sometimes speed up the recovery process.

Not all mangroves are the same (Friess et al., 2012). Consequently, what works in one place may not work in another. Restoration projects should therefore always consider to develop their own site-specific, project-specific restoration approach and design a plan that takes into consideration all relevant location-specific aspects and conditions.

The functionality of the restored mangroves should not be ignored (Bosire et al., 2008). What is needed is more than a collection of branches, roots and mud. Many of the ecosystem services will not simply be restored by some vegetation cover alone, but depend on the right mix of species (diversity) and the complex interdependencies between the various plant and animal species adapted to that particular place. Taking these aspects into consideration will also greatly contribute to the sustainability of the planted or restored mangrove stands and their resilience against external impacts.

Crucial for its success is the engagement of local coastal communities in any mangrove restoration or afforestation project. No mangrove rehabilitation effort is likely to yield a sustainable outcome without the support and engagement of local communities and consideration of their current land-use within the wider landscape. Where communities’ livelihoods depend on utilisation of the natural resources provided by the (former mangrove) areas targeted for restoration, it is imperative to consider alternative livelihood approaches/sources of income, to explore hybrid options such as integrated mangrove-aquaculture (silvo-fisheries) approaches, and to adopt a wider catchment/landscape-scale perspective (e.g. Mekong Delta Plan).

## Best practice approaches

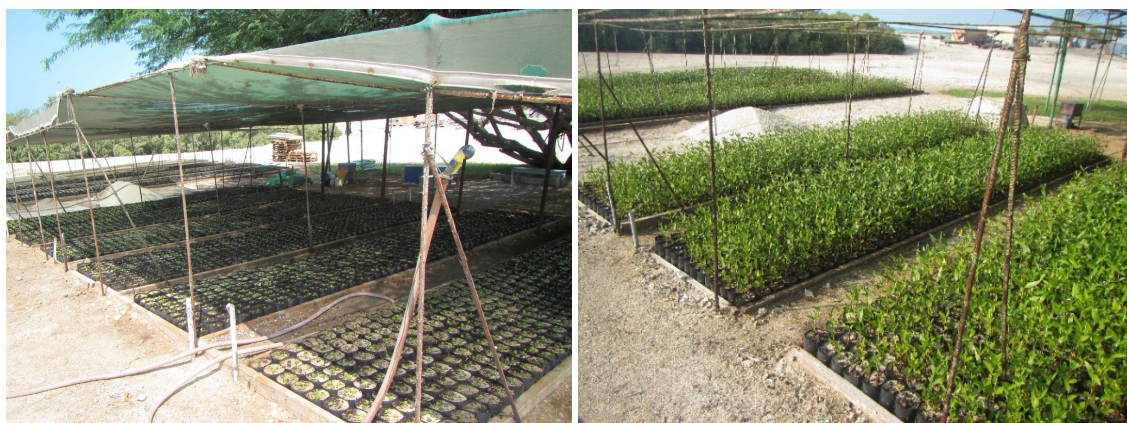
The preferred approach to achieve successful large-scale restoration is facilitating natural regeneration through recolonisation and succession, which can generally be accomplished by focusing on interventions that restore the appropriate environmental conditions (e.g. sediment supply & stability, tidal hydrology) instead of the indiscriminate planting of rows of one particular mangrove species based on practical convenience or commercial preference (Lewis, 2009). The best approach for mangrove restoration will also depend on the specific objectives of the ‘restoration’ effort. For example, planting mangroves for coastal protection may require a different approach than a restoration program that seeks to fully restore the mangrove ecosystem and its functions to its original reference state, including genetic- and species diversity, naturalness, trophic relationships and connectivity (Covington et al., 1999).

### To plant or not to plant?

Mangrove planting is hugely popular, but many of these planting efforts fail. A more effective approach is to create the right conditions for mangroves to grow back naturally, without the need for planting (Lewis, 2005; Kamali and Hashim, 2011; Wetlands International, 2018). Mangroves restored in this way generally survive and function better. In some cases, especially where recruitment of mangrove propagules is severely limited, for example when most mangroves (or certain species of mangroves) have disappeared almost completely over large geographic areas, planting can assist, speed up or enrich the natural recovery process. Planting programs can also be a powerful tool for environmental education or awareness building in rehabilitation projects involving local communities (e.g. school children) and in establishing a sense of local ‘ownership’ of the restoration outcome that can contribute to its sustainability in the long term (Quarto, 1999; Erftemeijer and Bualuang, 2002).

### Mangrove tree nurseries

Where active planting is desirable, the establishment of a mangrove tree nursery to raise large quantities of seedlings for the planting program may contribute to effectively achieving restoration at larger scales (Figure 1). There are several technical manuals available on how to set up a mangrove tree nursery (Siddiqi et al., 1993; Clarke and Johns, 2002; Thoi and Thinh, 2010; Ramasubramanian et al., 2020).



**Figure 1.** Impressions of a mangrove tree nursery at Mubarratz Island (Abu Dhabi, UAE)



### **Mangrove restoration – general considerations:**

- Mangroves play a key role in protecting shorelines from waves, storms and erosion, acting as self-repairing sea defences that can naturally adapt to climate change.
- Widespread losses of mangroves have driven global restoration efforts in recent decades.
- Many attempts to restore mangroves have failed completely or did not achieve their goals.
- Common reasons for failure include poor site selection, planting the wrong species and unsuitable conditions (especially tidal hydrology).
- Never rush immediately into planting, but first understand the system and the site conditions.
- First address the underlying cause of mangrove loss and lack of natural recovery.
- Planting is often not necessary; once environmental conditions are restored, mangroves will generally recover naturally (unless there is recruitment limitation).
- Planting mangroves at locations where they have never grown before is unlikely to succeed without significant site modifications (engineering) to create environmental conditions conducive to mangrove growth.
- Mangrove rehabilitation is unlikely to yield a sustainable outcome without the support and involvement of local communities and consideration of current land-use.

In the following sections, a summary of best practice approaches is presented for mangrove restoration after the degradation and loss of mangroves. Descriptions are based on an analysis of available literature from numerous case studies of mangrove rehabilitation efforts around the world (supplemented with personal experiences and observations by the authors; e.g. Figure 2). We highlight some of the most critical factors that should be considered when embarking on a mangrove rehabilitation program. The practical application of these various approaches for the program in Vietnam’s Mekong Delta will be discussed in the next Chapter.



**Figure 2.** Mangroves planted for shoreline protection in Kiribati (South Pacific)



## 2.2 Mangrove restoration along eroding coasts

The long-term sustainability of mangrove shorelines (in particular in the humid tropics) depends first and foremost on an adequate supply of sediment (Anthony and Goichot, 2020; Liu et al., 2021). In areas where mangrove vegetation has been removed along significant stretches of coastline, or where riverine sediment supply has been markedly reduced due to the construction of upstream hydropower dams, sediment instability may occur, shifting the balance from accretion to erosion (Winterwerp et al., 2005; Besset et al., 2019a). In the absence of the stabilising vegetation, muddy shorelines tend to fall into a state of perpetual erosion following mangrove removal (Figure 3). This is further worsened in areas where the natural sediment supply has been compromised, or where soil is subsiding due to excessive groundwater abstraction. Rehabilitation of mangroves under such circumstances can be particularly problematic, but potentially successful solutions have been described in Stanley & Lewis (2009), Anthony and Gratiot (2012), Winterwerp et al. (2013), Saengsupavanich (2013), Van Bijsterveldt et al. (2020), Erftemeijer et al. (2020) and Winterwerp et al. (2020).



**Figure 3.** Eroding (former mangrove) coastlines (Thailand)

The most successful solutions have adopted a hybrid engineering approach, combining (semi)-permeable groyne systems (e.g. bamboo fences) to promote sediment deposition with replanting of mangroves. Where coastal erosion along muddy shores is severe, additional measures can be considered, including low-crested revetments (though with variable results) or increasing sediment supply to the shore through jetting using dredgers (being tested in a trial in Indonesia), combined with the hybrid engineering solutions discussed above.

Sediment entrapment by multiple upstream hydropower dams, along with other river catchment management interventions, reduces sediment supply in river deltas, which can be overcome in several ways (Kondolf et al., 2014; Jones, 2020). One way to achieve this is through the construction of sediment bypass tunnel systems at the dams. These allow for controlled sediment release by diverting a portion of the sediments flowing from upriver around dams and allowing these to re-join the river downstream (Kondolf et al., 2014). Such controlled sediment flushing can also help to reduce reservoir siltation and extend the longevity and storage capacity of reservoirs (Espa et al., 2019).

There have been recent efforts in Guyana and Suriname (South America) to restore mangrove forests along the coast at places where the natural belt had disappeared or became too narrow to be sufficiently resilient (Anonymous, 2013; Toorman et al., 2018). Based on an

understanding of natural processes along the mangrove coastlines in Suriname (Erftemeijer and Teunissen, 2009) and Guyana (Machin and Lewis, 2015), and in-depth studies of the interactions between mangroves and hydrodynamics and between mangroves and sediment (Winterwerp et al., 2013; Toorman et al., 2018), and the effects of human interference on these processes (Anthony and Gratiot, 2012), restoration efforts involving a combination of sediment trapping and mangrove planting appear to show potentially promising results (Toorman et al., 2018), although there is need for more evidence from follow-up monitoring.



**Figure 4.** Permeable groynes (constructed from bamboo and brushwood) along the Guyana coast

Mangrove rehabilitation approaches implemented and tested in Guyana included the construction of sediment traps and permeable groynes/breakwaters constructed from bamboo and brushwood (Figure 4) in combination with the planting of pioneer mangroves and *Spartina* grass. Activities that accompanied these approaches included (1) the construction of restrictive gates and fences to reduce the impact of anthropogenic activities, (2) various community-based mangrove management and livelihood initiatives (such as tourism and beekeeping), and (3) an extensive public awareness and education campaign (Anonymous, 2013; Toorman et al., 2018; Winterwerp et al., 2020).



**Figure 5.** Impression of mangrove restoration results in Guyana

The Guyana Mangrove Restoration Project evolved after some initial failures in earlier plantings (pilots). Through adaptive management and application of the basic principles of Ecological Mangrove Restoration (EMR), the project has achieved increasingly promising results (Figure 5; Anonymous, 2013), perhaps partly aided by the arrival of a migrating mud bank. Restoration efforts between 2010 and 2015 involved the planting of some 500,000 *Avicennia germinans* seedlings (and in some areas *Spartina* grass) covering a total area of approximately 142 hectares. These efforts produced mixed but promising outcomes,



including the establishment and rapid growth of some 35 hectares of dense mangroves (Anonymous, 2013; Machin and Lewis, 2015; Adams, 2016). Similar efforts were undertaken in Suriname (see: <https://www.conservation.org/suriname/programs/building-with-nature>), but there is need for follow-up monitoring data to verify early claims of success.

A coastal protection project along an eroding mangrove coastline at Chachoengsao Province in Thailand achieved promising results with a combination of low revetments and bamboo fencing (Figure 6; Saengsupavanich, 2013). With a strong emphasis on stakeholder involvement and consideration of opinions from coastal communities, this project further demonstrated how various factors, such as engineering constraints, budget and resources limitations, and power distribution among different government departments complicated problem solving. However, in the end, stakeholder coordination resulted in the selection and implementation of an effective coastal protection approach with stakeholder acceptance.



**Figure 6.** Bamboo fencing and successful mangrove planting behind such fences along an eroding shoreline at Chachoengsao Province (Thailand)

At Demak in Java (Indonesia), permeable dams were built along 13.5 km of coastline suffering from severe erosion following mangrove loss, as a pilot project to test the feasibility of this innovative coastal rehabilitation approach (Figure 7) under the Building with Nature program (Winterwerp et al., 2016). Initially, the permeable dams trapped considerable amounts of sediment which raised the overall bed levels behind the dams. At some locations spontaneous mangrove colonisation was observed, and local communities planted *Rhizophora* in several of the sedimentation basins (Figure 8). However, follow-up monitoring revealed that in the longer term sedimentation rates reduced considerably. Mangroves survived in only a few of the most protected sedimentation basins. Elsewhere, juvenile mangroves disappeared after one or two years, most likely because the relative bed level dropped below mean sea level again. Besides consolidation of the deposits, subsidence was the main culprit, a process which had initially been underestimated. Consequently, the dams, which were initially emerging above MHW, subsided significantly (by some 25 cm) and gradually disappeared below MSL, thus becoming less and less effective in trapping sediment. In spite of these uncertainties, this Building with Nature approach for coastal rehabilitation was adopted by the Indonesian government and a total of 23.3 km of permeable dams have since been constructed at various locations in Java, Sumatra, Kalimantan and Sulawesi (Winterwerp et al., 2020).

### *Permeable structures trapping mud for mangrove recovery*

Healthy mangrove mud coasts are in a dynamic equilibrium; waves take sediment away and tides bring sediment in. The mangroves' root system helps to capture and stabilize the sediment. Nowadays, many tropical mud coasts face dramatic erosion resulting from the conversion of mangroves, disturbing infrastructure, sea level rise and soil subsidence. Coastal managers tend to fight coastal erosion with hard structures, but these disturb the balance of incoming and outgoing sediment and may cause further erosion. To stop the erosion process and regain a more stable coastline, the first necessary step is to reverse the loss of sediment. Permeable structures made of local materials

such as bamboo, twigs or other brushwood can be placed in front of the coastline. These structures let sea water pass through, dampening the waves rather than reflecting them. As a result, waves lose height and energy before they reach the coastline and mud can settle behind the structure. Once the erosion process has stopped and the shoreline starts accreting, mangroves can re-establish without being washed away. Over time, the mangroves themselves will attenuate waves and trap sediment again thus preventing erosion. This technique is currently being applied in the Mekong delta (Vietnam) and in Demak (Indonesia).



**Figure 7.** Permeable structures trapping mud for mangrove recovery (Source: Wetlands International, 2018).





**Figure 8.** Successful mangrove restoration along an eroding coast at Demak in Java (Indonesia) using permeable groyne structures to trap sediment

In each of the Guyana, Suriname and Indonesia case studies, construction of expensive, hard-structure sea defence engineering solutions such as seawalls, breakwaters and rock revetments all proved unsuccessful in stopping shoreline erosion. These hard structures restricted sediment movement, formed a barrier for propagule dispersal, compromised mangrove health and worsened erosion through the deflection of wave energy and scour. In contrast, low-cost and low-tech solutions such as permeable bamboo or brushwood groyne structures and parallel geotextile tubes have been much more successful, significantly increasing sediment deposition and stability, creating an environment conducive to (natural) seedling establishment and mangrove recovery. There is currently still no clear design technology for these permeable dam structures, but all were based on ‘best engineering practice’ combined with an adaptive approach that enabled learning through monitoring and analysis (see Wilms et al., 2020).

Sediment Trapping Units (STU) can be constructed with fences of vertical (bamboo) poles. We advocate the use of such fences instead of the common brushwood permeable structures, as the latter are less practical, requiring more expertise and skill to construct and more maintenance. Brushwood is lost easily, either by hydrodynamic stresses, or by biodegradation. Winterwerp et al. (2020) argue that mangrove habitat can develop in STU’s behind the dams if not more than about 50 – 60% of the incoming wave height is transmitted through the dams. This is reflected by the transmission coefficient  $k_t$ :

$$k_t = H_{s,0} / H_{s,t} \quad (\text{equ. 1})$$

where  $H_{s,0}$  and  $H_{s,t}$  are the significant wave height of the incoming and transmitted wave, respectively. Based on the work by Gijón et al. (2021a, 2021b), the transmission coefficient as a function of the fence porosity  $n$  can be established for one row of fences and for two rows, assuming an effective drag coefficient  $c_D = 5$  and pole diameter  $D = 0.1$  m (see Figure 9).

The porosity is defined as:

$$n = d / (D + d) \quad (\text{equ. 2})$$

where  $D$  is the diameter of the poles, and  $d$  their spacing (Figure 9). This may be achieved by a fence of one row of bamboo, with porosities not larger than about 20% ( $d < 0.25D$ ) at a

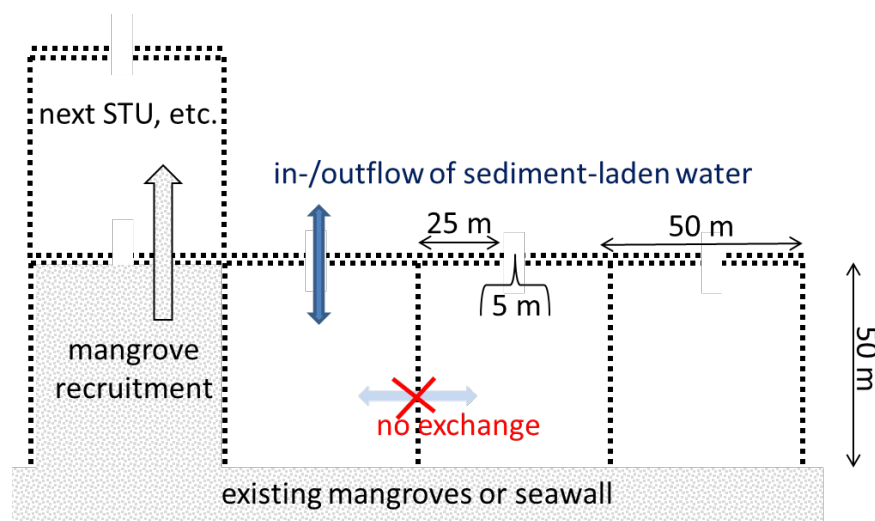


water depth  $h = 1$  m, and wave height  $H_s = 1$  m. However, the transmission coefficient is rather sensitive to water depth and wave height, as shown in Figure 9. However, the larger wave damping is required during stormy conditions, thus larger wave height. It is therefore recommended to experiment with a double row of bamboo fences, measuring efficiency of the single and double row configuration, as a double row provides redundancy in case of damage. Such redundancy must be weighed against the extra costs. These rows are placed at a distance of at least three diameters of the bamboo poles. It is noted that small spacings lower than 5 – 10% porosity are not recommended because of augmented wave reflection.



**Figure 9.** Indicative transmission coefficient  $k_t$  as function of fence porosity  $n$  for a single and double row fence and as a function of water depth  $h$  and wave height  $H_s$ , and definition of fence parameters (frontal view).

STU's are placed on the intertidal and we recommend STU's as large as possible to build out fast as possible, ranging from  $50 \times 50$  m<sup>2</sup> to  $200 \times 200$  m<sup>2</sup>. We recommend experimenting with varying the STU's size when deployed at a new site to optimise dimensions with respect to the local conditions.



**Figure 10.** Schematic diagram of serial  $50 \times 50$  m<sup>2</sup> STU deployment.

In all cases, 5 m openings must be constructed every 25 m or so in the fences to allow entrance of sediment-laden water. Stand-alone and neighboring STU's have "side walls" to prevent uneven distributions of deposition. The frontal fence, facing approaching waves may be constructed from double fences; for the side walls a single fence is likely sufficient under all

conditions if properly maintained. The openings are made in the frontal fence, generally facing the incoming waves, though other configurations may be considered when practical. Figure 10 (above) presents a schematic diagram of a serial application of STU's of 50×50 m<sup>2</sup> and subsequent extension in offshore direction. After some time, a new series of STU's can be constructed in front of the previous series, when mudflat heights are sufficient, and mangrove recruitment is advanced.

At Mubarras Island near Abu Dhabi in the United Arab Emirates, a different innovative approach was adopted, involving the planting of nursery-reared mangrove seedlings in artificially created tidal channels behind a temporary berm of dredged material that was erected to protect the young plantings from the sea (Figure 11; Erftemeijer et al., 2020). Once the mangroves have firmly established themselves and grown into mature stands, they start to function as natural sea defence after the protective berm of dredged material has eroded away. This novel approach offered a way to overcome the paradigm that mangroves require sheltered conditions for their initial establishment but, once established, provide effective protection of shorelines against the erosive forces of waves and storms (Erftemeijer et al., 2020).

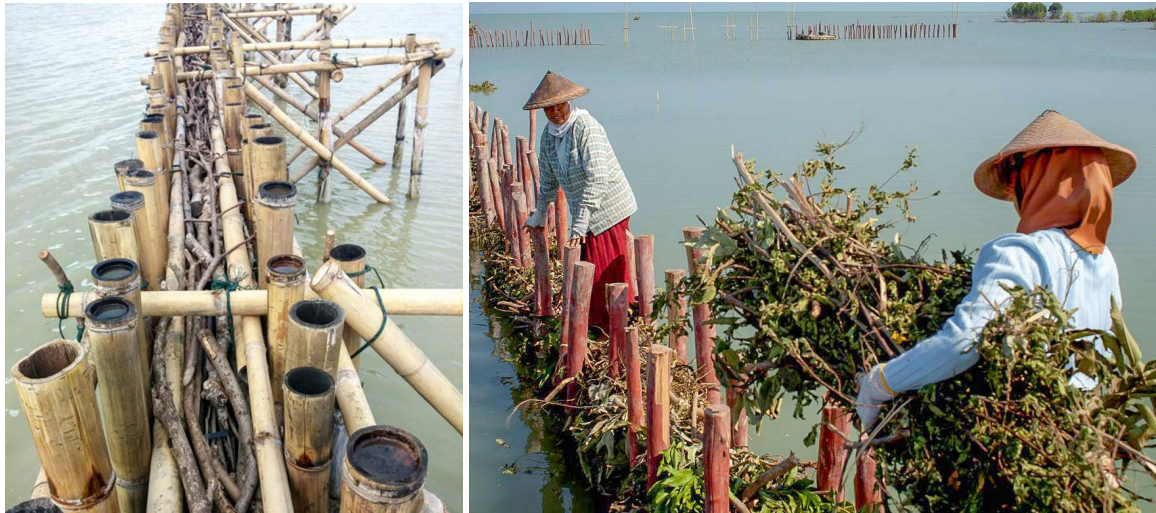


**Figure 11.** Mangrove planting inside artificially created tidal channel along a shoreline, temporarily protected from the sea by a berm of dredged material, to offer the newly planted mangroves time to establish and develop into a more mature stand that can function as a protective green belt against erosion (Abu Dhabi, UAE)

### Challenges for mangrove restoration along eroding shorelines

- **Floating debris and logs:** Saengsupavanich (2013) reported damage to their mangrove planting site by floating debris (from degrading bamboo fences). Floating (natural) debris, like dead mangrove tree logs and other wood can also be problematic, e.g. after typhoons, cyclones and tropical storms, or along severely eroding sites. Removal of such debris should be considered when it is causing substantial damage on plantings, blocking influx of propagules or preventing establishment of seedlings. Otherwise, it may be better to leave the debris, since it can contribute to wave attenuation. Remaining mangrove stumps are also best left in place, as their (dead) root systems still keep the sediment together and thus contribute to shoreline stabilisation.

- **Extreme conditions:** Extreme weather (e.g. storms, typhoons) can damage the restoration sites and constitute erosive events that can remove large quantities of sediment that had gradually built up behind the permeable groynes.



**Figure 12.** Close up impressions of permeable groynes, constructed from bamboo (left) and local wood poles with brushwood (right), as tested in Java, Indonesia

- **Degradation of fences/groynes:** Problems with degradation of the bamboo fences into debris within two to three years were reported from Thailand (Saengsupavanich, 2013). Problems with the durability of the groyne structures were also reported from Demak in Indonesia, where shipworm infestations caused rapid degradation of earlier dam designs built using poles of local wood (*Melaleuca cayuputti*). Bamboo proved a better alternative, although its lifespan was limited to two to three years. There were also problems with the availability and costs of large quantities of brushwood, and with the stability of the brushwood within the structures (kept in place with nets but easily damaged) and its rapid decay (Winterwerp et al., 2020). Ideally, the permeable structures need to stay in place long enough for mangroves to recover, which is the sum of the estimated sediment accretion rate and mudflat stabilisation (3 – 5 years), and rate of mangrove recovery (4 – 5 years). Materials for the vertical poles tested in Indonesia included bamboo, wooden poles (mangrove and nibung palm), coconut trunks, PVC (filled with concrete or empty), and steel (I or H beams) (Figure 12). Natural poles were often attacked by shipworm and generally lasted 1-2 years. The dams can be repaired, maintained and reinforced with PVC + concrete piles. Once mangrove colonisation behind the dams is well developed, the dam can be abandoned (Wilms et al., 2020).
- **Subsidence:** One of the challenges encountered by some mangrove restoration programs along eroding shorelines is (shallow) soil subsidence. Subsidence is a gradual settling of the ground owing to the subsurface movement of unconsolidated sediments. Subsidence usually occurs due to a combination of causes, such as groundwater (over)exploitation (esp. near urban centres), compaction of recent sediment deposits and a reduction in sediment supply (esp. in river deltas) due to hydropower dam construction in upstream catchments (Cahoon and Lynch, 1997; Lovelock et al., 2011; Swales et al., 2016). It can be

a challenge for mangrove restoration efforts when the permeable groynes 'sink' below critical levels, making them less effective in trapping sediment. Subsidence has also been reported as a challenge in the Mekong Delta, where it is exacerbating the risks and damages from sea level rise, floods and saline water intrusion (Tagaki et al., 2016)

### **Mangrove rehabilitation along eroding shorelines**

#### **Specific considerations:**

- Mangrove shorelines affected by significant erosion will not recover by themselves.
- Conventional engineering solutions and mangrove replanting fail in such areas.
- Potential solutions tend to adopt a hybrid engineering approach, e.g. by combining permeable groins (to promote sediment deposition) with replanting of mangroves.
- In areas with significant land subsidence or drastically reduced sediment supply, the effectiveness of such permeable structures decreases.
- Upstream sediment bypass systems at hydropower dams, sediment flushing at reservoirs and other upstream river catchment interventions can alleviate such sediment deficiencies, while extending the lifespan of the dams.
- The creation of a berm of dredged material or sand (with openings to allow for tidal inundation) can offer temporary shelter from the sea, allowing planted mangroves to establish and grow into mature stands before starting to function as a natural sea defence once the berm has eroded away.
- Where erosion is particularly severe, construction of low-crested revetments or interventions aimed at increasing the sediment supply to the shore (e.g. through jetting using dredgers, and/or catchment management interventions) may be considered in addition to those hybrid engineering solutions.
- **Key: restore sediment supply & stability.**
- **Tools:** permeable groynes (and upstream sediment bypass systems).
- **Challenges:** floating debris, limited life-span of groyne structures and subsidence.



## 2.3 Mangrove restoration after hurricanes or cyclones

Mangroves offer significant coastal protection against the impacts from severe storm events (Dasgupta et al., 2019; Costanza et al., 2021). However, cyclones, hurricanes, severe tropical storms and associated flooding events can cause substantial damage and die-off of mangroves (Figure 13; Jimenez and Lugo, 1985; Erftemeijer and Hamerlynck, 2005; Paling et al., 2008; Asbridge et al., 2018; Radabaugh et al., 2019; Salmo, 2021). Mangroves often do recover naturally from such damage or mortality and this recovery (by refoliation or resprouting) can sometimes be quite rapid (Krauss and Osland, 2020; Salmo, 2021). In March 1999, Cyclone Vance destroyed some 5,700 hectares of mangroves in the eastern Exmouth Gulf, northwestern Australia. Within a period of five years following the cyclone, 68% of the destroyed mangrove areas had already recovered through natural regeneration (Paling et al., 2008). There are other (similar) reports of natural recovery following severe storm events (see Roth, 1992, and reviewed in Paling et al., 2008).



**Figure 13.** Cyclone damage to coastal mangrove areas in Queensland, Australia (left) and the Philippines (right)

Sometimes, there may be scope for human intervention to facilitate the recovery of mangroves following damage by hurricanes, cyclones, typhoons, severe tropical storms and associated flooding. This is especially the case when there is evidence of significant (but usually localised) hydrological changes, such as ponding of certain areas through blocking tidal flows in and out of a mangrove site, lagoon, tidal creek or connecting channel, that may be caused by the shifting of major quantities of sediment during such extreme climatic events. This can easily kill-off mangroves when they are no longer experiencing the normal durations and frequencies of tidal exposure and inundation (Erftemeijer and Hamerlynck, 2005). In such instances, it would make sense to restore the inundation regime (tidal hydrology) by opening blocked areas in order to try and speed up natural processes of recovery (see Lewis, 2005). In rare occasions, removal of excessive shrub vegetation (e.g. *Acrostichum* ferns) or logs, wood and debris may be considered, if these hinder the influx of mangrove propagules or successful establishment of seedlings, and hamper the recovery (Erftemeijer and Teunissen, 2009).

As most mangroves are naturally adapted to natural disturbances such as (severe) storms and cyclones and are generally able to recover naturally from the damaging effects of such events, the primary focus of the post-cyclone mangrove rehabilitation efforts should be on alleviating anthropogenic sources of stress that could compromise the resilience of coastal mangroves against future cyclones (Krauss and Osland, 2020).

## **Mangrove rehabilitation after hurricanes or cyclones**

### **Specific considerations:**

- Mangrove areas impacted by hurricanes or cyclones most often will recover by themselves (15-30 years).
- Rehabilitation efforts may speed up the recovery process.
- Access to affected mangrove areas can be temporarily impeded by excessive flood waters (sometimes for weeks) following heavy rains after major cyclones (safety first).
- Post-cyclone mortality of mangrove trees from excessive sediment deposition and prolonged waterlogging may manifest with considerable delay (lag-time of months).
- Removal of logs and debris (or excessive secondary shrub vegetation) from affected areas may sometimes be considered.
- Restore (tidal) hydrology if the cyclone has shifted major quantities of sediment into the mangroves (causing ponding and mortality).
- Only consider planting if there is insufficient natural recruitment of propagules.
- **Key: remove obstacles to tidal flows and influx of propagules.**

## 2.4 Mangrove restoration in acid sulphate soil areas

In some waterlogged coastal areas, soils can be potentially acidic, which becomes a problem when these soils are disturbed (e.g. by excavation or dredging) or drained (exposed to air). The ecological effects of acid sulphate soils and the potential management policies and remedial measures have only recently begun to be clearly identified (Tri, 1996, Thomas et al., 2003; Powell and Martens, 2005). The acidic water which results from acid sulphate soils destroys food resources, displaces biota, releases toxic levels of aluminium, precipitates iron which smothers vegetation and microhabitat and alters the physical and chemical properties of the water (Sammut, 2000). The highest risk of acidic discharges into the environment from disturbed acid sulphate soils is likely to be in times of high rainfall, washing out the acidity and contaminants from the exposed soils (Russell et al., 2011). Export of acidity generated within soil pores into drains, water courses and estuaries is generally associated with rainfall events (Wilson et al., 1999), causing episodic fish kills.



**Figure 14.** Acid sulphate soil in Queensland (left) and lime-treated acid sulphate soil from mangroves in Darwin (NT, Australia)

While clearly affecting mangrove-associated fauna, there is little evidence to suggest that disturbance of (potential) acid sulphate soils and subsequent acidic drainage affects the growth and health of the mangrove trees themselves. However, high sulphide levels and reduced redox conditions can prevent mangrove seedling recolonisation of cleared areas (Smith et al., 1995; Isyrini, 2014). It is appropriate to consider treatment of exposed acid sulphate soil material (e.g. when dredged or excavated to create a trench for pipeline installation) in contained treatment pans with lime prior to disposal or re-use of the material (e.g. as topsoil layer for backfilling) to mitigate potential impacts from acidic runoff on mangrove-associated fishes, shrimps and other fauna in the vicinity of the site (Figure 14).

Lime-assisted tidal exchange rehabilitation efforts carried out in north Queensland have demonstrated that it is possible to considerably reduce acidification impacts and that the resulting improvements in soil and water quality can lead to rapid recolonisation of mangrove vegetation and successful recovery of fish and crustacean communities in areas affected by acidification (Russell et al., 2011; Luke et al., 2017). A recent study in abandoned shrimp pond areas in Sulawesi (Indonesia) characterised by acid sulphate soils, similarly demonstrated beneficial effects of restoring tidal hydrology on improving soil geochemical conditions and enhancing mangrove seedling establishment, survival and growth (Isyrini, 2014). Beneficial effects of tidal buffering on acid sulphate soil environments following hydrological restoration

were also previously reported by Indraratna et al. (2002). Such tidal flushing can be effective in addressing acid sulphate issues because the pH of seawater is high, which helps in neutralising the acidity, although this process may take time.

### **Mangrove rehabilitation in areas with acid sulphate issues**

#### **Specific considerations:**

- Disturbance of (potential) acid sulphate soils and subsequent acidic drainage – while posing a potential problem for mangrove-associated fauna (esp. fishes and shrimps) – seems to have little impact on the growth and health of the mangroves themselves.
- Avoid mangrove removal and excavation works in areas of potential acid sulphate soils during periods of heavy rainfall to prevent washing out of acidity and contaminants.
- Acidity does not pose constraints to natural mangrove recolonization of cleared areas.
- Rehabilitation efforts may speed up the recovery process.
- Treatment of excavated acid sulphate soil with lime in treatment pans prior to its (re-)use as backfill material may be considered to mitigate impacts from acidic run-off on fauna.
- A combination of lime treatment and rehabilitation of tidal exchange (natural flushing) can be a very effective approach to mitigate acidification impacts and restore affected areas.
- **Key: lime treatment and tidal flushing.**



## 2.5 Mangrove restoration after oil spills

Over the past six decades, there have been at least 238 notable oil spills along mangrove shorelines worldwide, releasing some 5.5 million tonnes of oil into mangrove-lined coastal waters, oiling around 2 million ha of mangrove habitat, and killing at least 126,000 ha of mangrove vegetation (Duke, 2016). Chronic impacts on mangroves have been observed across long periods (up to decades) following oil spills, with the symptoms most commonly observed including death of trees, defoliation and canopy thinning, leaf yellowing, reduced growth and height of surviving trees, and poor seedling establishment (Duke, 2016).

Mangroves may take anywhere from a few years to >30 years to recover from oil spill impacts (Burns et al., 1993; Hoff et al., 2010). Restoration projects may attempt to speed up this recovery process. Adequate tidal exchange is most critical to restoration success of oil-impacted sites. An oil spill alone rarely changes the basic geophysical appearance and shape of the mangrove ecosystem. For this reason, restoration after an oil spill may be easier than after an event that substantially changed tidal elevation or hydrology or decimated mangrove trees (Paling and Erftemeijer, 2013). However, an oil spill may often come as an additional impact on a mangrove ecosystem already degraded by human and industrial development, such as nearby refineries, ports or airfields. Cumulative or chronic impacts may decrease the resiliency of the mangrove ecosystem and increase the time it takes the system to recover or make it more difficult for the system to recover at all.



**Figure 15.** Mangrove area affected by an oil spill (left), and containment and clean-up efforts (right)

Post-spill recovery and restoration success depends in part on the response efforts during/after the spill itself (Figure 15). Non-beneficial spill response efforts that have degraded mangroves include forest cutting, impoundments, heavy equipment and personnel traffic, tree burning, onshore use of chemicals, and plantings in toxic soils (Getter and Lewis, 2003). The only long-term beneficial spill response method appears to be a combination of non-intrusive oil collection and booming techniques in heavily oiled, sheltered areas and close monitoring of natural recovery in lightly oiled mangrove forest areas (Getter and Lewis, 2003).

As with other ecosystems adversely impacted by oil spills, there are valuable lessons to be learnt from past mangrove restoration projects, including those that failed. Restoration projects need a clear goal from the outset that is based on understanding the mangrove ecosystem's natural ability to recover. Natural recovery of damaged or impacted mangrove forests after an oil spill depends on adequate supply of mangrove seeds/propagules and the

return of acceptable growth conditions for these seeds/propagules. Where the supply of propagules and/or densities of naturally recruited seedlings appear insufficient to allow adequate regrowth, broadcasting of seeds (propagules) or active planting of an oil-damaged site may be considered (Lewis, 1983; Getter et al., 1984).

If planting is chosen as the best course, seedlings will survive best when they are planted in a sheltered location and at appropriate tidal elevation levels for each species. Planted seedlings are often lost primarily because of erosion, predation, mortality from natural causes, planting at incorrect tidal elevations, and residual oil toxicity (Getter et al., 1984). Seedlings and propagules can survive even when planted in soils with residual oil contamination, though generally only after oil has weathered for 9-12 months. A large-scale restoration planting project at St. Croix in the U.S. Virgin Islands planted seedlings 8 years after heavy oiling from the Santa Augusta spill, with 40% survival after two years (Lewis 1979).

Residual oil that has contaminated soils in mangrove forests degrades very slowly, since these soils are typically anaerobic below the top few mm (Burns et al. 2000). Experiments and field studies examining the possibility of accelerating oil degradation through addition of nutrients or increased aeration have shown little advantage to these methods. During the first year after a spill, biodegradation occurs at very low levels, and the main routes of oil removal are dissolution and evaporation. Thus, it is critical during spill response to attempt to keep oil from penetrating into sediments. Some restoration-planting projects surround seedlings with clean, fertiliser-augmented soil so the new trees can establish themselves and develop root structures in uncontaminated soils, before having to contend with possible toxic effects from residual oil (Paling and Erftemeijer, 2013). It seems that once the toxic components of spilled oil have dissipated and broken down to more benign residual products, mangrove stands may re-establish in similar ways to non-oil impacted forests (Duke, 2016).

Hydrologic restoration to bring in clean marine water daily gives a natural cleansing effect to an affected area (Jackson and Lewis, 2000). In this way, oil residues that otherwise may resurface and cause mortality of mangrove plants and seedlings are washed away over time. Recovery may proceed more rapidly in areas of high tidal ranges where the effects of more flushing appears to break down deposited oil faster (Burns et al., 1993).

In cases of an oil impact, ensuring that oil does not reach mangroves is the surest way to prevent problems, but if oil does reach the trees, the application of proper clean-up techniques has in some cases proven to be helpful in speeding up mangrove recovery (Jackson and Lewis, 2000). Such recovery and clean-up of oil in mangrove areas should best be restricted to cool-water jetting, since most forms of mechanical recovery and chemical usage are likely to cause further environmental damage (IUCN, 1993).

## **Mangrove rehabilitation after an oil spill**

### **Specific considerations:**

- Prevent oil from entering mangrove areas.
- Evaluate pros/cons of clean-up & recovery of oil (by cool-water jetting only).
- Avoid damage from clean-up activities and toxicity of dispersants (not to be used).
- Consider residual oil toxicity in sediment (best practices: prevent settling or leave sites to recover naturally).
- Mangrove areas impacted by oil spills often recover by themselves (5-30 years).
- Rehabilitation efforts may speed up the recovery process.
- Ensuring sufficient tidal exchange is critical to success.
- Only consider planting if there is insufficient natural recruitment of propagules.
- Planting of propagules/seedlings only after oil has weathered for 9-12 months.
- Key: ensure tidal exchange (flushing), allow natural recovery.

## 2.6 Mangrove restoration efforts in (semi-)arid regions

There have been several restoration projects of mangroves in arid and semi-arid regions of the world (Figure 16). In the United Arab Emirates (UAE), successful afforestation programs since the 1980s have increased the total area of mangroves by almost 4,000 ha (Almahasheer, 2018). Three decades of persistent planting of some 500,000 nursery-reared mangrove seedlings at Mubarratz Island near Abu Dhabi (UAE) resulted in 16.5 ha of successfully established mangrove vegetation along the island and its causeway, offering cost-effective protection of some 7 km of shoreline (Erftemeijer et al., 2020). Mangroves were successfully introduced in Kuwait into (natural) intertidal zones and along shorelines of man-made islands (using stock plants from Bahrain and the UAE) and are now contributing to enhancing coastal biodiversity and habitat complexity (AbdoEl-Nil, 2001; Bhat et al., 2004; Al-Nafisi et al., 2009). In Eritrea, mangrove afforestation has been conducted successfully to support the livelihoods of local communities (Sato et al., 2005; Lewis et al., 2006).



**Figure 16.** Planted mangroves in (semi-) arid regions of the Pilbara, NW Australia (left) and Arabian Gulf (right)

In Senegal, a large-scale (700 ha) restoration project was initiated after dam construction, changes in hydrology and low rainfall had caused the degradation and loss of mangroves in the Saloum River estuary (Diop et al., 1997). The loss of the mangroves, on which many local communities depended for their livelihood, resulted in social upheaval (Conchedda et al., 2011), which prompted the restoration project. The restoration program, which adopted a carbon credit approach, successfully recovered ecosystem services, although not without further social conflicts associated with restrictions imposed on allowable use of the mangroves under conditions of the carbon offset scheme (Cormier-Salem and Panfili, 2016).

Other examples of restoration projects in (semi-)arid areas include the re-establishment of tidal flows in a site previously exploited for salt production in South Australia (Dittmann et al., 2019), mangrove restoration efforts at Port Hedland in the Pilbara region of north-western Australia (Erftemeijer et al., 2018; Pilbara Ports Authority, 2020) and in the Gulf of California, where mangroves were replanted in a previously cleared forest (Toledo et al., 2001).

Most of the restoration projects in arid regions have in common that the survival of planted seedlings is generally much lower than in the humid tropics. Moreover, the recovery of mangrove ecosystem functioning generally takes much longer (10-30 years) owing to the slow growth of the mangroves in these areas due to extremely low rainfall (resulting in high



salinities) and nutrient-poor soils (Erftemeijer et al., 2018; 2020; 2021c; Adame et al., 2021). Fertilisation of mangrove seedlings with treated sewage sludge had a beneficial effect on their growth performance in the nursery in Abu Dhabi, UAE, which was attributed to the higher nutrient contents and very low salinity of the sludge (Erftemeijer et al., 2021b). Similarly, soil enhancers, such as cocopeat or peat moss can also sometimes be used (esp. in mangrove nurseries) to enhance plant performance in arid regions (Erftemeijer et al., in prep). Mangroves in arid and semi-arid regions are highly vulnerable to changes in hydrology and rainfall (Almahasheer, 2018; Adame et al., 2021). In some parts of the Middle East and Pakistan, grazing by camels and goats can rapidly defoliate (replanted) mangroves and as such camels and livestock are best kept out of such areas, e.g. through fencing (Fouda and Al-Muharrami, 1995; Nagi and Abubakr, 2013; Shahid, 2015).

Restoration should first and foremost be focused on restoring the natural hydrology and encouraging the regeneration of native and dominant species (Lee et al., 2019). While this is important for all mangrove forests, it is particularly crucial for arid mangroves, where a small increase in salinity can exceed the physiological tolerance of mangroves, even for species highly adapted to hypersalinity, such as *Avicennia* spp.. Successful mangrove restoration in the extreme climatic conditions of arid regions is usually not characterised by a one-off planting campaign, but by sustained efforts year-after-year (even up to two or three decades), allowing for learning along the way and adaptive management to develop the best approach for local site conditions (Erftemeijer et al., 2018).

### **Mangrove rehabilitation in (semi) arid areas**

#### **Specific considerations:**

- Despite the often extreme environmental conditions, mangrove restoration in (semi) arid regions is technically feasible.
- Mangroves in (semi) arid areas are characterised by slow (and stunted) growth due to extreme climatic conditions, very low rainfall and nutrient-poor soils.
- Consequently, it takes longer in arid regions to achieve successfully regenerated mangrove forest stands (typically 10 to 30 years).
- Restoration of mangroves in arid climates should especially focus on restoring the hydrology (where this has been altered).
- Seedlings should be planted at the appropriate position in the intertidal to ensure correct tidal inundation for the right species (especially critical in arid areas).
- Use of soil enhancer (e.g. cocopeat), treated sewage sludge or fertiliser may improve seedling performance (if proven beneficial in pilot testing).
- Where relevant, grazing camels and livestock should be kept out of restoration sites.
- Sustain restoration efforts year-after-year, allowing for learning along the way to develop the best approach that works under site-specific conditions.
- **Key: Ensure/restore appropriate tidal hydrology and soil conditions.**

## 2.7 Mangrove restoration in abandoned aquaculture pond areas

Most parts of South East Asia and Central America have seen large-scale conversion of mangroves to aquaculture ponds during the past three decades, mainly associated with the rapid increase in the intensive production of cultured marine shrimp during the 1980s and 1990s (Erftemeijer, 2002; Primavera, 2005). However, this mono-species aquaculture production in many regions has proved to be unsustainable, due to a variety of reasons including inappropriate construction and management, poor environmental conditions, overstocking, pollution and disease (Stevenson, 1997; Primavera, 2005). A large number of shrimp ponds consequently are unproductive and currently lie idle, while the loss of mangroves has resulted in major negative consequences for coastal protection (Figure 17; Primavera et al., 2011; Van Wesenbeeck et al., 2015). Accurate assessments of pond disuse are difficult to obtain; however, unofficial estimates have suggested that by the turn of the century as many as 70% of ponds throughout SE Asia were disused after having been in production for a period of three to five years (Stevenson, 1997; Primavera, 2005). Ponds are abandoned due to diseases (bacteria, viruses, and other pathogens) that cause mass mortalities and crop failures. More than 50% of the total 239,323 ha of shrimp ponds in the Philippines were already abandoned by the mid-1990s (Yap et al., 1995).



**Figure 17.** Abandoned shrimp ponds (left) and replanted mangroves in former shrimp ponds (right)

Pond construction, shrimp culture, and pond abandonment have led to a range of alterations to the physico-chemical and environmental conditions in these former mangrove areas, including changes in soil properties, in hydrological conditions and in the flora and fauna composition of the pond areas. Mangrove rehabilitation or restoration in these areas must consider the causes of production failure, the environmental conditions remaining following disuse, the needs and preferences of pond owners and coastal managers, and technical constraints (Stevenson, 1997; Stevenson et al., 1999; Lewis et al., 2002; Lewis, 2005; Primavera et al., 2011; Di Nitto et al., 2013; Oh et al., 2017; Van Bijsterveldt et al., 2020; Sidik et al., 2021). Activated acid sulphate soils often prove to be one of the major obstacles for the redevelopment of abandoned aquaculture ponds (Stevenson, 1997).

Not all abandoned aquaculture ponds are in former mangrove areas, but quite a few have also been established in areas that used to be occupied by other habitats, including salt flats, marshes and freshwater swamps, as well as terrestrial areas and rice-fields (that may have been mangrove before). Not all 'abandoned' shrimp or fish pond ponds are necessarily

disused, some may still be farmed or harvested despite low yields. As such, they are sometimes referred to as ‘underutilised’ (Primavera et al., 2011). While there is massive widespread potential for mangrove rehabilitation in abandoned aquaculture areas worldwide to restore vital ecosystem services (including climate change mitigation and adaptation), studies in the Philippines suggest that – while technically feasible – it requires strong political will and capacity shifts to cancel leaseholds to rehabilitate former mangrove areas (Primavera et al., 2014; Duncan et al., 2016).

Key to the restoration of mangroves in abandoned aquaculture pond areas is restoration of the tidal hydrology. This is most effectively achieved by breaching the dykes surrounding the ponds in strategic locations (Figure 18; Sidik et al., 2021). Where necessary, dendritic tidal creek systems may need to be (re-)excavated to restore the original tidal hydrology (Stevenson et al., 1999; Lewis, 2005b; Oh et al., 2017). In the case of abandoned aquaculture ponds, pond walls can be regarded as physical barriers to the inflow of propagules and subsequent establishment of mangrove seedlings (Krauss et al., 2008). While simply breaching dykes seems to be one solution, it is not always so easy. The location and size of breaches is important, as inadequate breaches limit tidal exchange and therefore natural recovery of the mangroves (Stevenson et al., 1999). Reintroduction of tidal flow can also help restore natural sediment dynamics in former pond areas, as demonstrated in a case study in Perancak, Bali (Indonesia), which recorded substantial sediment deposition and increasing bed levels in former pond areas (after dyke breaching) followed by rapid natural mangrove recruitment and seedling establishment of non-planted *Avicennia* spp. in areas that had been planted with *Rhizophora* sp. over a period of 14 years (Sidik et al., 2021).



**Figure 18.** Small dyke preventing drainage (left) and creation of a breach to restore normal tidal hydrology (right)

Excavation activities during pond construction and/or build-up of sludge during pond operation may have altered the surface elevation of the areas used for aquaculture. Therefore, rehabilitation of appropriate surface elevations may be another crucial first step in successful mangrove rehabilitation of disused ponds in some areas (Oh et al., 2017). Successful restoration also depends on adequate training programs, learning from mistakes, and the utilisation of the basic principles of adaptive management (Stevenson et al., 1999).

Assisted regeneration, that is, planting of nursery seedlings or wildings, may be required if natural recruitment is too slow to keep up with mangrove loss. Healthy mangrove areas generally have abundant recruits of propagules and seedlings with densities in the order of

tens- to hundreds of thousands per hectare (Primavera et al., 2011). When the hydrology has been restored through strategic dyke breaches or culverts, the tidal flow can bring numerous propagules back naturally into abandoned ponds (Di Nitto et al., 2013). Where the availability of propagules is low, e.g. due to a lack of healthy mangrove stands in the wider landscape nearby, or recruitment is limited due to some other reason (e.g. soil conditions), regeneration can be further assisted by bringing in propagules or planting seedlings into the abandoned ponds after the hydrology has been restored (Primavera et al., 2011; Xiong et al., 2021).

### **Mangrove rehabilitation in abandoned aquaculture pond areas**

#### **Specific considerations:**

- Not all abandoned aquaculture ponds are in former mangrove areas, or had been converted first to rice-fields by earlier generations.
- Not all 'abandoned' aquaculture ponds are necessarily disused, some may still be farmed or harvested despite low yields.
- Ongoing land tenure & ownership can be an obstacle to mangrove rehabilitation in (abandoned/disused) aquaculture pond areas.
- Restoration of tidal hydrology and overcoming recruitment bottlenecks are critical for successful mangrove restoration in abandoned aquaculture pond areas.
- Tidal flows and drainage should be re-established by strategic breaching of pond dykes and restoration of former channels and creeks.
- Rehabilitation of appropriate surface elevation may be required if this was significantly altered during pond construction (excavation) or pond operation (sludge accumulation).
- Where recruitment of mangrove propagules and seedlings is hampered or limited, regeneration can be assisted by bringing in propagules or planting of seedlings.
- **Key: restore tidal hydrology/surface elevation, supplement propagule supply, resolve land tenure**



## 2.8 Mangrove restoration in abandoned rice farms

In several parts of the world, there is a long history of rice cultivation in mangrove areas, especially in West Africa where some 200,000 ha of mangroves were cleared for rice production in Guinea Bissau, the Gambia, Guinea, Senegal and Sierra Leone over the past 100 years (Agyen-Sampong, 1991; Wolanski and Cassange, 2000). Examples in other regions include the Rufiji Delta in Tanzania (Japhet et al., 2019), Odisha State in India (Chauhan et al., 2017), Rhakine State in Myanmar (San, 2020) and the Mekong Delta in Vietnam (Veettil et al., 2019a). The use of mangrove areas for agricultural production is often driven by poverty and population pressure in coastal areas, as the farms typically have low yields due to the salinity and acid sulphate soil conditions, although some areas benefit from regular deposits of silt after annual flooding (Agyen-Sampong, 1991; Balde et al., 2014; San, 2020). Salinity intrusion can also be a significant problem in rice farms in mangrove areas, sometimes rendering them unsuitable (Wolanski and Cassange, 2000). The economic and environmental viability of conversion of mangrove forest to rice cropland has been drawn into question, owing to the relatively low yields of rice farms, much higher economic benefits from mangroves and significantly enhanced carbon equivalent emission by rice paddy ecosystems (Chauhan et al., 2017).

A recent USAID-funded ‘West Africa Biodiversity and Climate Change’ (WA BiCC) project in Bonthé, Sierra Leone, initiated the introduction of agri-silviculture (integrating mangrove planting as living fences along rice farms) in coastal areas where mangroves had been encroached for rice farming (though yielding low productivity) during years of famine and civil war. This integration of rice and mangrove planting on farms was piloted on 20 farms in six communities and is seen as a win-win solution to allow for their co-existence, ensuring synergy between nature conservation and sustainable agriculture (Tetra Tech, 2017).



**Figure 19.** Rice fields in mangroves in Guinea-Bissau (left) and breaching of dykes for mangrove restoration

In Guinea Bissau, large tracts of mangroves have been converted for the development of rice farms (dyked and drained) within the Cahheu National Park and its buffer zone (Figure 19). Many of these rice fields were abandoned after a few years as salt water intruded and soils acidified. Subsequent ‘ecological’ mangrove restoration in these areas focused on hydrological restoration by breaching dykes surrounding the rice farms, which improved soil conditions and facilitated natural regeneration of 200 ha of mangroves, the early results of which are promising (Figure 20; De Boer, 2017; Wetlands International, 2021). The restoration

of the mangroves was accompanied by the establishment of an 8,000 ha community-based protected forest in the buffer zone of Cacheu, along with the development of sustainable livelihood alternatives (Wetlands International, 2021).



**Figure 20.** Mangrove restoration in abandoned rice fields in Guinea Bissau (in progress)

A recent study in Tanzania demonstrated that even after 15 years, natural recovery of mangroves in abandoned rice farms (in former mangrove areas) in the Rufiji Delta was far from complete, with grass cover limiting the recovery due to obstruction of propagule dispersal (Japhet et al., 2019).

### **Mangrove rehabilitation in abandoned rice farms**

#### **Specific considerations:**

- Not all rice farms are in former mangrove areas or may be truly abandoned.
- Consider land tenure & ownership in (abandoned/disused) rice farming areas.
- Some abandoned rice farming areas may show good natural recovery of mangrove vegetation and may not need major intervention.
- Restoration of (tidal) hydrology is critical for successful mangrove restoration in abandoned rice farm areas (especially where there is lack of natural recovery).
- Tidal hydrology & flushing should be restored by strategic breaching of farm dykes.
- Hydrological restoration will also improve soil conditions, which can be especially important in acid sulphate soil areas.
- Ecological Mangrove Restoration (focusing on restoring environmental conditions) will facilitate natural recovery of the mangroves (and may not require any planting).
- **Key: restore hydrology & soil conditions (flushing), resolve land tenure/ownership.**

## 2.9 Mangrove-aquaculture solutions

Two types of solutions to integrate mangrove and aquaculture can be distinguished: aquaculture within the mangroves, and mangrove-aquaculture systems that are usually called silvo-fisheries or aqua-silviculture (Figure 21). The latter systems integrate mangroves in the aquaculture landscape, or allow aquaculture ponds in a mangrove landscape, and have been proposed for mangrove restoration or maintenance. The planting of mangrove trees on the bunds only is one of the popular silvo-fisheries systems, in particular in Indonesia, but it has no significance for coastal protection and marine biodiversity (Bosma et al. 2014), and will not be discussed below.



**Figure 21.** Integrated mangrove-silvofisheries (left) and mud crab culture in mangrove pen in the Philippines (right)

### Aquaculture and other livelihood options within the mangroves

In Guangxi Province of China, an eco-farming aquaculture system with underground tubes for fish, an above-ground culture of clams and worms was installed before mangroves were rehabilitated for ecotourism and public education (Fan et al., 2013). Field trials showed that net annual income reached 6,000 USD/ha only per year in high saline areas, and less than 17,000 USD/ha per year in low saline areas. This eco-farming aquaculture system is surge-resistant, easy-to-operate, and management-wise, it is cost-efficient. The products are high in quality and value. However, installation cost was higher than that of building an above-ground irrigation and drainage system: 30,000 compared to 5,000 USD/ha, respectively.

Elsewhere, farmers have been allowed to install (1) pens for mud-crab farming or (2) cages for culturing various species within the existing or new coastal mangroves forests. Both pens and cages, as well as the system in Guangxi, when scaled, disturb the mangrove ecosystem, and the respect of rules and regulations regarding mangroves is difficult to enhance due to inaccessibility. Apparently, these may perhaps be for livelihood options, but not for mangrove area expansion.

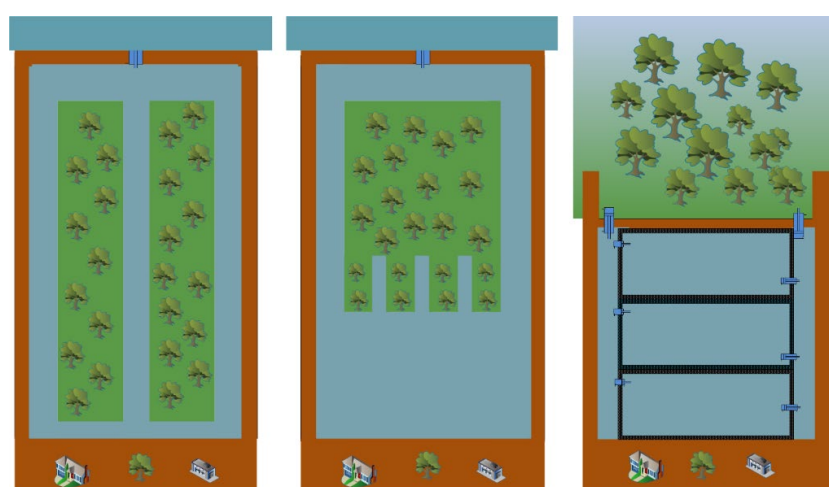
However, these livelihood options can be sustainable only if realised by cooperatives, or other participatory collective institutions, that make decisions transparent and enforce the regulations. Then, within the mangrove, one can harvest snails (*Cerithidea obtusa*) and oysters such as the Pacific oyster (*Crassostrea gigas*), both having a high market value. Thereto, the tide needs to remain active, although seasonally a net could prevent outflow of



the larger pieces, and the seeds can freely flow and benefit to the ecosystem. Such mangrove forests can also harbour honeybees and produce timber; some mangrove species produce high quality timber.

### Mangrove-aquaculture systems

Silvo-fisheries integrate mangrove tree culture and brackish-water aquaculture activities (Bacongus, 1991; Primavera & Agbayani, 1996; Fitzgerald, 2002; Bosma et al., 2014; Ahmed et al., 2018). Silvo-fisheries is a multiple use system that promotes a harmonious co-existence between fishery species and mangrove tree species in a landscape integrated system that encompasses both aquaculture practices to provide regular income and mangrove preservation for timber production mainly (Bacongus, 1991; Bush et al., 2010; Bosma et al., 2014). Among the present types of mangrove-aquaculture systems, all, but one, have little or no function for coastal protection, and have only one or two ecosystem functions (left and centre panel in Figure 22 below). In fact, they have been designed to produce timber and do not allow optimal shrimp production (Bosma et al. 2014). These silvo-fishery systems have been extensively studied in Indonesia, Thailand and Vietnam. Most studies converge around an optimal mangrove-pond ratio of 60% within the farm-dykes to reach optimal forest cover or economic goals for shrimp farmers (Truong and Do, 2018). Without intensification of the aquaculture component, the farm area needs to be more than 6 ha to provide a fair livelihood (Bosma et al., 2014).



**Figure 22.** The two Vietnamese silvo-fisheries systems, integrated (left panel) and mixed (middle panel) designed mainly to produce timber, and an associated mangrove-aquaculture system (right panel) in which the mangrove is not enclosed within bunds and hence can express most of its ecosystem functions (Bosma et al. 2014)

The negative impact of the mangroves on the aquaculture component can be reduced by regular trimming of the trees along the water, or by using species other than the timber-oriented *Rhizophora apiculata*, which needs only few inundation days. To optimise these systems, others have studied the effect of the leaves of various mangrove species on the performance of shrimp (Tran and Yakupitiyage, 2005; Alam et al., 2021).

Vietnam farmers have recognised that mangrove forests near their pond can reduce risk of diseases, but most of them choose for the short-term benefit of intensive systems for many



reasons (Bosma et al., 2019). However, most policy makers do not know that the annual total economic value of a mangrove ecosystem can be higher than that from intensive shrimp culture (Bosma et al., 2014). Moreover, the high contribution from mangrove to the national economy is sustainable, while that of intensive shrimp culture is mostly short-lived (Rivera-Ferre, 2009), because over time, shrimp ponds converted from mangrove forests have had negative environmental impacts (Primavera 1991), and increased social inequality in the local areas (Adger 1999; Joffre et al., 2016). To avoid those negative impacts and reach most ecosystem functions and their economic value, the mangroves in a silvo-fishery system need to be connected directly to the open waters of the estuary and not enclosed between pond-bunds (see example discussed below).



**Figure 23.** Climate-smart associated mangrove-aquaculture system along the coast of a Philippine island

### **Sustainable silvo-fisheries: associated mangrove-aquaculture**

Only in the Philippines, Colombia and a pilot area in Indonesia, climate-smart mangrove-aquaculture systems have been studied, practiced or tested. In climate-smart and ecologically sound systems, a wide area of mangroves is maintained between the ponds and the water intake point (Figure 23). In the Philippines these systems evolved only after its government forbade farmers in cutting mangroves along the sea in 1982 (Primavera, 1991). In Colombia, well-thinking entrepreneurs maintain a mangrove belt to get a better quality of intake water (Gautier et al., 2001). In an Indonesian project, among the 377 farmers who had previously been trained in aquaculture field-school, about 90 farm households realised a mangrove component in their ponds) in 2019 (Yuniati et al. 2021; Rejeki et al. 2020).

In the Indonesian pilot area, most farmers still practice traditional milkfish-(shrimp) aquaculture because of inappropriate or no training at all combined with poor water quality, which prevent them from intensifying their shrimp production. However, after going through a proper season-long training on low-external input system (LEISA) in farmer-field-schools, the farmers tripled their shrimp yields and doubled their income from one of their ponds. Through these results, the project recovered the overall cost of the training within a year (Widowati et al., 2021). Another analysis of the project data showed also that smaller ponds gave higher yields ( $r=-0.2$ ). This evidence counter-argued against farmers' thinking that their

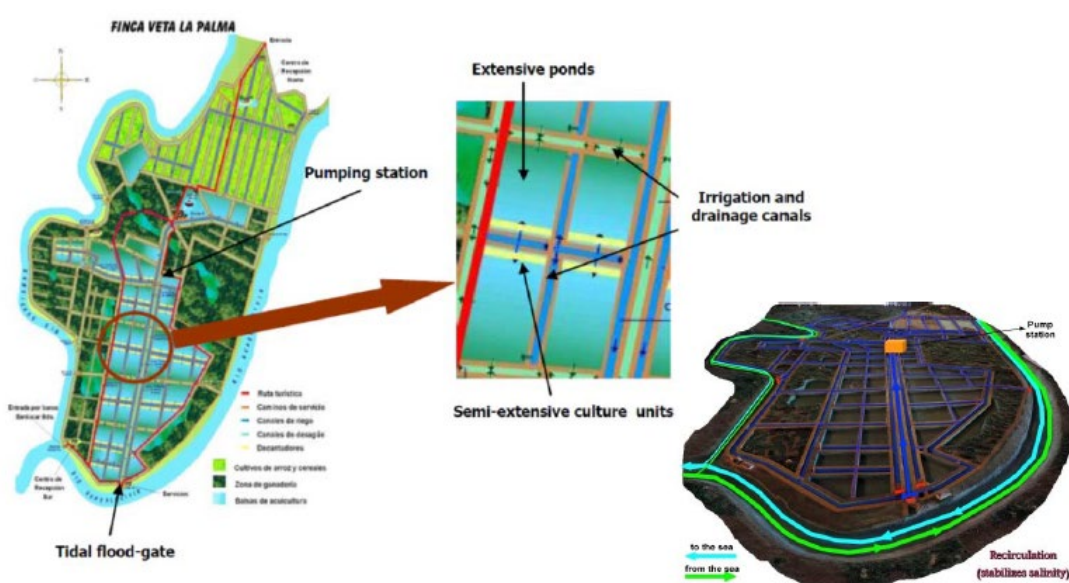
income would suffer when they convert a piece of their pond into mangroves. In the first year after making the mangrove-aquaculture ponds, although the farmers did not apply LEISA yet, and part of the stock was washed away because of the floods during that period, they still caught plenty of seafood species in the trap placed in the pond gates; the increased catch gave them a good income (Yuniati et al., 2021).

### **Mangrove : aquaculture ratio in associated mangrove-aquaculture**

The ideal mangrove-to-pond aerial ratio for optimal water (quality) management is still open to debate. Recommended figures have varied from a mangrove-to-pond ratio of 20:1 (Primavera, 2007) for cleaning all heavy metals, to 4:1 for maintaining ecosystem health (Saenger et al., 1986; Fitzgerald 2002; Barbier et al., 2008). Observed figures in the field are close to 1:1 (Tendencia et al., 2012) or even lower (Primavera et al., 2007).

Higher mangrove-to-pond ratios are needed to process nitrogen (2:1 to 5:1) and phosphate (6:1 to 9:1) when the mangroves are enclosed (Shimoda et al, 2005 and 2007). Along coasts, a relatively lower ratio ( $<0.2:1$ ) would be enough to remove dissolved nitrogen (Rivera-Monroy et al. 1999), phosphate and ammonia (Jitthaison et al. 2012). In the latter case, most of the N and P may be assumed to be used in the marine ecosystem; thus, N and P do not accumulate in the soil or the water. Thus, for mangroves that are not enclosed, along the waterways in an estuary, a ratio mangrove to pond of 1:5 would be enough for optimal water quality.

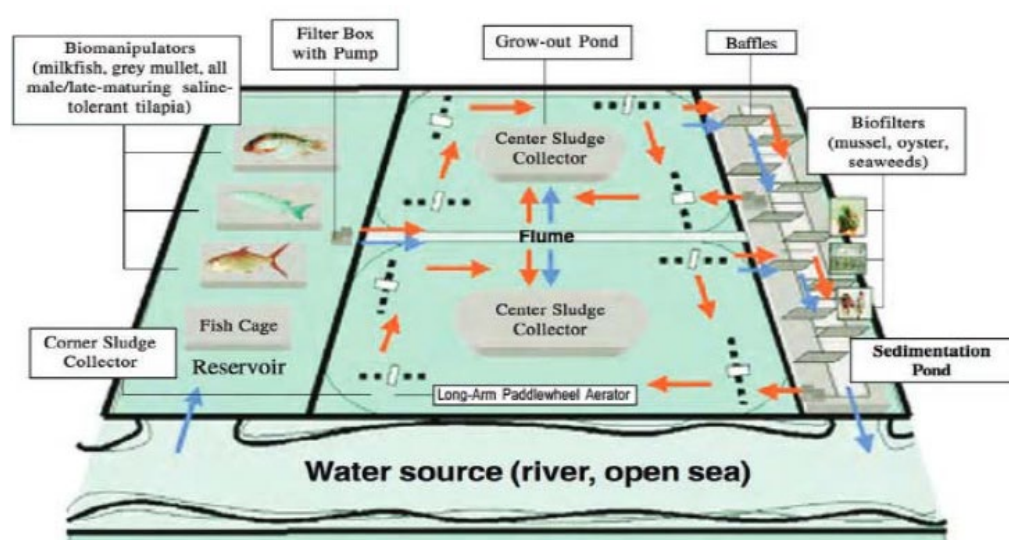
In any of these approaches, whether implemented at farm level or at landscape scale, the recommended ratio of mangrove to pond area should only be seen as a rough guide to aim for, as this will highly depend on the quality, age, density, height and health of the mangroves, local site conditions, as well as on the type and intensity of aquaculture that is being pursued. Strict enforcement of specific mangrove-to-water ratios as restrictive laws on fish- or shrimp farmers is unlikely to be the most effective approach in achieving sustainable aquaculture and mangrove preservation (Baumgartner et al., 2016).



**Figure 24.** Sustainable aquaculture farm (Finca in Veta La Palma, Spain) using tide, pumps, and carefully separated flows of intake water and effluents that are integrated in the landscape (Tonneijck et al., 2016; Fernández-Rodríguez et al., 2018).

### Landscape design with associated mangrove-aquaculture.

Sustainable shrimp aquaculture, with low disease risk and high productivity, does not just emerge after establishing mangrove greenbelts along the waterways. The streams of inlet and outlet water also need to be managed appropriately. The first step of good water management is having both an inlet and an outlet gate, but if the outlet water is then used by a neighbour, the potentially contaminated effluents carrying disease agents will infect these ponds. Thus, the inlet and outlet streams of small farms need to be separated at landscape level, as do well-managed large farms (see Figures 24 and 25).



Source: from Ballao and Tookwinas (2002).

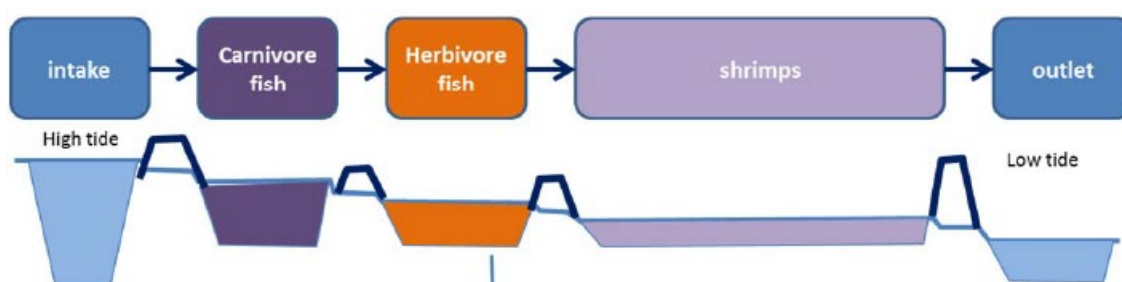
**Figure 25.** Example of cascade of low discharge re-circulating shrimp farm lay-out, within the inlet water storage caged milkfish, grey mullet and tilapia, and in the settling reservoir with mussels, oysters and seaweed (Troell, 2009).

Larger farms also set up ponds for filtration and sedimentation in their system, and a system needing little intake water by recycling the water in ponds with seaweed, mussel and oysters. To recycle the water, on at least one of its shrimp farms in Thailand, Charoen Pokphand Food (CP), known throughout south-east Asia as animal feed producer, reduced their shrimp pond area by half and stocked the other areas with tilapia and seaweed (Figure 26). The farm's total shrimp production remained the same, but cost like medicines diminished; thus benefits increased. The systems, as used and described here, confirm ecological principles on reducing disease pressure (Sorgeloos and De-Schryver, 2020).



**Figure 26.** An intensive shrimp production farm in Thailand with some of its pond area stocked with seaweed and Tilapia to aid in recycling its wastewater (Courtesy: Patrick Sorgeloos).

Partly aligned with above, and following the green-water principle applied by most shrimp farmers in the Philippines (SEAFDEC/AQD, 2011), Tonneijck et al. (2016) proposed a cascade aquaculture system for the Mekong delta (Figure 27). In the proposed three-step cascade aquaculture system, the first pond is stocked with carnivore fish (e.g. seabass) to eat potential disease carriers. In the second, the herbivore tilapia is stocked to improve water quality before the water is diverted to the shrimp ponds. Milkfish (*Chanos chanos*) was shown not to be an efficient bio-manipulator for shrimp water; species like mullet (*Mugil cephalus*) have not been studied. Moreover, to respect environmental laws or to protect other farmers, the effluents might need to be treated before these can be disposed of in a waterway. Thus, to function optimally, each farm would need at least two types of complementary ponds: treatment reservoirs, also for water storage, and settling ponds. In the settling ponds clams (e.g. blood-cockle, *Anadara granosa*, and *Meretrix lyraya*) and seaweed (e.g. *Gracillaria verrucosa*) can be cultured for the market. The pond with carnivores become superfluous, if farmers are allowed and can afford to use a specific pesticide (e.g. teaseed) and the farms use a high quality filter for the inlet water (SEAFDEC/AQD, 2011).



**Figure 27.** Top-view and crosscut of a three-step cascade aquaculture system with first a pond with carnivore fish eating potential disease carriers, then herbivore tilapia to improve water quality and finally shrimp ponds (reproduced from Tonneijck et al. 2016).

The area (volume) of the first two upstream ponds depends on the need of water for the shrimp ponds. SEAFDEC/AQD (2011) advises to use 25% for the reservoir of the area that is stocked with shrimp. The company in Thailand stocked 50% of its ponds with seaweed plus



tilapia to recycle the water. In such a landscape, individual farmers can decide for super-intensification as proposed by Nguyen et al. (2019) to keep shrimp culture sustainable. When using an outdoor Recirculation Aquaculture System (RAS) this will decrease water-use and thus the volume of the reservoirs. As the former needs aerators and plastic or concrete lining, others propose more accessible aquaculture systems using the function of the pond soil and feed stimulating within-pond recycling of the fish faeces (Joffre & Verdegem, 2019).

### **Mangrove-Aquaculture (silvo-fisheries) solutions**

Special considerations:

- Aquaculture within mangroves disturbs the ecosystem, illegal practices will be hard to control, and does not contribute to the expansion of the mangroves.
- Silvo-fisheries with impounded mangroves, or mangroves on the dykes, do not contribute to coastal protection or marine biodiversity.
- Along shorelines within the estuary, mangrove rehabilitation is easier, than in front of the coastline.
- A fringe of 20-50m of mangroves along the waterways where aquaculture farms get their water and dispose their waste water, protects the dykes, cleans the water, and provides space for water storage to reduce flood risks.
- Sustainable aquaculture needs separated canals for inlet and outlet water, and a cascade of ponds to reduce risks of disease outbreaks and environmental pollution.
- To reduce water-use and the risk of diseases, shrimp farms can recycle water in ponds with seaweed and tilapia.
- The drain for the waste water can have settling barriers below water level, and finally be open to the tide to provide a good water flow for clams.

Tools: Space for mangroves to regrow along waterways, behind which shrimp etc. can be farmed sustainably in a cascade of ponds with a.o. tilapia and seaweeds.

Challenges: Participatory re-parcelling of land, and compensation of drop-outs.

Key: Cascades of ponds reduce risk and cost, while maintaining total shrimp production.

## 2.10 Mangrove rehabilitation of pipeline crossings and access corridors

Where shoreline crossings of oil or gas pipelines through mangrove habitat are deemed unavoidable and necessary for the transportation of oil or gas from an offshore field to an onshore processing plant, clear-felling of a strip of mangrove vegetation (corridor) along the intended route is generally required. Widths of such corridors through mangroves for pipeline installation vary but usually are in the order of 50-75 m to allow access for the equipment needed for trenching and pipeline installation, while the trench itself is usually much narrower in width (depending on the diameter and number of pipelines involved). Removal of the mangrove vegetation in the corridor and the trenching itself may create sediment instability and erosion, especially in areas with strong (tidal) current flow velocities and/or other hydrodynamic (and/or wave) exposure. Backfilling of trenches with rock material is usually done for reasons of stabilisation and protection of the pipeline, but application of a top layer of sediment (preferably the original material where possible) back to the original substrate level (as prior to trenching) is recommended, to facilitate recovery of the mangrove vegetation following completion of the pipeline installation and trench backfilling works (Figure 28; Erftemeijer, 2013).



**Figure 28.** Mangrove reinstatement in pipeline corridor (Darwin Harbour, NT, Australia) (Images: Josh Corbett)

If sediment instability is an issue of potential concern (e.g. at sites with high flow velocities and wave exposure or in macrotidal areas with strong tidal currents), there are various (temporary) sediment stabilisation techniques available, including geotextile matting, bamboo fences, ground consolidators ('geohooks') or similar structures. Where feasible, this may be combined with mangrove planting of the open areas (corridor and backfilled trench) to speed up regeneration of the mangroves to provide natural (re-) stabilisation of the sediment (Paling and Erftemeijer, 2013). In Darwin Harbour, chicken wire fences placed perpendicular to the tidal flow within the corridor helped to trap mangrove propagules, which facilitated the recovery process.

### **Rehabilitation following decommissioning of (other) infrastructure**

The general principles discussed above for reinstatement of mangrove vegetation following pipeline installation can also be applied following the removal/decommissioning of infrastructure from mangrove areas (e.g. pipelines, temporary access roads, conveyers). The most important aspect for the rehabilitation of such areas is to first return the site to its original topographic state and hydrological regime, allowing for natural regeneration,

implementing planting where necessary (or to speed up the recovery process) and carry out appropriate monitoring (Paling and Erftemeijer, 2013).

### **Rehabilitation of pipeline crossings and access corridors**

#### **Specific considerations:**

- Removal of mangrove vegetation in access corridors and pipeline trenches may create sediment instability and erosion. Therefore, it is prudent to minimise clearing and trenching as much as possible. Management of the potential erosion may also be warranted.
- Backfilling of trenches (preferably with a top layer of original sediment material from the area) back to the original (pre-trenching) substrate level is essential for recovery.
- Sediment stabilisation techniques (e.g. geotextile matting, bamboo fences, ground consolidators) may need to be considered in areas with a high risk of sediment instability.
- Mangrove areas impacted by pipeline crossings and access corridors will eventually recover by themselves (15-30 years).
- Rehabilitation efforts may speed up the recovery process.
- Consider trapping of propagules and planting of mangrove seedlings in the open areas (corridor and backfilled trench) to speed up the rehabilitation.
- **Key: backfill stabilisation and propagule trapping.**

## 2.11 Mangrove planting on dredged material

The last few decades have seen a rise in beneficial use application of dredged material for the restoration and/or creation of marine and coastal wetlands worldwide, including mangroves and salt marshes (Figure 29). Particularly in the USA (largely through the work of the US Army Corps of Engineers) there is a considerable body of well-documented experience with the establishment of such ‘dredged material wetlands’.

In Florida, >250 spoil islands of 1 to 2 ha in size from historic dredging operations (1930-1970) were naturally colonised by mangroves, saltmarsh and terrestrial vegetation, following processes of natural succession (Lewis and Lewis, 1978). At Mubarratz Island near Abu Dhabi (UAE), ~17 ha of *Avicennia marina* mangroves were successfully planted on dredged material over three decades using 500,000 nursery-reared seedlings in an area where previously no mangroves existed (Erftemeijer et al., 2020). In Singapore, two new mangrove areas (encompassing a total area of 13 ha) were created on dredged material to compensate for the loss of mangrove habitat due to the construction of an offshore landfill site at Pulau Semakau (Chuan, 2009). The site was protected by a rock perimeter bund, underlain with geofabrics, and engineered to correct tidal positioning, with an excavated system of artificial tidal creeks. Initial mangrove test plantings all died as the fresh mud was still too acidic, but after a delay of four months to let the seawater neutralise the soil pH, the two areas were successfully planted with 400,000 mangrove seedlings (Chuan, 2009). There are similar sites at Lake Maracaibo in Venezuela (Pannier and De Pannier, 2008) and at Navachista Bay and Chiapas lagoon in Mexico (Chargoy and Hernández, 2002), all involving dredged material deposits that were either actively planted with mangroves to help stabilise the deposits or colonised naturally over time.



**Figure 29.** Mangroves on dredged material in the UAE (left) and a dredged spoil island, Alafia Bank, USA (right)

Case studies, research results and accumulated experience worldwide provide a considerable amount of information about options for beneficial use of dredged material for coastal (mangrove) wetland restoration and establishment. Key aspects to be considered during the planning and design of beneficial reuse of dredged material for creating coastal habitats, derived from a recent review (Erftemeijer, 2019) of 39 case studies in 11 countries, are summarised below. Although many of these examples have been successful, they were not without challenges. In exposed areas, there was often a need for (temporary) protective hard engineering structures to ensure stability of fine dredged material deposits. Achieving the appropriate tidal elevation at a site to create hydrological conditions conducive for



mangroves proved complex and involved patience, owing to processes of consolidation and subsidence of the dredged material following dewatering. Colonisation by vegetation, whether spontaneous or assisted, and birdlife typically took several more years. Consequently, the timescales involved were not always matching the initial expectation of developers or the general public.

### **Mangrove planting on dredged material**

#### **Specific considerations:**

- Contaminated sediments should be avoided; use of fines such as silt or clay generally requires the construction of confining dykes.
- Consider the required volume and compaction of dredged material as a function of size, depth and desired tidal elevation of the wetland area.
- Moving dredged material more than a few kilometres from a dredging site is seldom feasible.
- Avoid areas of high wave energy and ensure shelter from strong swell, wave action, extreme tidal flows and storms.
- Avoid major adverse effects on water flow, sediment transport or navigational safety.
- Consider existing environmental values of the site selected for dredged material wetland development (avoid a net loss of environmental benefits).
- Dredged material wetlands located far from existing sources of plant recruitment may need to be planted; planting may speed up development of vegetation cover.
- Where possible, create new wetlands in ways that mimics natural landscape features.
- Consider morphological (e.g. slope and shape of intertidal mudflat), hydrodynamic and sediment-dynamic aspects in the design.
- Consider availability of dredging equipment and site accessibility when planning the works.
- Construction of special features, such as tidal creeks and tidal pools or structural complexity at a site will increase costs and may require innovative approaches.
- **Key: ensure appropriate bed level (tidal inundation) and stability/protection.**

## 2.12 Nature-based (hybrid engineering) solutions

Climate change and continued population growth are accelerating the need for diverse solutions to coastal protection. Nature-based solutions are particularly relevant in coastal areas where urban or agricultural landscapes and natural environments meet, as a response to climate change, population growth, and urbanisation (Pedersen Zari et al., 2019). Traditionally shorelines are armoured with artificial, non-adaptive structures, which come with significant economic, environmental and social costs. While hard structures will continue to have a place in coastal protection, alternative methods that are more sustainable and climate-resilient should be more broadly adopted into the future where appropriate. Nature-based methods (through “soft” or “hybrid” techniques) have the potential to play important roles in climate adaptation and mitigation because of their ability to reduce the threats of coastal erosion and flooding and provide co-benefits such as carbon sequestration (Morris et al., 2021).



**Figure 30.** Examples of hybrid engineering: mangrove-breakwater combinations in Australia (left) and Philippines (right)

Nature-based methods use the creation or restoration of coastal habitats for hazard risk reduction. This can be done through restoring the habitat alone (“soft” approach), or in combination with hard structures that support habitat establishment (“hybrid” approaches). Recent years have seen increasing initiatives worldwide to develop, test and apply a range of nature-based hybrid engineering techniques to mitigate the impacts of coastal hazards (Figure 30).

Depending on the type of shoreline (hard, sandy, muddy) and prevailing local conditions, nature-based hybrid solutions for coastal protection may include coastal sand dune restoration, ‘sand engine’ mega-nourishments for beach replenishment, combining seagrass restoration and artificial reef construction, and combinations of mangroves and breakwaters, or salt marshes and oyster reefs (Morris et al., 2021; Sreeranga et al., 2021). Shellfish reef restoration by placing oyster shells in coir bags or steel cages has been used successfully in several countries worldwide as breakwater alternative to attenuate wave energy and protect salt marsh or seagrass habitat in the lee of the reef (e.g. Borsje et al., 2011; Scyphers et al., 2011; Sharma et al., 2016; Morris et al., 2019) (Figure 31).

In the application of shellfish reefs for shoreline protection, potential disconnects between ecological and engineering functions of reefs can lead to uncertainty about their effectiveness in affording protection, thus limiting their broader application. Integrating both engineering and ecological principles, multidisciplinary testing of oyster reef applications to address knowledge gaps can lead to their more effective application (Morris et al., 2019).



**Figure 31.** Examples of hybrid engineering: oyster reefs and saltmarshes in Australia (left) and oyster reefs and seagrass on tidal flats in the Netherlands (right)

Over-engineering should be avoided (Mitch, 2014; Morris et al., 2019) and the use of heavy equipment be kept to a minimum to prevent excessive costs (Van Eekelen and Bouw, 2020). To optimise costs per unit area (ha) in order to achieve cost-effective results at larger scales, overly ‘cosmetic’ gardening approaches or ‘artificial’ landscape designs are best avoided. Also, shoreline armouring with artificial structures such as seawalls can have negative impacts on mangroves and their ecosystem functions (Heatherington and Bishop, 2012).

### **Mud nourishment**

Mud nourishment or the purposeful placement of thin layers of sediment (e.g. dredged material) in an environmentally acceptable manner to achieve a target elevation or thickness, has been applied in efforts to support coastal infrastructure and/or create, maintain, enhance, or restore ecological function (Ray, 2007). There are many documented case studies of successful applications of thin-layer placement of dredged material in salt marshes (Shafer, 2002; Ray, 2007; Baptist et al., 2019), but examples of such applications in mangroves are very few to date. Mangroves may also benefit from purposeful application of dredged material to keep up with sea level rise (esp. in areas that are also undergoing subsidence) or to aid certain mangrove restoration projects (Vogt, 2010; Mitra, 2013; Li et al., 2015).

### **Offshore windfarms**

A new, emerging development in Vietnam is the construction of offshore windfarms. The first to be constructed in the Mekong Delta was a 540 Ha (99.2 MW) nearshore windfarm in Bac Lieu Province, which started operations in 2016 (Anonymous, 2021). There are plans for several more of such windfarms in coastal Vietnam (as described in PDPD planning documents). There are concerns that these windfarms may alter the process of fine sediment deposition in the nearshore coastal areas of the Mekong Delta (Klaus Schultz (GiZ) and others,

pers. comm., December 2021). Windfarms situated at 7-10 km from the shore in the UK were found to have negligible impacts on coastal (sediment transport) processes (Cooper and Beiboer, 2002). However, other studies of European windfarms suggest that this depends on the layout of the farms, their distance from shore and the timeframe of the investigations (Esteban et al., 2011). Clever planning and engineering design of windfarms in the Mekong Delta region (supported by systematic model evaluations of various master designs and alternative locations) might under certain conditions potentially result in desirable impacts on sediment transport processes in the coastal region and offer co-benefits for coastal protection and aquaculture opportunities (Buck et al., 2004).

### **Nature-based (hybrid engineering) solutions**

#### **Specific considerations:**

- Nature-based approaches incorporate the creation or restoration of coastal habitats in the design of sustainable solutions for coastal hazard risk reduction.
- This can be achieved through restoring the habitat alone (“soft” approach), or in combination with hard structures that support habitat establishment (“hybrid” approaches).
- Successful application of nature-based solutions requires a multi-disciplinary approach and integration of ecological and engineering principles.
- Nature-based hybrid solutions for coastal protection along tropical muddy coastlines may include combinations of mangroves & breakwaters or mangroves & oyster reefs.
- Over-engineering, excessive use of heavy equipment, overly ‘cosmetic’ gardening approaches and ‘artificial’ landscape designs are best avoided.
- Mangroves may benefit from mud nourishment (purposeful application of dredged material or thin-layer placement) but there are very few examples of such applications to learn from.



## 2.13 Blue carbon mangrove plantations

Mangrove forests capture and store exceptionally large amounts of carbon and are increasingly recognised as an important ecosystem for carbon sequestration (Mcleod et al., 2011). The widespread loss of mangroves over the past 50 years has resulted in the loss of significant amounts of Carbon stored in the vegetation, litter and soil, thereby contributing to the causes of climate change. Consequently, there are increasing efforts underway to protect and restore mangroves worldwide in order to preserve this critical 'blue carbon', i.e. carbon stored in marine and coastal habitats (Ahmed et al., 2018; Kairo et al., 2018; Chowdhury et al., 2019; Elwin et al., 2019).



**Figure 32.** Young mangrove plantations in a former shrimp pond (left) and on an intertidal mudflat (right)

While conversion of mangroves into shrimp farms is assumed to result in significant losses of ecosystem carbon, a recent study in Thailand suggests that some abandoned ponds contain deep mangrove soils (>2.5 m) still intact with large carbon reservoirs that can exceed 865 t carbon per hectare (Elwin et al., 2019). The study also found that mangrove carbon pools can rebuild in abandoned ponds over time following mangrove recovery through active hydrological restoration (Elwin et al., 2019). The adoption of integrated (organic) mangrove-shrimp cultivation, with mangrove vegetation covering >50% of the total farm area, may also hold potential for preserving a significant proportion of the carbon stocks held in mangrove areas (Ahmed et al., 2018).

Unless new intertidal habitat is created, for example by using dredged material (see Section 2.11 earlier in this Chapter), the main focus of increasing the total mangrove area for blue carbon storage is through the rehabilitation and restoration of mangrove vegetation in areas where it was lost, such as in abandoned aquaculture ponds or rice farms (Figure 32), along eroding shorelines or other areas where the original mangrove habitat was degraded or lost.

Some recent studies suggest that anthropogenically planted and naturally regenerated mangroves eventually appear to store similar levels of C (generally within 7 – 17 years, occasionally up to 35 years), but monospecific plantations may store less than multi-species vegetations (Nam et al., 2016; Richards et al., 2020; Bukoski, 2021).

A recent review of emerging global case studies of successful blue carbon projects demonstrated the importance of involving members of the local community in all stages of

planning and implementation and incorporating livelihood aspects as part of the restoration project design (Wylie et al., 2016).

### **Blue Carbon mangrove plantations**

#### **Specific considerations:**

- Mangrove forests capture and store exceptionally large amounts of carbon.
- Conversion of mangrove habitat results in significant losses of carbon.
- Some abandoned aquaculture ponds in former mangrove areas may still contain original mangrove soils (>2.5 m deep) with considerable carbon reservoirs.
- Silvo-fisheries approaches (with >50% of the vegetation kept intact), hold potential for preserving carbon stocks held in mangrove areas.
- The carbon storage capacity of mangroves (and their carbon pools) can rebuild over time following mangrove restoration.
- Anthropogenically planted and naturally regenerated mangroves eventually appear to store similar levels of carbon.
- Monospecific mangrove plantations may store less carbon than undisturbed natural (mixed-species) mangrove ecosystems.
- Involve local communities in all stages of planning and implementation.
- Incorporate livelihood aspects as part of the restoration project design.

### 3 LESSONS LEARNT FROM THE CASE STUDIES AND THEIR POTENTIAL APPLICATION IN THE MEKONG DELTA

This Chapter aims to draw lessons learnt from the various Case Studies described in the previous Chapter, with a particular focus to their relevance for application for increasing coastal resilience in the Mekong Delta region in Vietnam.

#### 3.1 Understanding of the mangrove system

Before embarking on any mangrove restoration or rehabilitation effort, it is critical to first understand the mangrove ecosystem within the settings of the broader (abiotic) site conditions. So, it is important to first know the particular mangrove species that occur at the site, and to understand their ecology. In particular, one should understand what are the main environmental drivers determining their survival and growth, their patterns of reproduction, propagule distribution, and conditions for successful seedling establishment, at the site.

##### Physical site conditions

The ability of mangroves to successfully establish themselves at a site, and their subsequent health, growth and ability to survive there, depends first and foremost on the physical conditions at the site:

- **Overall site description**, including water depths, bed slopes and topography (gradients, drainage channels), extent of mangrove forest, proximity of rivers or open coast, land use of the hinterland, etc.
- **Hydrodynamic conditions**, including tidal ranges, tidal velocities, wave conditions, salinities, residual currents, flushing and circulation, etc.
- **Sediment supply**, including oceanic supply (governed by nearshore hydrodynamic processes that drive coastal sediment dynamics – as described in more detail in Section 4.1.2 for the Mekong Delta), riverine supply (governed by rainfall in the catchment that causes seasonal floods, and affected by land management practices, soil erosion and upstream hydropower dams), as well as sediment transport capacity and paths;
- **Sediment stability**, which is determined by the degree of shelter/exposure to wave energy and can be influenced by the presence of existing mangrove vegetation or other nearby ecosystems such as seagrass beds, shellfish reefs or coral reefs, or by man-made structures such as breakwaters, dykes, groynes, bunds, walls, levees or artificial reefs;
- **Hydrology**, including issues affecting availability of fresh water and water quality, as well as the tidal hydrology which determines the duration and frequency that a site is inundated by the tide (hydro-period), which may or may not exceed the tolerance of individual mangrove species (Van Loon et al., 2016). Tidal flushing also influences sediment quality (biogeochemical processes, nutrient availability, pore water salinity, dilution of contaminants) and water quality.

##### Propagule dispersal and seedling establishment

The process of natural recovery of mangroves depends first and foremost on recruitment, the production of propagules by the mangrove trees and the subsequent dispersal of these

propagules into the surrounding environment (with the tidal flows), as well as their chances of successful establishment, which is determined by site conditions. Where recruitment is limited or constraint, for example through a lack of source populations or an inability of propagules to reach (through obstacles or lack of hydrodynamic connectivity), human intervention such as removing obstacles or active ‘seeding’ or planting may be required. Planting on bare intertidal mudflats rarely leads to successful results (Erftemeijer and Lewis, 2000; Balke et al., 2011) and where it does (e.g. in areas that are rapidly accreting) there is usually no need for planting as natural recovery soon outpaces any human efforts, unless there is recruitment limitation by a lack of natural propagule supply. Establishment success differs between species, as each mangrove species is adapted to a particular set of environmental conditions (primarily related to their tolerance to inundation and salinity. Some biological factors can also interfere with successful establishment, such as propagule predation (esp. crabs), excessive bioturbation (causing uprooting of seedlings), barnacle infestation and grazing of newly established seedlings.

### **Factors determining mangrove ecosystem resilience**

The resilience of a mangrove stand depends on its size/extent (amount of intact habitat), species diversity (important for a range of biological processes and ecosystem functions, including pollination and succession), genetic diversity (determining the ability to adapt to change) and connectivity to other nearby mangrove areas (facilitating exchange of energy and supporting species, and supplying new recruits to enable recovery after major disturbance events) (McLeod and Salm, 2006; Alongi, 2008). In the face of climate change, the ability of mangroves to cope with and adapt to rising sea levels is also dependent on the availability of space for them to retreat further inland, which can be impeded by the presence of adjacent infrastructure and/or agricultural land (‘coastal squeeze’). Access to a sufficient sediment supply is another critical factor determining the ability of mangroves to adapt to sea level rise.

### **Role of mangroves in coastal protection and resilience**

Mangroves are widely acknowledged for their role in coastal protection and resilience from the effects of climate change (Gedan et al., 2011; Dasgupta et al., 2019; Costanza et al., 2021). Mangroves have been shown to significantly reduce the impact from wind and swell waves (McIvor et al., 2012a) and storm surges (McIvor et al., 2012b). Studies have documented a gradual raising of the soil surface elevation in mangroves as an adaptation to rising sea levels (McIvor et al., 2013). Mangroves have also shown to offer significant protection from the effects of cyclones/typhoons/hurricanes and tsunami waves (Kraus and Osland, 2020; see also Chapter 2.3), and can prevent coastal erosion during hurricanes (Pennings et al., 2021). The extent to which mangroves offer such protective services depends on a variety of factors, including stem density & thickness (determining hydraulic resistance), biomass, height/age, species composition, spatial heterogeneity of the vegetation, and the width of the mangrove zone (‘greenbelt’).

### **Determining the width of mangrove greenbelts for coastal protection**

There is no golden rule for determining the best width of a protective ‘greenbelt’ of mangroves along tropical shorelines (Primavera, 2005). Simply put: the wider the belt the



lower the risk of destructive impacts from a severe storm event. The effectiveness of coastal protection by a belt of mangroves as 'green infrastructure' along a shoreline increases exponentially with the width of that greenbelt (Koch et al., 2009). Larger mangrove rehabilitation sites in the mid- to upper-intertidal zone are expected to deliver much greater wave attenuation and other climate change mitigation and adaptation benefits than narrow, low-intertidal rehabilitated mangrove fringes (Duncan et al., 2016).

The width of a resilient (self-sustaining) mangrove green belt for coastal protection is a function of:

- The tidal range in relation to the mean bed slope within the belt (when more gentle, a wider habitat is found between mean sea level and high-high water spring);
- The wave climate (heights and periods) and the rate of wave damping within the forest. The latter are a function of the bed slope in the forest and the site-specific characteristics of the mangrove vegetation, such as stem thickness, stem density (and stilt root density & thickness in the case of *Rhizophora*), and mean canopy closure (Quartel et al., 2007);
- The intensity and frequency of extreme events which may damage the mangrove stands;
- The recovery potential of the mangroves, which also includes availability of sediment to compensate for possible habitat losses by erosion;
- The presence of wave reflecting structures in the back of the green belt.



**Figure 33.** Example of a mangrove 'greenbelt' protecting aquaculture areas from the sea (Mekong Delta)

There are some suggestions for a rule of thumb in the literature that express wave height reduction as a function of the width of a protective mangrove greenbelt (Quartel et al., 2007; Duncan et al., 2016). Mangroves can reduce the height of wind and swell waves by up to 66% over 100 m, and reduce the water level of storm surges between 5 and 50 cm per km of mangrove width (McIvor et al., 2012), although these figures are likely to vary between sites. A more open vegetation will be less effective in wave attenuation than a dense mangrove stand. Model calculations can assist in determining the required width of a mangrove greenbelt to attenuate regular wave heights to a 'safe height' (usually taken as 0.3 m) behind the forest (Bao, 2011). It should be noted that greenbelt requirements for storm surge attenuation may be substantially greater.

In Indonesia, government regulations have adopted a formula of  $130 \times$  difference between highest tide and lowest tide in a calendar year to determine the appropriate width of mangrove green belts (Ilman et al., 2016). For example, if the difference is 2 m, the width of green belt is 260m. Despite ensuing debates about its applicability to many of Indonesia's coasts, the regulation is the main operational guidance pertaining to coastal forest management (Ilman et al., 2016).

Coastal greenbelts of mangroves will also reduce salinity intrusion and fulfil a range of other ecosystem services, the effectiveness of which also depend on their width. Some studies suggest that 130 -150 times the tidal range is considered a minimum width for mangrove greenbelts to ensure sufficient tidal exchange of nutrients, food and sediment (Fiselier, 2016).

Most recommendations for the minimum width of mangrove greenbelts along shorelines for coastal protection have been in the order of 100-600m, but the wider the more resilient (Primavera, 2005; McIvor et al., 2012; Van Bijsterveldt et al., 2020). This is comparable to the planned width of 300m + 300m for the fully-protected and protected zones, respectively, for the Mekong Delta in the 2<sup>nd</sup> half of the 20th century (Hong & San, 1993).

For the width of greenbelts along riverbanks and waterways within an estuary few studies have been done, but some countries have prescribed guidelines. Generally speaking, the width of greenbelts along the banks of rivers and channels can be less wide, as these banks do not experience the significant effect of waves as mangroves along the shoreline. In the Philippines, a width of 20 to 50m has been suggested; the closer to the sea the wider the mangrove belt should be (Primavera, 2005a). In Indonesia, Forestry regulations of 1978 prescribed a standard width for mangrove green belts of 10m wide along riverbanks and watercourses (Ilman et al., 2016). In Mexico, protection of a 20m-wide strip of riparian mangrove vegetation along coastal lagoons and a 30m-wide strip along the banks of permanent rivers and channels is required as a minimum (Dominguez-Dominguez et al., 2019). Besides its width, the length of the riverbank along which a belt of mangrove vegetation is preserved is also important.

In some areas of the Mekong Delta, shrimp ponds may have to be given up to restore a wide enough mangrove greenbelt for coastal resilience (Figure 34).



**Figure 34.** Example of a shrimp farm in which a farmer gave-up part of his pond to expand the mangrove ‘greenbelt’ (Timbulsloko, Demak, Indonesia).

### 3.2 Human activities

At present most of the mangroves in the Mekong Delta are located in the protected and semi-protected zones along the coast. In most of these semi-protected zones, farmers have been contracted and allowed to convert about 40% of the forest into ponds. We recommend that these forests remain as large as possible for the protection of the environment. When these are removed for reasons of infrastructure improvement, a compensation, i.e., reforestation elsewhere, should become compulsory. This reforestation can be combined with land reallocations to improve the water management infrastructures for shrimp aquaculture.

In the Mekong Delta, the expansion of the mangrove area within the estuaries is physically easier to realise than that along the coast, and is important for two reasons: protection of the dykes and space for water. Moreover, making space for mangrove equals making space for water to reduce risk of flooding. The dykes and reduced risk of flooding are important for all infrastructures related to human activities. Only harbours and cities can afford the heavy-duty cays that can replace the mangroves. In rural areas near the coast in the zones dominated by brackish water, the most rewarding human activity that can be practised is shrimp farming, but, again, separating the stream of inlet and outlet water is crucial for sustainability. To include mangrove in the aquaculture dominated landscape, we propose to re-parcel the land.

#### **Mangrove and water management for aquaculture**

The separation of inlet and outlet water is crucial for an efficient aquaculture (see 2.9 and 3.2). Mangroves play a double role at both ends by preparing the inlet water and by cleaning the outlet water. Moreover, these forests protect the dykes that separate the ponds from the waterways in the estuary. To be cost-efficient in an estuary with many small aquaculture farms, as in the Mekong Delta, that separation needs to be designed at landscape level, just

as plains for irrigated rice. In such landscapes, the mangrove area and ponds with communal interest can be managed by a community, company or cooperative (Figure 35).



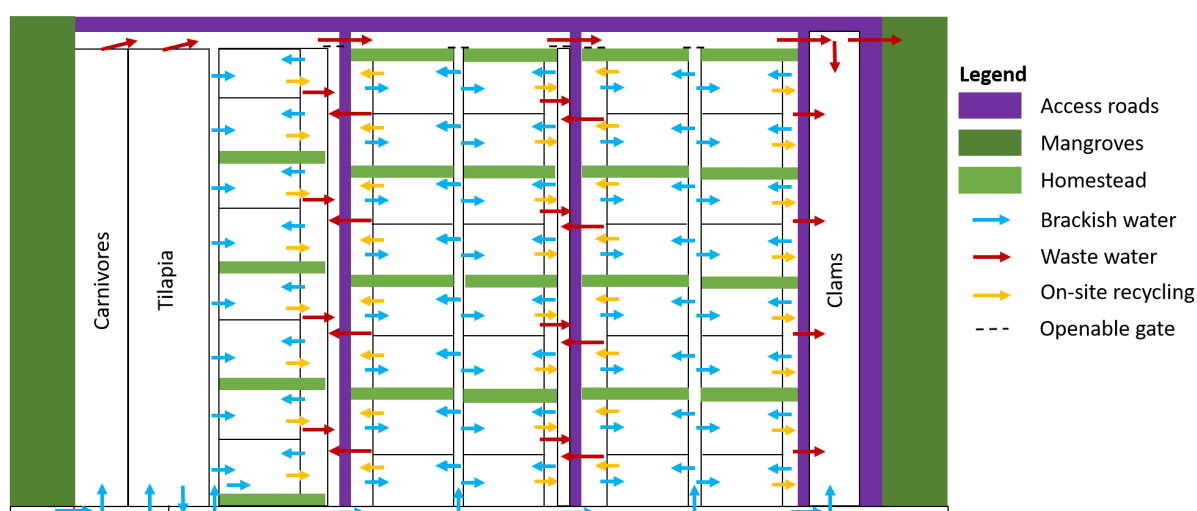
**Figure 35.** A brackish water landscape with a cascade of aquaculture ponds, behind a mangrove greenbelt; the brackish water intake is located in the estuary (from Tonneijck et al. 2016).

Moreover, first, the inlet water for shrimp farming requires a specific water quality and water from the estuary can't be used indiscriminately and need to be treated. Second, the effluents might need to be treated (according to environmental laws or to protect other farmers) before these can be disposed of in a waterway. Thus, to function optimally, each farm would need at least two types of complementary ponds: reservoirs for treatment and for settling. However, in reality small farmers of shrimp take the risk to do without those at the cost of frequent disease outbreaks. Therefore, Tonneijck et al. (2016) proposed a cascade landscape with two treatment ponds upstream (Figure 27). We propose to add a settling pond downstream to keep ahead of environmental demands, and thus, the following sequence in the cascade aquaculture system (Figure 36):

1. The inlet water from the brackish estuary flows in the first upstream treatment pond stocked with carnivores (e.g., sea bass). These eat the organisms that potentially carry disease agents, and thus clean the brackish inlet water of potential disease agents. If laws allow the use of specific pesticide (e.g. teaseed) and the farms use high quality filter for the inlet water the pond with carnivores becomes superfluous (SEAFDEC/AQD, 2011).
2. The second pond is stocked with the herbivore tilapia to adjust the water to the quality that's best for the shrimp's performance (Widowati et al., 2021; SEAFDEC/AQD, 2011). The area of the two reservoirs should be 20 to 25% of the total pond area (see below).
3. The water from the tilapia pond provides all shrimp ponds/farms downstream with intake water through central canals. The reservoirs mentioned above can also be drained independently from that central canal.
4. At the level of each farm, the above steps can be expanded with a pond with seaweed plus tilapia, parallel to the shrimp pond(s) to recycle water and reduce risks of water shortage, and disease outbreaks. If this is common practice, the total area of first reservoirs can be reduced with the area of this recycle ponds. With such a reservoir, the farms are future proof.
5. The effluents of all shrimp farms are diverted to one central drain that ends downstream in the estuary, and if possible, another waterway of the estuary. If the effluents of the shrimp farms are too rich, according to local environmental laws, this drain could include either a communal sedimentation pond with clams and seaweed or an artificial wetland. Such a wetland can border at a mangrove forest, or be part of it.



For the water quality to be improved in the two communal reservoirs with the carnivorous fish and with the herbivore tilapia, these should be stocked at low density: 5 to 10,000 fish/ha (SEAFDEC/AQD, 2011). This means also that either no feed should be added to these ponds, or just specific feed/fertiliser to improve or maintain the water quality. The maximum production of such non-aerated ponds without feeding is 2,000 kg/ha (Bosma & Verdegem, 2011); 4,000 kg/ha might be reachable if feeding is optimised for within-pond recycling of the fish faeces (Joffre & Verdegem, 2019). These yields can/should be reached without aerators. The water colour of the carnivorous and tilapia pond should be checked daily, and with appropriate measures, be kept at brownish-green (indicative for a good phytoplankton balance and mostly dominated by *Chaetoceros* spp). In the tilapia pond, the phytoplankton, with counts of 12 to 20 ml<sup>-1</sup>, should always dominate the zooplankton, and this ratio needs to be checked regularly (e.g., weekly). The optimum stocking rate of the clam and seaweed in the communal settling pond depends on the nutrients charge in the effluent water. The stocking rate of clams should remain far below 100 gr m<sup>-2</sup> to avoid recharge with waste, and that of seaweed should be adjusted to optimal growth.



**Figure 36.** Plan with communal mangroves (>4 Ha), ponds with carnivorous fish (2 Ha), tilapia (3 Ha), clams (>2 Ha) and separated inlets and outlet for 20 shrimp farms (1.25 Ha) with each 2 ponds and a recycling reservoir. The farms can have different dimensions, and the plan can be expanded to more shrimp farms with larger reservoirs for treatment, and more mangroves.

The volume of the tilapia pond will also depend on the need for water storage which depends on two seasonal factors: evaporation and availability during intake hours of water for refill of the carnivorous pond with the appropriate quality, in particular, its salinity level. When operated year-round, the total water consumption of a shrimp farm will be around 30,000 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> (Verdegem and Bosma, 2009). However, the water need will be lower when water is recycled in the fourth pond as done by CP in Thailand (Figure 26), as proposed above (Figure 36). Then, also the area occupied by the reservoir may be smaller.

The income from the communal ponds and mangroves will provide capital for the canal maintenance and may generate some income for the shrimp farmers. The first will be a cost reduction and the second, a bonus for the shrimp farmers.

### **Re-parcelling**

Larger farms (companies) design for themselves a system with separated water streams for ponds, and do this, hopefully, within a mangroves landscape (Figures 23, 24, 35). However, in a landscape with many individual small farms, such as the Mekong delta, re-parcelling of land and a land reform are needed. For a land re-parcelling to be successful, after the identification of a potential area, the population, in particular the farmers, need to be informed/trained and participate in the redesign (Hou-Jones et al. 2021; Meij and Vintges, 2021).

Such infrastructural developments for polyculture-cascade aquaculture systems should be considered on a par with those applied for the development of irrigation. Hardly any public investments have specifically been done in the water management of aquaculture, as most ponds were built in former rice fields, or by individuals along communal canals or waterways. Unit investment of US\$ 2000 to 5000 per ha is not uncommon in irrigation for food security, while the economic returns of rice and corn are low. As the economic returns of aquaculture in a sustainable landscape are much higher, the investment in re-parcelling is economically warranted – especially as the cascade will lead to a stabilisation of production and decrease of disease-incurred losses (see 4.1.4). These investments and developments will then need to be geared and fitted into the hydraulic and land use patterns of each locality, for which a diversity of cascade options can be designed (Tonneijck et al., 2016).

Ideally, the intake is situated in another branch of the estuary than the outlet. On narrow islands in the estuary, such a design and reallocation will be easy. On wider islands, additional main canals might be needed to make sure that the drain ends downstream of all shrimp farms.

### **Accompanying measures**

When re-parcelling the land, the number of farms will diminish, and the owners who lose their farm need to be compensated. In the Mekong delta, quite some farmlands are underexploited and in the hands of absentee owners, or the elderly awaiting their retirement. Some farmers can be offered livelihood alternatives (land elsewhere, Fitzgerald, 2000), while others, just financial compensation (absentee landowners) plus lifelong pension (elderly who stop farming). Describing the Indonesian systems, Fitzgerald (2000) proposed already incentive options, such as tax rebate or abatement on property tax of silvofisheries farms, or higher property tax on brackish water ponds that are abandoned, or low in production in former mangrove areas. In Indonesia, some districts already apply abatements for abandoned ponds with mangroves. Such “green tax” has been advocated as a means of addressing environmental externalities, and is cheaper to administer, does not distort economic activity and is fair (OECD, 2011).

The communal areas without shrimp can be managed as joined stock company (JSC), or cooperatives by the remaining shrimp farmers. These JSC need a manager and staff, and thus

offer permanent employment for persons who have difficulty living with the risk-related stress of farming but want to remain in the area.

### **Mangroves and livelihood activities**

Special considerations:

- The economic benefits of sustainable shrimp aquaculture are hard to beat in coastal zones where mangrove is the climax vegetation
- Sustainable shrimp farming requires separated canals for inlet and outlet water, and a cascade of ponds to reduce the risks of disease outbreaks and environmental pollution, and can be combined with an expansion of mangroves
- In a landscape with many small farms, water management infrastructure needs to be rebuilt and the land needs to be re-parcelled to make aquaculture sustainable
- The mangroves, the treatment reservoir and the settling pond can be managed as cooperative or Joined Stock Company

Tools: Re-parcel land to create a landscape with a cascade of ponds between fringes of mangroves

Challenges: Identify farmers/owners that can give-up their land and/or stop farming

Key: Participation of farmers/owners in re-parcelling

### 3.3 Mangrove restoration experiences

The past decades have seen a growing momentum of mangrove restoration efforts and initiatives across the globe in response to the consequences of widespread degradation and loss of mangroves and their associated ecosystem services. The following is a summary of key lessons learnt from a review of these efforts and associated literature.

#### Terminology

With regards to mangrove restoration, the terms ‘restoration’, ‘afforestation’ and ‘rehabilitation’ are often used interchangeably, although they actually refer to different things. ‘Restoration’ refers to returning an ecosystem back to its original state or condition (as much as possible), ‘rehabilitation’ refers to improving, augmenting or enhancing a degraded area with the expectation of a return of some improved level of ecosystem functioning, while ‘afforestation’ refers to planting of mangrove vegetation in areas where it previously didn’t exist.

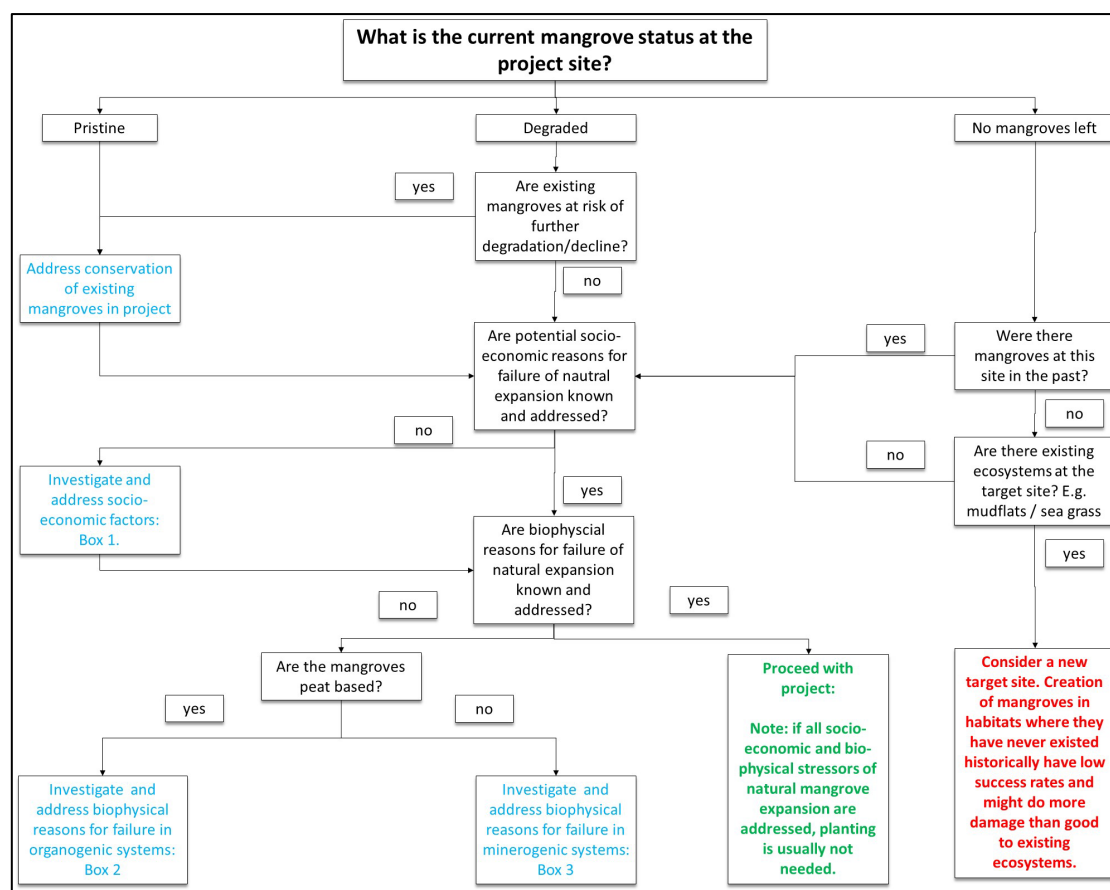
#### Objectives

Mangroves have been planted/restored for a wide range of objectives, such as re-greening, forestry (timber production), coastal protection, fish habitat enhancement, biodiversity conservation, strengthening of resilience, climate adaptation, full ecosystem restoration, livelihood rehabilitation, carbon farming and wastewater treatment.

#### Approaches

Efforts have ranged from small-scale manual planting activities at the community level to large-scale integrated climate-change adaptation programmes incorporating hydrological restoration, hybrid-engineering approaches and other nature-based solutions. Methodological approaches ranged from ‘assisting natural recovery’ through restoring conditions or recruitment, plantations of preferred species for forestry purposes, and community-based planting programs using propagules or nursery-raised seedlings, to beneficial re-use of dredged material, hybrid engineering and nature-based solutions.





**Figure 37.** Decision framework for mangrove restoration (©Wetlands International)

## Decision-making framework for planning of mangrove restoration

The appropriate start for any mangrove restoration program is to establish a conceptual framework for decision-making, including stages for planning, implementation and monitoring. A problem tree analysis can be useful to identify root causes and formulate project objectives (Sayaka and Erftemeijer, 1997). An example of a decision-making framework for mangrove restoration is presented in Figure 37. Logical step of the overall decision-making process include site identification and selection, identification of objectives, site characterisation, impact analysis, selection of approach/methodology for actions and measures (incl. estimation of costs), monitoring (using success indicators), and the publication/sharing of findings (Zaldivar-Jimenez et al., 2010). By far the most practical and proven methodology for successful mangrove restoration is the ‘Ecological Mangrove Restoration’ approach or EMR:

## Ecological Mangrove Restoration (general principles)

The following ecological principles, considerations and practical suggestions are based on a well-established process called "Ecological Mangrove Restoration" (Stevenson et al., 1999; Lewis, 2005; Brown, 2006; Brown et al., 2014; Lewis and Brown, 2014; Quarto and Thiam, 2018), building on lessons learnt from rehabilitation attempts worldwide (Erftemeijer & Lewis, 2000; Lewis, 2001; Primavera & Esteban, 2008) and supplemented with findings and recommendations from the most recent literature and personal experience:

- 1) **Understand the individual species ecology** of the mangroves at the site, in particular their patterns of reproduction, propagule distribution, and successful seedling establishment;
- 2) **Understand the hydrological patterns** (in particular the depth, duration and frequency of tidal inundation) that control the distribution and successful establishment and growth of (targeted) mangrove species. Determining the normal tidal hydrology of existing natural mangrove plant communities (at a reference site) in the area in which one wishes to do restoration is perhaps the single most important factor in designing a successful mangrove restoration project (Lewis, 2005);
- 3) **Understand the morphodynamic requirements** (i.e. fine sediment dynamics determined by the interaction of waves, tides and sediment) for successful mangrove establishment and survival, which is especially important along eroding mangrove-mud coasts (Stanley and Lewis, 2009; Winterwerp et al., 2013);
- 4) **Assess modifications of the original mangrove environment** that currently prevent natural regeneration (recovery after damage);
- 5) **Restore environmental conditions**, especially the tidal hydrology (for example through rehabilitation of creeks, removal of small dams further inland or installation of culverts to ensure appropriate tidal inundation characteristics and sufficient freshwater flow towards the mangrove stands) or sediment supply/stability and any other modifications that prevent mangrove regeneration;
- 6) **Facilitate natural regeneration** (through steps 5) of the mangrove vegetation by encouraging natural recruitment of mangrove propagules and successful plant establishment;
- 7) Only **consider actual planting** of propagules, field-collected seedlings, or nursery-reared seedlings after determining (through steps 1-6) 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. Plant the proper mangrove species (i.e. comparable to those found in similar, nearby reference sites) at suitable locations (above MHW), within the available “window of opportunity” (Balke et al., 2011), and at appropriate spacing, usually between 1-2 m (ICRI, 2020; Kairo and Mangora, 2020);
- 8) **Incorporate socio-economic aspects** by considering community participation, awareness, land ownership and alternative livelihood issues (Erftemeijer & Bualuang, 2000; Barbier, 2006, Barbier, 2008; Lewis, 2009; Hou-Jones et al., 2021), and address monitoring requirements.

## Common reasons for failures

Most mangrove restoration projects fail when there is too much emphasis on (immediate) planting of seedlings or propagules while neglecting the importance of hydrology and sediment dynamics. Poor site selection, choice of wrong species, unsuitable hydrological regime and lack of sediment supply/stability are some of the main reasons for failure. Survival of planted seedlings (even if in the right place) can sometimes be low due to barnacle infestation (Angsupanich and Havanond, 1996; Satumanatpan and Keough, 1999), seaweed & plastic entanglement (Erftemeijer, 2020; Van Bijsterveldt et al., 2021), livestock grazing (Minchington et al., 2019), crab infestation (Dangremond, 2015; Pearce, 2015), uprooting during storms and localised ponding after severe rainfall (Erftemeijer and Hamerlynck, 2005).

## Site selection considerations

Mangrove rehabilitation should preferably only be attempted in areas that formerly had mangrove vegetation, which was degraded or lost by some perturbation or human impact. Planting mangroves in areas that never supported mangrove growth before and expecting

some degree of success would require considerable ‘engineering’ measures, the costs of which should be weighed against the relatively low probability of success in such areas. Some important considerations for site selection include: understand why natural regeneration is not occurring at a site (e.g. unsuitable hydrology, sediment shortage, lack of shelter/stability, recruitment limitation); do not plant in water channels, seagrass beds, mudflats or raised sand flats; consider community involvement to ensure protection of the rehabilitation site; and resolving land ownership/use and accessibility issues (Lewis, 2009).

### **Overcoming ‘coastal squeeze’**

Where built infrastructure, agriculture, salt ponds or (shrimp) aquaculture ponds are placed (directly) behind coastal mangroves, it can prevent the otherwise natural migration of the mangrove habitat landward, resulting in habitat being ‘squeezed’ between advancing sea levels and extant hard infrastructure (Doody, 2012; Pontee, 2011; Lester and Matella, 2016; Besset et al., 2019b). This was also found to be a potentially significant issue in the Mekong Delta by Phan et al. (2015) who emphasised the need to maintain a sufficiently wide mangrove forest strip along the coast to attenuate wave energy, promote sedimentation and sustain a stable and healthy mangrove forest. Adequate responses to coastal squeeze involve integral coastal management strategies (regulating activities and use of space), implementation of mitigation measures and adaptive management plans to respond to emerging problems, restoration of natural habitats, landward relocation of certain infrastructure, and monitoring (Silva et al., 2020). An effective adaptive solution to coastal squeeze is to maintain enough space behind the mangroves to allow for a landward retreat of the mangrove vegetation in response to rising sea levels (Leo et al., 2020).

### **Coping with sea level rise and soil subsidence**

Most mangroves are generally considered capable of adapting and keeping pace with projected rates of sea level rise through vertical accretion and surface elevation change, provided they can trap sufficient sediment to keep up with the rising sea levels (McLeod and Salm, 2006), although the findings of a recent study suggest an upper threshold of 7 mm yr<sup>-1</sup> as the maximum rate of sea level rise beyond which mangroves may fail to keep up with the change (Saintilan et al., 2020). Thus, in areas deprived of adequate sediment supply or significant land subsidence, this ability to adapt to sea level rise may be compromised, resulting in increased vulnerability to climate change (Ward et al., 2016).

### **Cost and benefit considerations**

The costs to successfully restore both the vegetative cover and ecological functions of a mangrove forest have been reported to range from around 250 USD/ha to more than 216,000 USD/ha, depending on the degree of intervention and labour (topographic re-alignment, engineering, earth-moving, manual planting, monitoring etc.) that is required (Lewis, 2001). Mangrove rehabilitation projects can be classified into three categories: (1) planting alone, (2) hydrologic restoration (with and without planting), and (3) excavation or fill, with and without planting (Lewis, 2001). The first type, planting only, although inexpensive (in the order of a few hundred dollars/ha) often does not succeed due to a failure to appreciate the physiological tolerances of mangroves to tidal inundation and/or inappropriate site selection.

The second type, hydrologic restoration, can be done at similar or somewhat higher costs, and with proper planning has a fairly high success rate in many areas (e.g. restoration of abandoned aquaculture ponds). Planting in such cases should only be done if natural recolonisation fails, and can double the cost of a project. The third type, excavation and fill, is usually only carried out in areas where major modifications to hydrological characteristics and morphodynamics are required to (re-)establish the appropriate environmental conditions to allow for mangrove growth. This type is more expensive due to the costs of large-scale earthmoving measures and other engineering interventions.

A preliminary social-cost-benefit analysis of the mangrove restoration with permeable dams and some pond conversion for one village (Tambakbulusan) in Demak regency, Indonesia showed that restoring mangrove greenbelts is highly beneficial (Hakim et al., 2018). The baseline scenario assumed subsidence and erosion similar as villages closer to Semarang, where most lands were gradually engulfed the last 25 years. For such a period, a *no-intervention* scenario for the 724 ha would cost close to 3 million USD due to loss of land, infrastructure and livelihoods. Investing about 90,000 USD on *recovering mangrove-only* or on *improving aquaculture-only*, would generate benefits of about 8 and 1 million USD, respectively. Simultaneously investing on both mangrove recovery (climate change mitigation) and aquaculture improvement (climate change adaptation) would yield almost double: 15 million USD (Hakim et al., 2018).

### **Mangrove restoration guidelines**

For further details, reference is made to a number of excellent mangrove restoration manuals and guidelines, such as Field (1996), Marchand (2008), Lewis and Brown (2014), Primavera et al. (2014), Kairo and Mangora (2020) and ICRI (2020). For building the Associated Mangrove Aquaculture systems along waterways within the estuaries, see Bosma et al. (2020).



**Figure 38.** Mangrove restoration behind bamboo fences (Mekong Delta)



### 3.4 Involvement of local communities and stakeholders

Community participation in mangrove restoration (Figure 39) can be one of the most successful approaches to ensure long-term sustainable outcomes of a rehabilitation program (Primavera and Agbayani, 1996; Quarto, 1999; Erftemeijer and Bualuang, 2002; Camacho et al., 2020; Hou-Jones et al., 2021). Involving local communities or other local stakeholders in mangrove restoration and co-management is essential, as it offers participants a sense of empowerment and involvement in resolving their own environmental, social and economic issues of community development (Quarto, 1999; Quarto and Thiam, 2014). Participatory approaches promote a sense of ownership of the local natural capital (securing user rights) and transparency in its management (through shared governance) while valuing and strengthening existing traditional knowledge and local people's ability to identify solutions (Hou-Jones et al., 2021; Meij and Vingnes, 2021, and see also <http://coastal-protection-mekongdelta.com/> and <https://www.giz.de/en/worldwide/73408.html>).

However, community/stakeholder participation and co-management is a process that needs to be carefully managed and can come with its challenges and disadvantages, such as the potential for conflicting priorities (short-term (individual) socio-economic benefits versus long-term (communal) environmental solutions), diffusion of project objectives and expectations, reduced government control, risk of conflicts within or between neighbouring communities, and slower progress (Erftemeijer and Bualuang, 2002). This was also echoed in a recent study in the Mekong Delta that found that most measures failed to effectively conserve the mangrove forests (as a common pool resource) when private benefits from mangrove conversion increased with the pressures of population and economic development (Pham et al., 2019a; 2019b).



**Figure 39.** Villagers at work in a community-based mangrove restoration project in Indonesia

## 4 MANGROVE RESTORATION AND ASSOCIATED NATURE-BASED SOLUTIONS FOR COASTAL RESILIENCE IN THE MEKONG DELTA

### 4.1 Mangroves of the Mekong Delta

#### 4.1.1 Mangrove distribution and extent in Vietnam

Mangroves are located throughout the coast of Vietnam, being most pronounced around the Red River Delta in the north and the Mekong Delta and Ca Mau Peninsula in the south of the country (Hong and San, 1993; Veettil et al., 2019b). It has been estimated that mangrove forests in Vietnam covered an area of 400,000 ha prior to the Vietnam War, and that the use of herbicides ('agent orange' or dioxine) and napalm during the war destroyed more than 100,000 ha of mangrove forest (Hong and San, 1993). Decades later, the vast majority of the herbicides have broken down and washed away, leaving only localised remnant hotspots in areas where they were stored or leaked around former US military installations (Dwernychuck et al., 2006).

Further loss of an estimated 200,000 ha of mangrove forest during the post-war period has occurred due to the creation of shrimp ponds, salt ponds, paddy cultivation and infrastructure development (Hong and San, 1993; Truong and Do, 2018; Veettil et al., 2019a). For example, in Ben Tre Province alone, 52% of the total mangrove area was cleared for aquaculture and infrastructure between 1998 and 2015 (Veettil et al., 2019a). Similarly, in Tra Vinh province, total coverage of mangrove forests have decreased by 50% since 1965, mostly due to conversion for shrimp farming during 1995-2001 (Thu and Populus, 2007). Recent estimates (2013 and 2017) suggest that currently only some 150,000 to 168,000 ha of (mostly secondary or replanted) mangrove forest remains in Vietnam (Hai et al., 2020; Le et al., 2021).

Mangrove forests in Vietnam can be divided into four zones (Hong and San, 1993): Zone 1 (northeast) between Ngoc cape to Do Son cape (~39,400 ha in 1991); zone 2 (northern delta) from Do Son cape to Lach River mouth (~7,000 ha in 1991); zone 3 (central coast) from the Lach river mouth to Vung Tau cape (~14,300 ha in 1991); zone 4 (coast of southern Vietnam) between Vung Tau Cape and Ha Tien (~191,800 ha in 1991).

#### 4.1.2 Floristic composition and associated fauna

The floristic composition of mangroves in Vietnam is largely dependent on climatic and environmental conditions, which vary across the country (from north to south). Hong and San (1993) list 69 (true) mangrove species for Southern Vietnam, with *Rhizophora stylosa*, *R. mucronata*, *Kandelia candel*, *Avicennia officinalis*, *A. alba*, *A. lanata*, *Lumnitzera racemosa*, *Sonneratia alba*, *Dolichandrone spathacea*, *Phoenix paludosa*, *Acrostichum aureum*, *Clerodendron inerme*, *Acanthus ebracteatus*, *Cerbera manghas* and *Thespesia populnea* being the most dominant. Due to climatological differences, several mangrove species common in the South, such as *Sonneratia alba*, *Rhizophora apiculata*, *R. mucronata*, *Bruguiera cylindrica*, *B. parviflora*, *Avicennia alba*, *A. officinalis* and *Nypa fruticans*, are absent in the north, especially due to cold winter temperatures (Hong and San, 1993).

Mangroves in Vietnam are exceedingly rich in abundance and diversity of associated fauna, including fishes (258 species), molluscs (1969 species), shrimps (101 species), crabs (51 species), polychaete worms (51 species) and various other invertebrates, birds (some 386 species in the Mekong Delta alone) and a range of terrestrial animals (including 17 species of mammals and a variety of snakes and lizards). The mangroves play a key role in sustaining coastal fisheries and aquaculture and have been supporting a wide variety of traditional uses (Hong and San, 1993). An estimated 240,000 small-scale fishermen in Vietnam (44% of total) depend directly on mangrove-associated fisheries (Spalding and Leal, 2021).

#### 4.1.3 Tidal hydrology and mangroves

The natural patterns of tidal inundation and drainage are vital for the health & survival (and re-establishment) of mangroves. Van Loon et al. (2016), who conducted extensive hydrological investigations in the coastal mangroves of Southern Vietnam, summarised the individual inundation tolerance of different mangrove genera/species as follows:

**Table:** Hydrological classification of common Southeast Asian mangrove genera/species (Van Loon et al., 2016)

| class | elevation<br>[cm + MSL] | duration of inundation<br>[min per day] | duration of inundation<br>[min per inundation] | species                                                                  |
|-------|-------------------------|-----------------------------------------|------------------------------------------------|--------------------------------------------------------------------------|
| 1     | <0                      | >800                                    | >600                                           | none                                                                     |
| 2     | 0–50                    | 400–800                                 | 450–600                                        | <i>A. alba</i> Blume, <i>Sonneratia</i> sp.                              |
| 2*    | 50–100                  | 250–400                                 | 200–450                                        | <i>Avicennia</i> sp., <i>Rhizophora</i> sp., <i>Bruguiera</i> sp.        |
| 3     | 100–150                 | 150–250                                 | 100–200                                        | <i>Rhizophora</i> sp., <i>Ceriops</i> sp., <i>Bruguiera</i> sp.          |
| 4     | 150–210                 | 10–150                                  | 50–100                                         | <i>Lumnitzera</i> sp., <i>Bruguiera</i> sp., <i>Acrosticum aureum</i> L. |
| 5     | >210                    | <10                                     | <50                                            | <i>Ceriops</i> sp., <i>Phoenix paludosa</i> Roxb.                        |

#### 4.1.4 Mangrove management in Vietnam

Drivers of mangrove deforestation and degradation are complex, and often associated with provincial economic development strategies. Balancing environmental protection and economic development priorities in Vietnam has been challenging. Previous mangrove policies and projects in Vietnam aimed at offering social and economic incentives for mangrove protection were impeded by insecure tenure, land grabbing, elite capture and inequitable benefit-sharing, further hampered by overlapping and unclear mandates and responsibilities among government agencies at central, provincial and multilateral levels (Pham et al., 2019a).

After independence, with support from several donors, the government implemented reforestation with accompanying resettlement schemes along the coast of the Mekong Delta. Then, the protected forests along the coast were distinguished in two zones: the 300m strict protection zone and the 300m semi-protected zone (Vien, 2008). However, due to population growth, weak control on (semi-) protected forests (Truong and Do, 2018) and poverty, the mangrove cover was reduced again to a little more than the protected national forests along the coast in favor of developing (bio-)industry. In the coastal zone, the push to produce shrimp has led to a huge conversion of mangrove into shrimp farms between 1980 and 1995; e.g., in Bac Lieu and Ca Mau about 66,250 ha was converted (Buu & Phuong, 1999 cited by Tran et al., 2012a).

Since 1994-95, the management of mangrove forests in (South) Vietnam has been decentralised, allowing mangrove areas to be allotted and contracted to households for protection, management and sustainable use. There are three types of mangrove forests in Vietnam: protection, special use, and production. The protection forest (along the coast) is managed by the Forest Management Board, while most of the production forest is managed by the individuals under contract with semi-public Forest Companies or Board that also contract the households for silvo-fisheries and forest management (Tran et al., 2012a; Truong and Do, 2018).

A summary of the main policy and legislation regarding the restoration and development of coastal mangroves in Vietnam since 2015 is given in the textbox below:

#### **Mangrove restoration policy and legislation in Vietnam (period 2015-2020)**

##### Government policies for mangrove restoration and development:

- Decree No. 119/2016/ND-CP dated August 23, 2016 issued by the government on a number of policies on management, protection and sustainable development of coastal forests in response to climate change
- Decree No. 156/2018/NĐ-CP dated 16/11/2018 issued by the Government on Enforcement of a number of articles of the law on forestry, with Article 7 mentioning the basics for enlarging mangrove belt:

*Article 7:* Protection forests for tide shielding or sea encroachment prevention satisfying the following criteria:

- a. The width of forest stretch is from 300 to 1000 m according to each ecosystem in case of coastal aggradations for coastal stability
- b. The minimum width of forest stretch (shorelines) is 150 m in case of coastal erosion
- c. The width of forest stretch (river banks) is at least 20 m measured from the dyke footing with at least 2 stretches of forest trees for river mouths
- d. The width of forest stretch is at least 100 m (dyke available) and 250 m (without dyke) for any coastal lagoon

##### MARD Technical guidelines for mangrove restoration and development:

- Decision No. 1025/QĐ-BNN-TCLN dated April 8, 2016 of MARD on the promulgation of economic and technical norms for nursery, planting and care of mangroves
- Decision No. 1026/QĐ-BNN-TCLN dated April 8, 2016 of MARD on Promulgation of Technical Guidelines for afforestation of mangroves: *Kandelia obovata* & *K. candel*, *Aegiceras corniculatum*, *Avicennia officinalis*, *Bruguiera gymnorhiza* and *Sonneratia caseolaris*
- Decision No. 5365/QĐ-BNN-TCLN dated December 23, 2016 of MARD on Promulgation of Technical Guidelines for afforestation of mangroves: *Avicennia alba*, *Avicennia marina*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Sonneratia alba* and *Lumnitzera racemosa*
- Decision No. 608/QĐ-BNN-TCLN dated February 13, 2018 of MARD on Promulgation of Technical guidelines for agro-forestry-fishery production in coastal forest areas
- Decision No. 1162 (2021): Protection and development of coastal area and climate change 2021-2030 (including priority restoration of 6,800 hectares of mangroves for coastal protection in Mekong Delta)

##### National standards for construction, inspection, monitoring and evaluation of mangrove restoration initiatives (MoST in collaboration with MARD):

- TCVN 12510-2:2018: Protection forests for breaking waves and encroaching on the sea
- TCVN 12509-3:2018: Planted forests after the basic construction period - Mangrove species group



The households are paid an amount of money for managing the forests (though not in the case of silvo-fisheries). While this policy change was intended to reduce deforestation, it appears that the incentives the scheme currently provides are not always sufficiently attractive to discourage further mangrove conversion into other uses, unless the gains of a more secure land use right (by certificate) are accompanied by longer-term user responsibilities, restrictions and higher pay (Pham and Truong, 2019; Truong et al., 2021). The current incentives scheme encourages the planting of new mangroves but fails to adequately protect and conserve existing mangrove areas (Pham and Yoshino, 2016; Pham et al., 2019b). More favourable market conditions, such as higher wages and greater non-agriculture employment opportunities, are economic drivers that may contribute to reducing mangrove deforestation (Pham and Truong, 2019; Truong et al., 2021).

Recent initiatives across Asia have demonstrated that in order to link incentive allocation to mangrove-related activities and results in a fair and effective way, rights and responsibilities of participating entities need to be defined and clarified in several key areas, including mangrove ownership and use rights, administrative and financial management, activity implementation and monitoring (Albers and Schmitt, 2015; Broadhead et al., 2016).

#### **4.1.5 Past mangrove restoration programs in the Mekong Delta**

##### **Historic overview previous planting and restoration efforts**

Between 1975 and the early 1990s, the Government of Vietnam planted an estimated 53,000 ha of mangroves nationwide, followed by a further 14,000 ha between 1991 and 2002 through a series of mangrove rehabilitation projects with the support of international NGOs in eight provinces (Quang Ninh, Hai Phong, Ninh Binh, Thanh Hoa, Nghe An, Ha Tinh, Thai Binh and Nam Dinh) (FAO, 2007b). These efforts ranged from simple small-scale planting schemes by local communities to large-scale integrated restoration programs under expert guidance and supervision, implemented under a range of (sometimes challenging) conditions, with varying levels of success (including failures). Where successful, the planted and/or restored mangroves are contributing as a nature-based solution for coastal protection from the effects of climate change and sea level rise, mitigating the impact of natural disasters, protecting sea dyke systems and other infrastructure, and conserving biodiversity.

A number of programs and projects, both national and international, have invested in mangrove protection and restoration over the last 20 years (Pham et al., 2013). Some of these initiative were supported by international NGO's, such as, IUCN, Mangrove For the Future (MFF), UNESCO, WWF and UNDP. Some of the major mangrove reforestation programs up to 2010 included the following (Que et al. 2012):

- Mangrove regeneration project at Thai Binh, Nam Dinh and Ninh Binh provinces during 1990–1993, financially supported by various organisations, including ACTMANG of Japan, KVT (Netherlands), the Danish Red Cross and UNICEF of United Kingdom;
- Mangrove restoration projects at Thanh Hoa province implemented by the Government during 1996/1997 (PAM project 5325) and 1992-1997 (PAM project 4304).

- Mangrove restoration project at Quang Ninh province during 1996–2005, financially supported by the Red Cross Japan.
- World Bank-funded project on protection and development of coastal wetland areas in the South of Vietnam during 2002–2007, which was a continuation of the Rehabilitation of Mangrove Forest Project (1996-1999) under the Coastal Wetlands Protection and Development Project (Bentham et al., 1999). Nearly 6,000 ha of new mangroves were planted in the Mekong Delta under this program, along with a further 20,000 ha planted under the Vietnamese Government Program 661 during the same period.
- Restoration and development of coastal protection mangrove forests in Kien Giang and Bac Lieu provinces, a GIZ-funded project implemented during 2006–2010, which included extensive trials of planting mangroves behind bamboo/wooden fences (Figure 40).



**Figure 40.** Mangrove planting behind bamboo/wooden fences along the Mekong Delta

An estimated total area of 18,068 hectares of mangroves were planted in the various Mekong Delta provinces during the period from 1998 to 2010, with different species including *Sonneratia caseolaris*, *Avicennia alba*, *Avicennia marina*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Bruguiera parviflora*, *Bruguiera cylindrica*, *Ceriops decandra*, *Lumnitzera racemosa*, *Xylocarpus granatum*, *Nypa fruticans* and *Thespecia populnea* (GIZ, 2012).

More recent data from the Ministry of Agriculture and Rural Development recorded that during the period from 2015 to 2020, a total area of 10,080 ha of mangroves were planted and an additional 3,726 hectares restored nationwide, including 2,110 ha planted and 895 ha restored in the Mekong Delta Provinces (MARD, 2021). According to local plans, a further 5,200 ha will be planted and 3,800 ha restored nationally over the next five years (2021-2025) in response to global climate change, salinity intrusion and sea level rise. This will include 2,837 ha (to be planted) and 1,550 ha (to be restored) in the Mekong Delta (MARD, 2021).

### **Lessons learnt from previous programs**

Important lessons can be learned from the various mangrove restoration efforts and associated research findings in the past, that will help prevent a repeat of past mistakes and boost the success of future mangrove restoration efforts in the Mekong delta.

For mangrove restoration along eroding shoreline areas, solutions need to pay attention to many aspects. First of all, there must be a gently sloping alluvial flat towards the sea. The

encroachment must be done gradually and step by step, preferably in order from the inside to the outside. The potential for permeable solutions such as T-groynes or bamboo/wooden fences to reduce wave action and trap mud (promoting accretion; Figure 41), should be studied under local wave conditions and optimise economic and environmental efficiency. For the long-term strategy, a master plan should be made indicating areas where mangroves can be planted, where they can be planted with support structures, and where they cannot be planted (Hanh et al., 2018).



**Figure 41.** Mangrove rehabilitation involving *Melaleuca* and *Eucalyptus* fences in Kien Giang province to reduce wave energy and cause sediment accretion

Mangrove rehabilitation efforts in abandoned aquaculture pond areas should pay attention to changes to the quality and composition of the soil (Figure 42). According to an assessment by the Research Institute of Forest Ecology and Environment, the embankment of shrimp pond areas in Dong Rui commune, Tien Yen district, Quang Ninh province had changed the physical and chemical properties of mangrove soil. The banks surrounding the pond changed the tidal regime, thus limiting the alluvial depositions inside the shrimp ponds and resulting in a change of the mechanical composition of the pond's soil, e.g. the percentage of sand was higher than soil under natural mangroves. The content of organic matter and easily digestible phosphorus in the abandoned shrimp ponds had increased due to the feed residues.



**Figure 42.** Mangrove planting (*Rhizophora apiculata*) in abandoned shrimp ponds in the Mekong Delta

The process of excavating and dredging each year had caused the bottom of shrimp ponds to harden and the soil to mature, limiting the regeneration and growth of mangrove species. Regular water retention had made the soil in shrimp ponds more acidic than the soil under



natural mangroves (pH 3.11 - 3.17) causing aluminisation. For this site, planting mangroves with seedlings nursed in potting bags had 20-30% higher survival rates than planting with propagules and 3 species of *Rhizophora stylosa*, *Avicennia marina* and *Kandelia candel* with a density of 2,500 - 4,000 trees/ha could be used for afforesting on this site (Giang, 2016).

In the Mekong Delta, a GIZ pilot in Vinh Trach Dong commune, Hoa Binh district, Bac Lieu province (2011-2014) showed that mangrove restoration in abandoned shrimp ponds or highlands prone to less tidal flooding requires levelling or digging trenches (ditches) for areas where tidal water cannot reach. Among the tested local mangrove species, such as *Lumnitzera racemosa*, *Ceriops tagal*, *Xylocarpus molucensis* and *Intsia bijuga*, the first showed the best results on the grounds prone to less tidal flooding (Clough et al., 2016).

In 2015, Dat Mui Forest Management Board of Ca Mau province was supported by the ICMP program from GIZ and planted 11.8ha of mangrove trees in shrimp ponds of 20 households in Vien An commune, Ngoc Hien district, Ca Mau province. The initial planting density was 10,000 propagules/ha after levelling of the soil. Plants grew slowly, but the survival rate after 2 years of planting reached 82% (To, 2017).



**Figure 43.** Mangrove rehabilitation behind bamboo fencing in Soc Trang and Bac Lieu provinces

In Bac Lieu province, GIZ's reforestation program tested a method to make mono-specific mangrove plantations more resilient. The program created gaps of 80-100 m<sup>2</sup> in mono-specific forest stands where they planted other mangrove species to increase the diversity and boost the resilience (Summary Report, phase 1 of ICMP/CCCEP, 2011-2014).



**Figure 44.** Permeable bamboo fences and T-groynes (Mekong Delta, Vietnam)



In Soc Trang, Kien Giang and Bac Lieu provinces, T-fences and double fences made from bamboo or *Melaleuca* trees effectively reduced waves and trapped silt at suitable sites, creating habitat conditions for successful mangrove rehabilitation by planting the appropriate tree species (Figures 43, 44, 45 and 46). However, fences made of local materials were not durable and easily destroyed by extreme weather conditions such as storms and stronger waves than usual, so they needed to be repaired more often, increasing costs. In the long run, costs might be lower if more durable materials (such as cement poles) are used from the beginning (Clough et al., 2016).

Using bamboo fences, in 2020, Cu Lao Dung district, Soc Trang province, planted 40 hectares of *Avicennia* in potting bags in areas with too narrow forest belts. The province continues actions to widen the mangroves and to maintain the newly restored forest belt (Soc Trang Report, 4/2021).



**Figure 45.** Successfully recovering mangroves behind fences in Kien Giang province, Mekong Delta

Outside the Mekong Delta, one of the main initiatives has been in the Red River delta: the total terrestrial and marine area in the districts of Thai Thuy, Tien Hai, Giao Thuy, Nghia Hung and Kim Son, covered over 137,000 ha (<https://en.unesco.org/biosphere/aspac/red-river-delta>). There, the 'MANGROVE Project - Developing Knowledge-based Approaches to Reconcile Multiple Demands' (INCO-CT-2005-003697) observed that replanted mangroves and mudflats were a safe haven for migratory birds, that collecting and culturing clams on mudflats was a competing claim. Maintenance of the sites was challenged by insufficient embedding of action plan from pilot villages at the commune level and the dynamics caused by many interest groups. The project main advisor, Prof. Phan Nguyen Hong (Vietnam National University, Mangrove Ecosystem Research Division) stated that the quality of mangrove plantations can improve if *Kandelia obovata* is, after some time, gradually complemented or replaced with *Sonneratia caseolatis* to both diversify the mangroves species and enhance the protection role of the new forest. The research division had successfully planted *Sonneratia apetala*, growing quicker and being more tolerant to salinity and cold than *S. caseolaris* (Hong, oral communication, 2008; Hanoi workshop July 2008).

According to Thoi et al. (2018), mangrove rehabilitation and planting efforts at more difficult sites would benefit from a study on species composition and distribution of mangroves at (similar) existing sites, such as submerged coral reefs in coastal areas and islands of the South-Central Coast provinces. The Southern Sub-Institute of Development Research on Forestry

Science has proposed to test some species for planting on sand, gravel, rocks and coral debris: *Avicennia marina*, *Rhizophora stylosa*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Lumnitzera racemosa*, *Excoecaria agallocha* and *Aegiceras floridum*.



**Figure 46.** Successful mangrove recovery in Soc Trang Province (2012-15) following construction of bamboo T-fences that promoted sediment trapping and stability, allowing natural recruitment and regrowth (©GIZ)

These programs have provided the following valuable lessons on mangrove management:

- **Appropriate site assessment** is essential for successful restoration and replantation of mangrove forests, but has not been properly done in the past (Marchand 2008; Que et al. 2012; Phuong et al. 2016). Such an assessment requires good understanding of the biophysical conditions of the area planned for restoration and existing forest area intended for improvement. This includes soil properties, soil maturity, tide regime (depth, duration and frequency of tidal inundation) and water salinity, which are key factors influencing the growth of mangrove trees. Understanding biophysical conditions is the foundation for assessing the suitability of mangrove tree species according to their ecological requirements. Site assessment also applies to forested areas where the intention is to enhance and protect forest functions, such as improving forest canopy, diversifying tree species, and increasing tree density.
- **Ensuring quality seedlings is critical for forest restoration.** Depending on site conditions and silvicultural interventions, seedlings differ in term of ages, top height and stem diameter. The sources of propagules/seeds are also important to ensure quality seedlings. Lessons to date shows that seedlings should be produced locally to better fit local conditions.
- **Costs for mangrove restoration** are considerably higher at sites affected by strong sea waves, along eroding coastlines, at sites experiencing deep and frequent tidal inundation,

and on sandy soils. The cost of planting 1 ha of mangrove in such sites ranged from VND 90–500 million per ha (USD 4,000–22,700), compared to USD 1,000–2,000 per ha as suggested from key stakeholder interviews. The cost of setting up supporting construction items (e.g. sea wave break fences and sedimentation traps) is very high. For example, costs of installing sea wave break fencing in Kien Giang under the GIZ project were reportedly in the order of VND 350–400 million (USD 16,000–18,000) per km (Que et al. 2012). Costs for silt trap fences and wave barrier fences using *Melaleuca* poles at this site were approximately 10,000 and 20,000 USD/km respectively, with annual maintenance costs of 2,000–4,200 USD/km (Cuong et al., 2015).

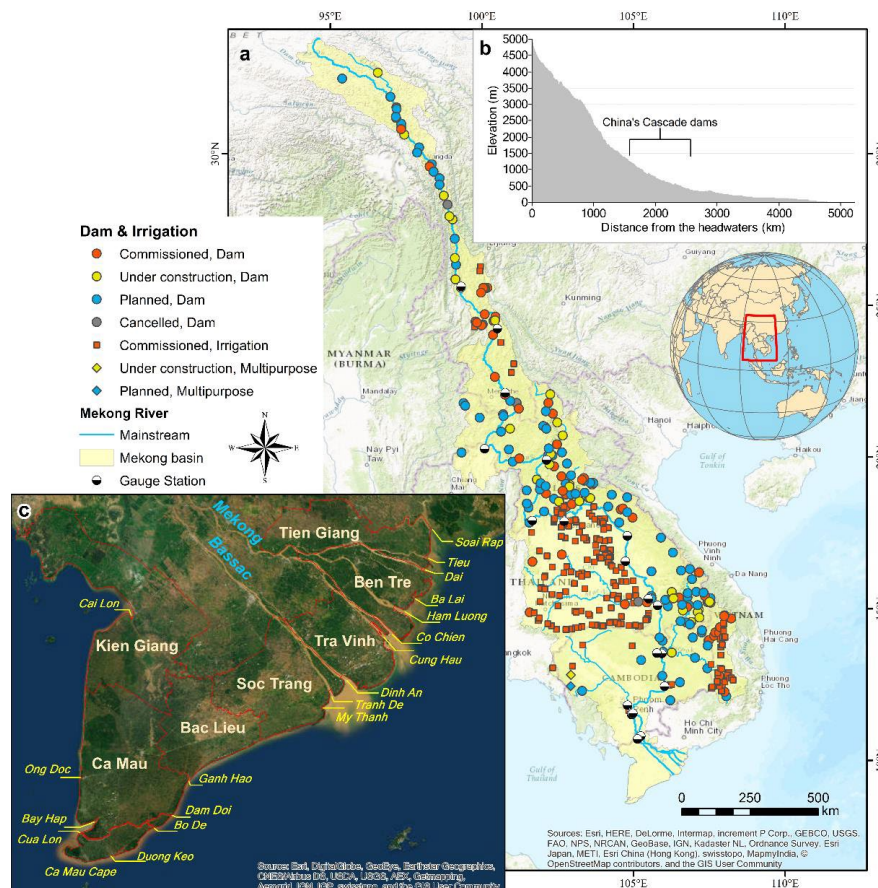
- **Mangrove management needs to generate and sustain livelihoods for forest-dependent people and communities.** Many people depend on mangrove forest areas for their livelihoods and income. Livelihood development models should be built on the consensus of local communities and their experiences.
- **Local communities and community-based organisations must be engaged in forest management.** Local communities and unions for women and young people play a central role in the success of mangrove forest management and development. However, local communities still manage or jointly manage only a limited area of mangrove forests. Several good examples of co-management of mangrove forests are in Hai Phong (Sam 2014); in Dong Rui, Quang Ninh (Phuong et al. 2016); and especially the community-based approach to management and restoration of mangrove forests in Da Loc, Thanh Hoa (Reed et al. 2014).
- **Monitoring and evaluation (M&E) of mangrove restoration projects is critical** for the long-term sustainability of mangrove forests. However, in most past programs, M&E was insufficient. A poor database of mangrove forests and restoration projects has led to poor planning for mangrove forest management and development.



## 4.2 Hydrology of the Mekong Delta

### 4.2.1 General description

In the following the abbreviation MD refers to the Mekong delta. Figure 47 shows the Mekong catchment ( $795 \cdot 10^3 \text{ km}^2$ ) with an overview of dams and reservoirs. The figure below also shows the project area, with the provinces Tien Giang, Ben Tre, Tra Vinh, Bac Lieu and Soc Trang, located around  $9^\circ$  latitude. At this latitude, effects of the earth's rotation (Coriolis) do not play a role in flow field and sediment transport along this part of Vietnam's coast.



**Figure 47.** Catchment of Mekong River with Mekong delta insert and overview of dams and reservoirs in the catchment (from [Xing Li et al., 2017](#)).

Figure 48 presents the major tributaries of the Mekong River in the delta. Note that the two main branches are referred to with different names:

- the main left branch: Mekong or Tien River
- the main right branch: Bassac or Hau River

In the following we distinguish between:

1. the estuary, i.e. the lower reaches of these rivers, where seawater and river water mix,
2. the estuary mouth, i.e. the coastal zone in front of the estuary, and
3. the coastal zone, i.e. the remainder of the coast.



The first two zones are found in the provinces Tien Giang, Ben Tre, Tra Vinh and the northern part of Soc Trang, while the latter two in the provinces Bac Lieu and the southern part of Soc Trang (Figure 49).

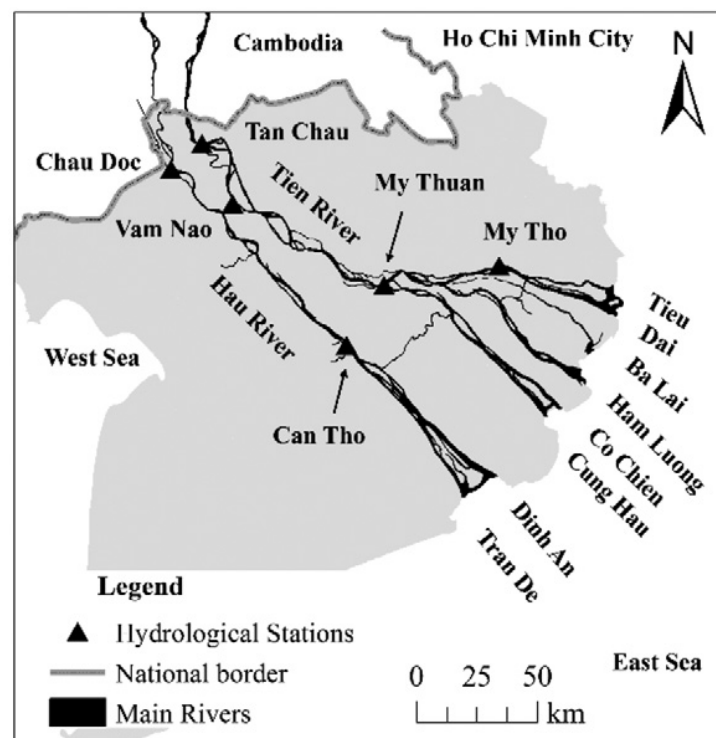


Figure 48. River's tributaries in the Mekong Delta (Van Binh et al., 2020).

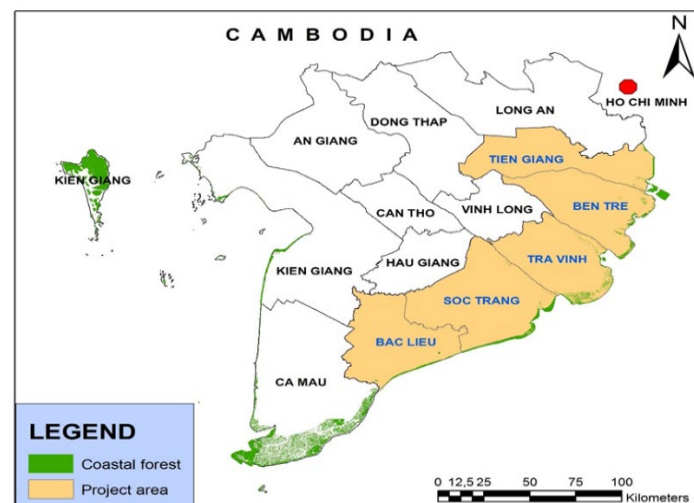
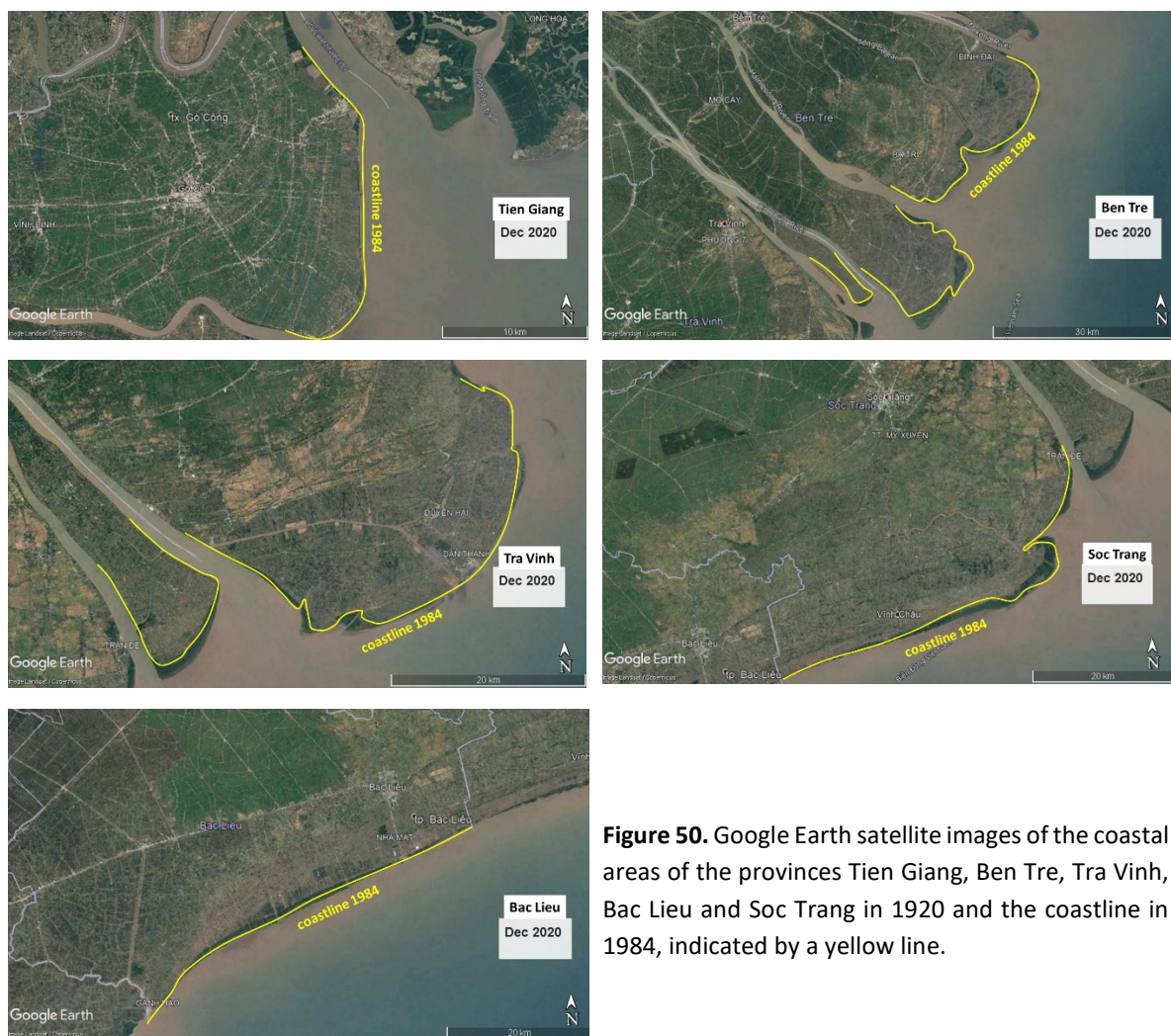


Figure 49. Map of project area, showing the five coastal provinces.

Figure 50 presents Google Earth satellite images from 2020 and the coastline in 1984 of these five provinces. These coastlines are characterised by narrow and wider fringes of mangroves with seawalls/dykes in the back to protect the low-lying hinterland from flooding.



**Figure 50.** Google Earth satellite images of the coastal areas of the provinces Tien Giang, Ben Tre, Tra Vinh, Bac Lieu and Soc Trang in 1920 and the coastline in 1984, indicated by a yellow line.

## 4.2.2 Climate

The coastal region of the Mekong Delta is characterised by a tropical (sub-equatorial) climate with high temperatures (averaging 28°C) and high humidity (averaging ~80%). The climate in the MD is governed by the monsoon (Dang Thi Ha et al., 2018). The rainy season in the Mekong Delta lasts from (late) May till October/November, when the SW-monsoon governs the local weather, bringing rain with a monthly mean precipitation of 8 – 10 mm/day (Li et al., 2019). The total annual rainfall is in the order of 1,500-1,800 mm per year (average 1670 mm), 85 – 90% of which falls in June – October (Bravard et al., 2013). The dry season lasts from November – April, when the NE monsoon generally brings dry weather with a few mm of rain per day, at most. Figure 51 presents the offshore wind roses, showing that the winds are more or less parallel to the coastline, or at small angles. These winds drive residual currents and waves (see below).

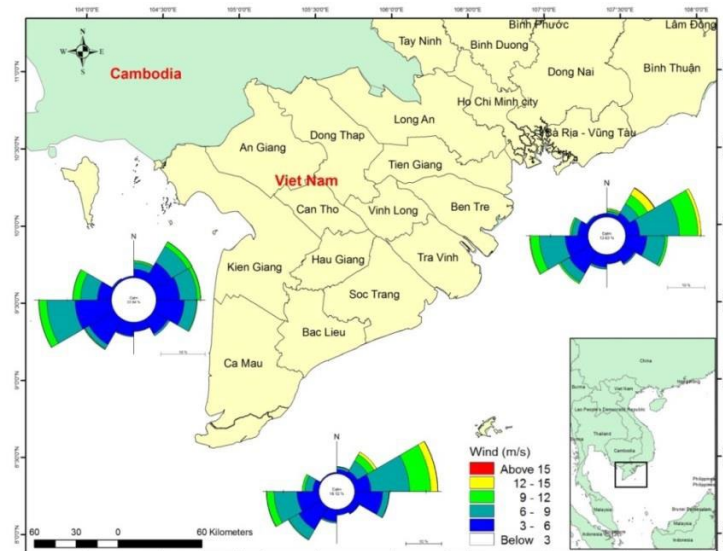
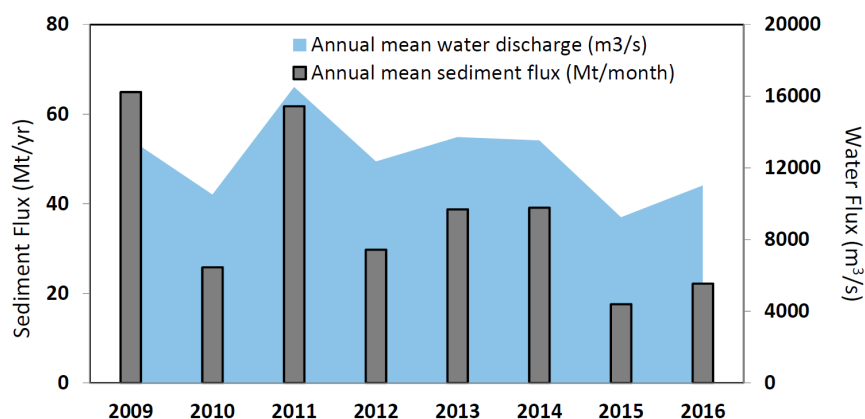


Figure 51. Offshore wind roses (Phan et al., 2019, Phan 2020).

Vietnam is frequently hit by typhoons. According to Anh et al. (2017), 786 typhoons hit Vietnam in the 20<sup>th</sup> century, though the Mekong Delta is affected less than northern Vietnam. Jain and Loy (2009) summarise the surges induced by Typhoon Sally (1996, 1.8 – 2.4 m surge), Linda (1997, 1.2 – 1.5 m surge), Xangsane or Chanchu (2006, 3.9 – 5.5 m surge) and Lekima (2007, 1.2 – 1.5 m surge). In particular the effects by Linda were studied in more detail, and surges of 1 – 2 m along the coast of Ganh Hao (at the border of Bac Lieu and Ca Mau) have been reported. Generally, typhoons bring heavy rains as well, but no data are available.

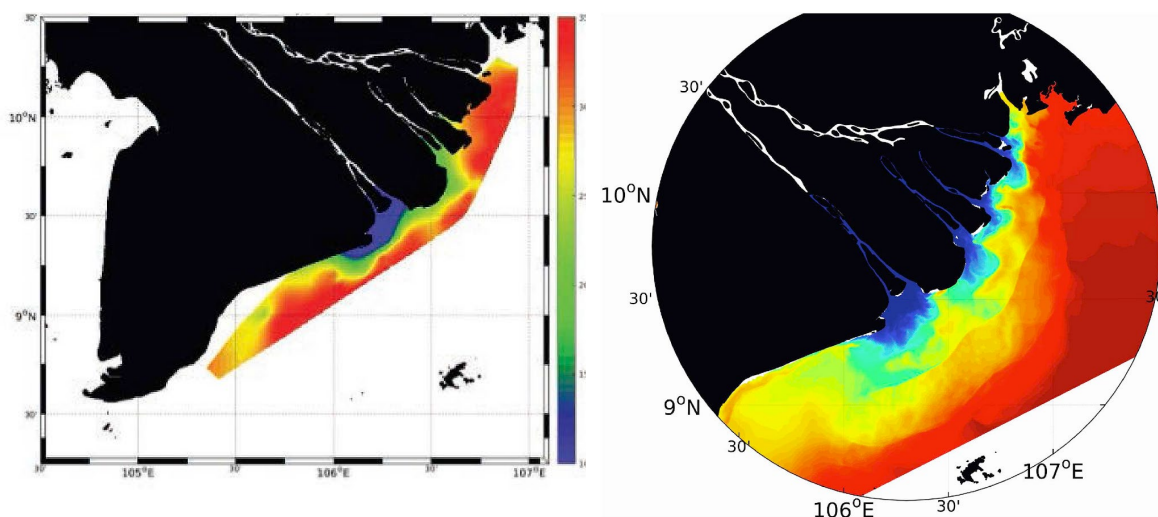
#### 4.2.3 River flows

The riverine system in the Mekong delta is quite complicated owing to the many branches and bifurcations. Next to these natural channels, complex networks are formed by man-made irrigation channels, inundating rice fields. The major contributors of fresh water are the Bassac (Hau) River and Mekong (Tien) River (Dang Thi Ha et al., 2018). As the focus of this report is on the MD coasts, the distribution of fresh water through these many branches is not relevant. Their joint discharge is shown in Figure 52, varying between 10,000 and 15,000 m<sup>3</sup>/s since the 1970s (early data not shown). The wet season brings high river discharges through the two main branches Bassac/Hau and Mekong/Tien (80 – 85%). A smaller, but still significant part of the discharge is overland (15-20%). Large parts (about 50%) of the delta get seasonally flooded with inundation depth up to 3 m, mainly the Plain of Reeds and the Long Xuyên Quadrangle. The river flow also exhibits considerable seasonal variations, with maxima in August – October, and minima in March and April.



**Figure 52.** Total freshwater discharge to the MD, data form two stations about 90 km from the river mouth at Can Tho (Bassac River) and My Thuan (Mekong River) (Dang Thi Ha, et al. 2018).

Fresh water from the rivers is rapidly dispersed into the East Sea, as winds are more or less parallel to the coastline, and Coriolis effects are negligible. In other words, one does not expect narrow bands of fresh in the foreshore which may trap fine sediment through gravitational circulation. Indeed, low salinities only occur near the mouth of the rivers, with slightly lowered salinities along the coastline further south, as shown in Figure 53. This implies that only in the direct vicinity of the estuary gravitational circulation plays a role (Thanh et al., 2017; Marchesiello et al., 2019), keeping fine sediment near the coast, thus along the provinces Tien Giang, Ben Tre and Tra Vinh.



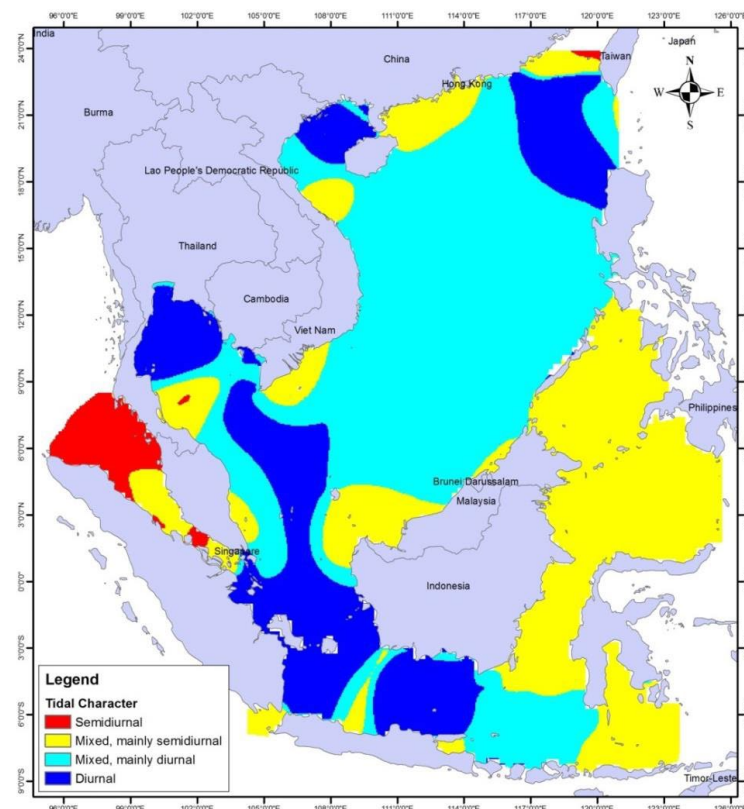
**Figure 53.** Measured salinity distribution in coastal zone. Left panel: dry season, February – March 2017, right panel wet season, October 2014 (Marchesiello et al., 2019).

#### 4.2.4 The tide

The Mekong Delta is characterised by mixed macro-tides with a maximum tidal amplitude of 3.8 m (mean 2.5 m) and a strong diurnal inequality (Wolanski et al., 1996). Whereas the tide in the major part of the East Sea is diurnal, it is mixed in the waters around the Mekong delta

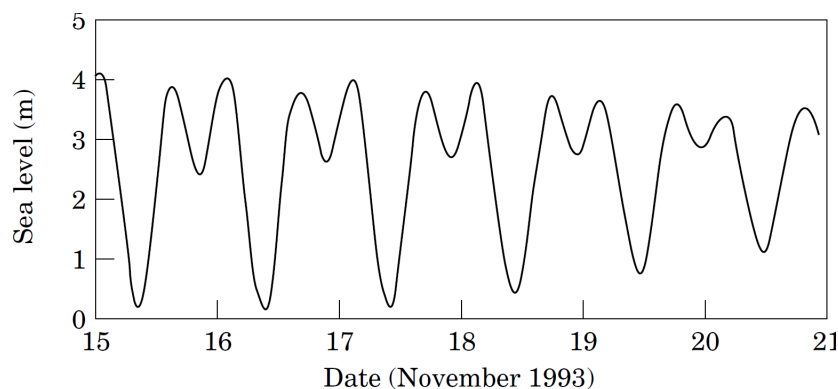


(Figure 54). Phan et al. (2019) and Phan (2020) suggest that this anomaly is caused by shoaling and resonance of the  $M_2$  semi-diurnal component.



**Figure 54.** Tidal character in East Sea (Phan et al., 2019 and Phan, 2020).

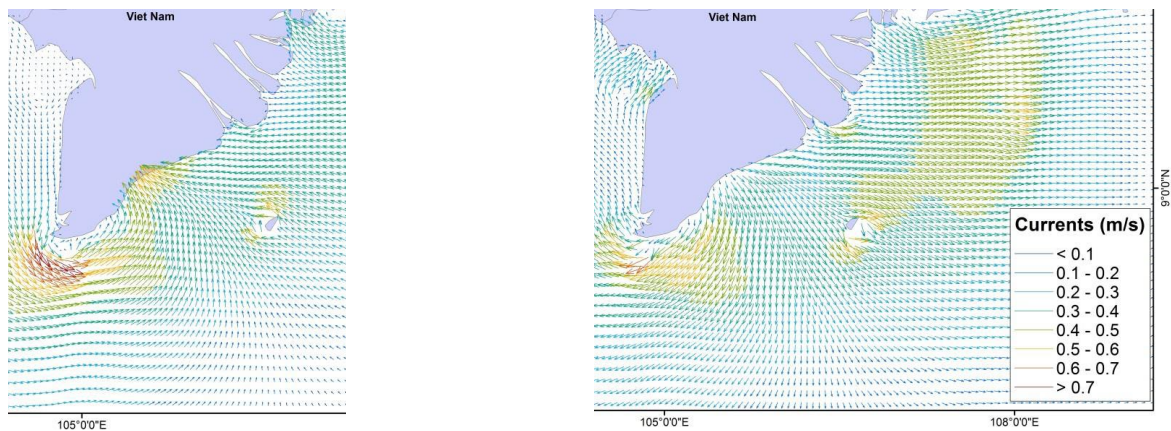
Figure 55 (below) presents an example of the tide, measured near Vung Tao, a little north of Tien Giang province (Wolanski et al. 1996). The mean tidal range near the river mouth amounts to 2.5 m, with maximal values of 3.8 m, lowering in southern direction.



**Figure 55.** Typical tidal curve measured at Vung Tau (Wolanski et al., 1993).

Figure 56 presents computed tidal velocities (Phan et al., 2019 and Phan, 2020) ranging from a few dm/s up to about 0.6 m/s during rising tide (left panel) and up to 0.7 m/s during falling tide (right panel). Note the nodal point around Bac Lieu province with flow velocities more or less perpendicular to the coastline. During rising tide, tidal velocities north of that point are directed southward, and south of this point northward. During falling tide, the opposite

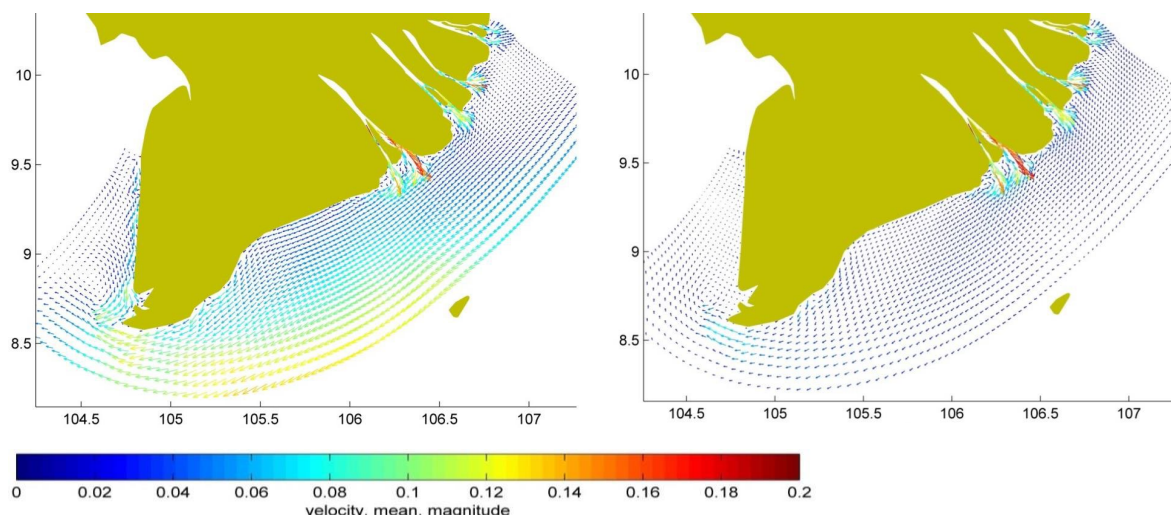
pattern is observed, i.e. northward tidal velocities north of the nodal point and southward velocities south of that point. This has important consequences for the transport of fine sediment, as discussed below.



**Figure 56.** Computed typical tidal velocities during rising tide (left panel) and falling tide (right panel) (Phan et al., 2019 and Phan, 2020).

#### 4.2.5 Residual currents

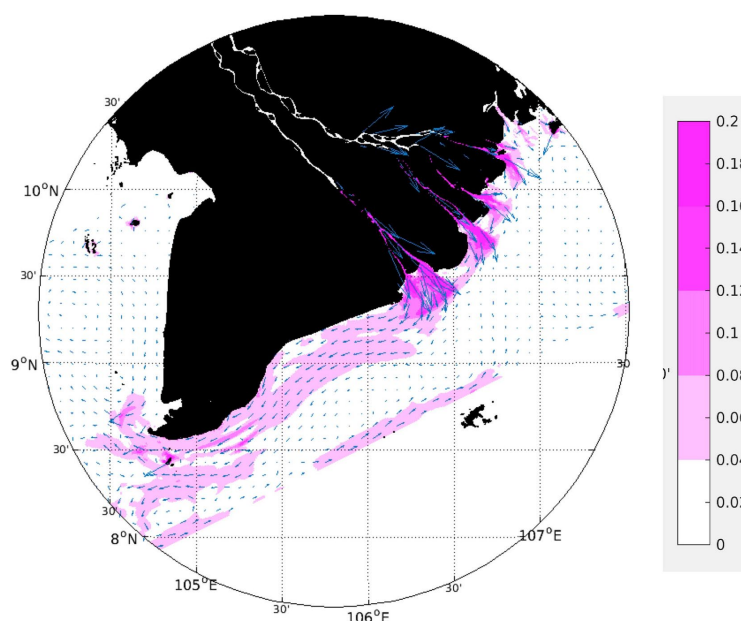
According to the modelling study by Phan et al. (2019) and Phan (2020) tide-induced residual currents are a few cm/s in southern direction. These currents are overshadowed by the role of the monsoon winds, yielding a southward residual of 2 – 4 cm/s during winter and 0 – 1 cm/s during summer into northern direction (Figure 57).



**Figure 57.** Computed residual currents (tide and wind) during winter conditions (left panel) and summer conditions (right panel) (Phan et al., 2019 and Phan, 2020).

Figure 58 present residual currents along the MD coastline, computed by Marchesiello et al. (2019), showing a southward pattern with velocities mainly between 4 and 8 cm/s, i.e. a bit higher than predicted by Phan et al. (2019) and Phan (2020).

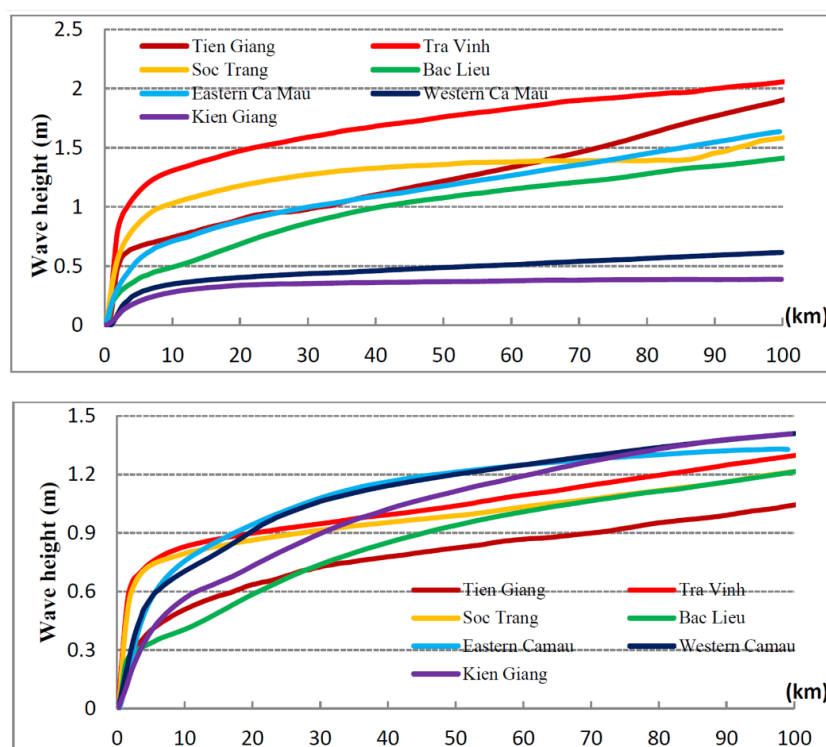
#### 2014 annual mean currents



**Figure 58.** Computed annual mean residual currents in 2014 (Marchesiello et al., 2019).

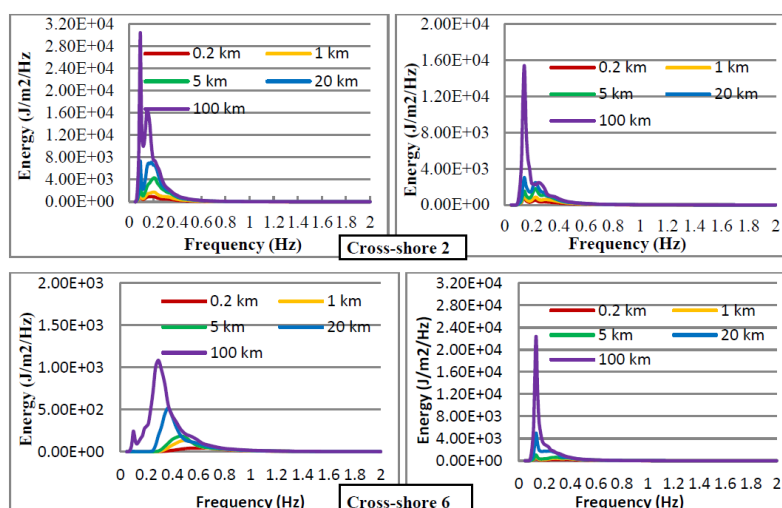
#### 4.2.6 The wave climate

No nearshore wave statistics could be found. However, Phan et al. (2020) and Phan (2020) carried out a numerical modelling study computing the wave transformation from locations 100 – 200 km offshore towards the coast (Figure 59).



**Figure 59.** Computed cross-shore wave height transformation in winter (upper panel) and in summer (lower panel) (Phan et al., 2020 and Phan, 2020).

As the breaker zone is found between 1 and 5 km offshore (see Section 4.3), nearshore wave heights are much smaller. The computed wave energy dissipation along two cross-shore trajectories is presented in Figure 60. As a result, winter wave heights at the Tien Giang shoreline measure 0.5 – 1.5 m with periods of 4 – 5 s, while summer wave heights at the Ca Mau shoreline measure 0.2 – 0.8 m with periods between 3.5 and 5 s.



**Figure 60.** Computed cross-shore Transformation of the wave energy spectra along the Mekong deltaic coast in winter (left column) and summer (right column) monsoon climate (Phan et al., 2020 and Phan, 2020).

#### 4.2.7 Groundwater extraction

Groundwater exploitation is a major cause of land subsidence, which in coastal areas poses a flood inundation hazard that is compounded by the threat of sea-level rise (SLR). In the lower Mekong Delta, most of which lies <2 m above sea level, over-exploitation induces large-scale hydraulic head (i.e., groundwater level) declines (Erban et al., 2014). During the past 25 years, the delta is estimated to have sunk an average of ~18 cm as a consequence of groundwater withdrawal. Current average subsidence rates due to groundwater extraction are estimated to be in the order of 1.1 to 1.6 cm yr<sup>-1</sup>, with some areas subsiding up to 2.5 cm yr<sup>-1</sup> (Erban et al., 2014; Minderhoud et al., 2017). If groundwater pumping continues at present rates, significant portions of the Mekong Delta will likely experience ~1 m (0.42–1.54 m) of additional inundation hazard by 2050 (Erban et al., 2014). See Section 4.3.2 for a more detailed discussion on subsidence.

#### 4.2.8 Salinity intrusion

The coastal area of the Mekong Delta has low and flat terrain. Climate change, sea level rise, subsidence in the delta and hydropower dams in the upper Mekong region are negatively impacting the Mekong Delta causing a lack of sediment supply, coastal and riverbank erosion, saltwater intrusion and droughts. There is a risk that as much as 40% of the Mekong delta area could be submerged if the sea level rises by 100 cm toward the end



of 2100. Drought and saltwater intrusion frequently occur in the coastal provinces of the Mekong Delta (Erban et al., 2014). In the period before 2012, salinity of 4 ppt only penetrated some 35 – 45 km inland. Since 2012, salinity intrusion has been increasing, reaching a record high 90 km inland in 2016. The dry season of 2020 saw severe drought and salinity intrusion causing extensive damage to agricultural lands, including 39,000 ha of rice production and >3,500 ha of durian plantations in Tien Giang province, and resulting in freshwater shortages for 95,600 households. Five provinces Ca Mau, Kien Giang, Ben Tre, Tien Giang and Long An declared a state of emergency due to salinity intrusion and several communities had to be evacuated. Saltwater intrusion has been worsened by the sediment starvation of the delta (Eslami et al., 2019), following to the construction of upstream dams, as described elsewhere in this report.

### Summary of Mekong Delta hydrodynamics

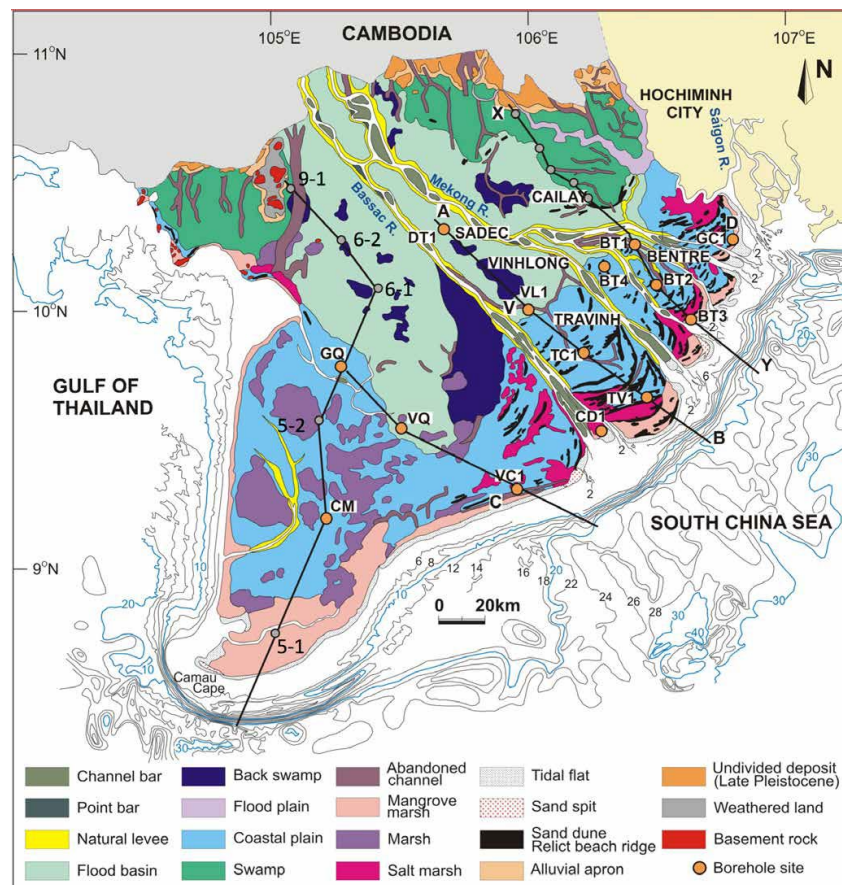
- The effects of earth rotation (Coriolis) are negligible;
- The climate is governed by the wet SW monsoon (May – October/November) and the dry NE monsoon (November – April);
- The Mekong Delta is frequently hit by typhoons, inducing surges, high waves and large rains;
- Typical river discharge varies between 10,000 and 15,000 m<sup>3</sup>/s, with pronounced seasonal variations – peak discharges in August – October, minima in March and April;
- Fresh water is dispersed rapidly in the coastal zone; gravitational circulation is important only within the estuary and close to its mouth in the coastal waters, thus along the provinces Tien Giang, Ben Tre, Tra Vinh and the northern part of Soc Trang;
- The mixed tide has ranges between 2.5 and 3.8 m;
- Tidal velocities during rising tide do not exceed 0.6 m/s, whereas during falling tide, maximal tidal velocities are below 0.7 m/s;
- Residual currents are driven by tide, wind and waves; during the NE-monsoon, they are directed southward and measure 2 – 4 cm/s, during the SW monsoon, the northward residuals are 0 – 1 cm/s;
- Waves lose much of their energy over the shallow, 5 – 10 km wide clinoform; in winter wave height and period measure 0.5 – 1.5 m and 4 – 5 s, during summer these are 0.2 – 0.8 m at 3.5 – 5 s.

## 4.3 The sedimentary system of the Mekong Delta (eastern coastline)

### 4.3.1 Geomorphology of the Mekong Delta and its historic developments

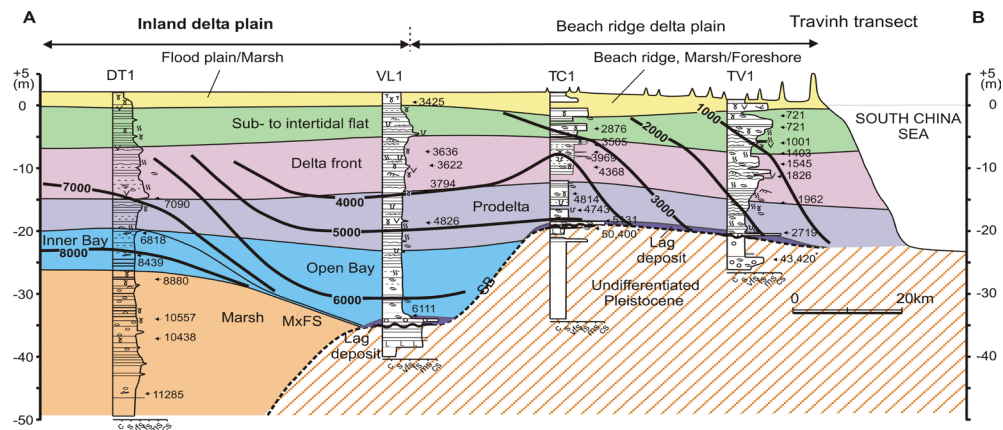
#### Geomorphological characterisation of the Mekong Delta

Figure 61 shows the geomorphology of the Mekong delta and near-shore bathymetry, as presented by Liu et al. (2017). While most clinoforms (sloping depositional surfaces) are 50 – 60 m thick, it is only 20 – 30 m thick in the Mekong Delta. As a result, the -10 m isobath is found 10 – 30 km from the shoreline. This explains the mild onshore wave climate, described above. The MD clinoform can be considered as the 10 – 30 km wide coastal fundamnet or shelf of the delta, where human activities occur. Historically, this fundamnet has been formed by sand and mud carried by the rivers. As the sand input from the rivers has decreased considerably (Section 2.3.3), the coastal fundamnet is expected not to grow or be maintained anymore, possible shrinking over time – however no data on this can be found in literature.



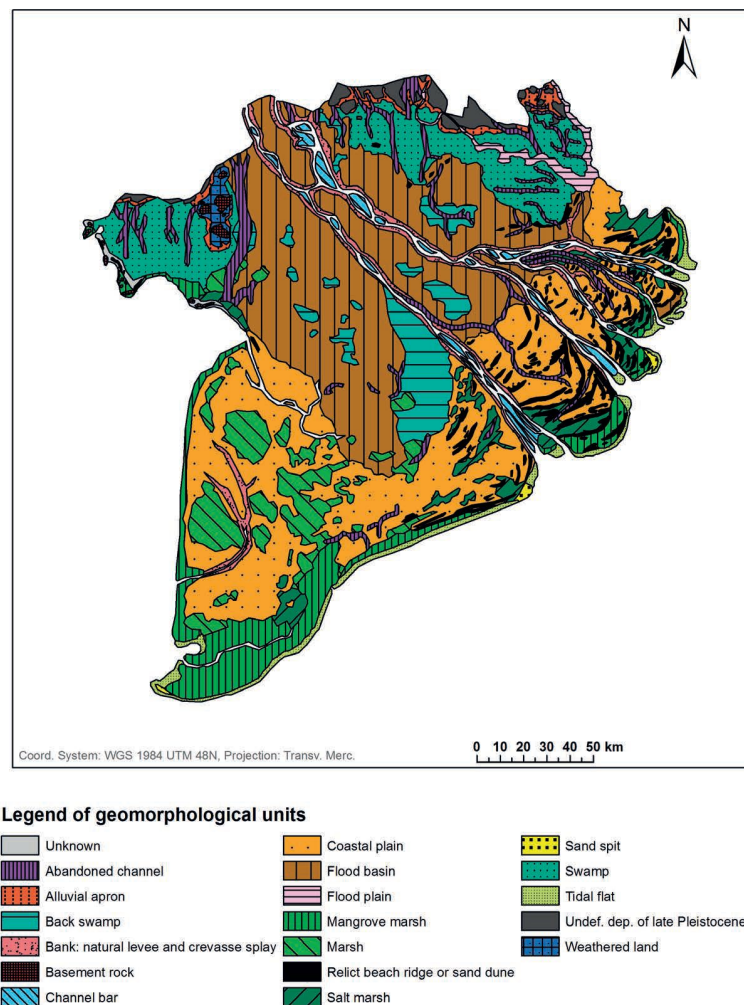
**Figure 61.** Geomorphology of Mekong Delta (Liu et al, 2017).

The MD clinoform is highly stratified, as depicted in Figure 62, showing transgression of the delta over many 1000s years BP. Layers of sand and consolidated mud alternate. Groundwater is extracted from the sand layers, repleted with pore water from the mud layers, as a result of which further consolidation occurs, manifesting itself by subsidence (sand layers hardly consolidate).



**Figure 62.** Depositional facies (clinoforms) showing delta transgression and stratification (Liu et al, 2017).

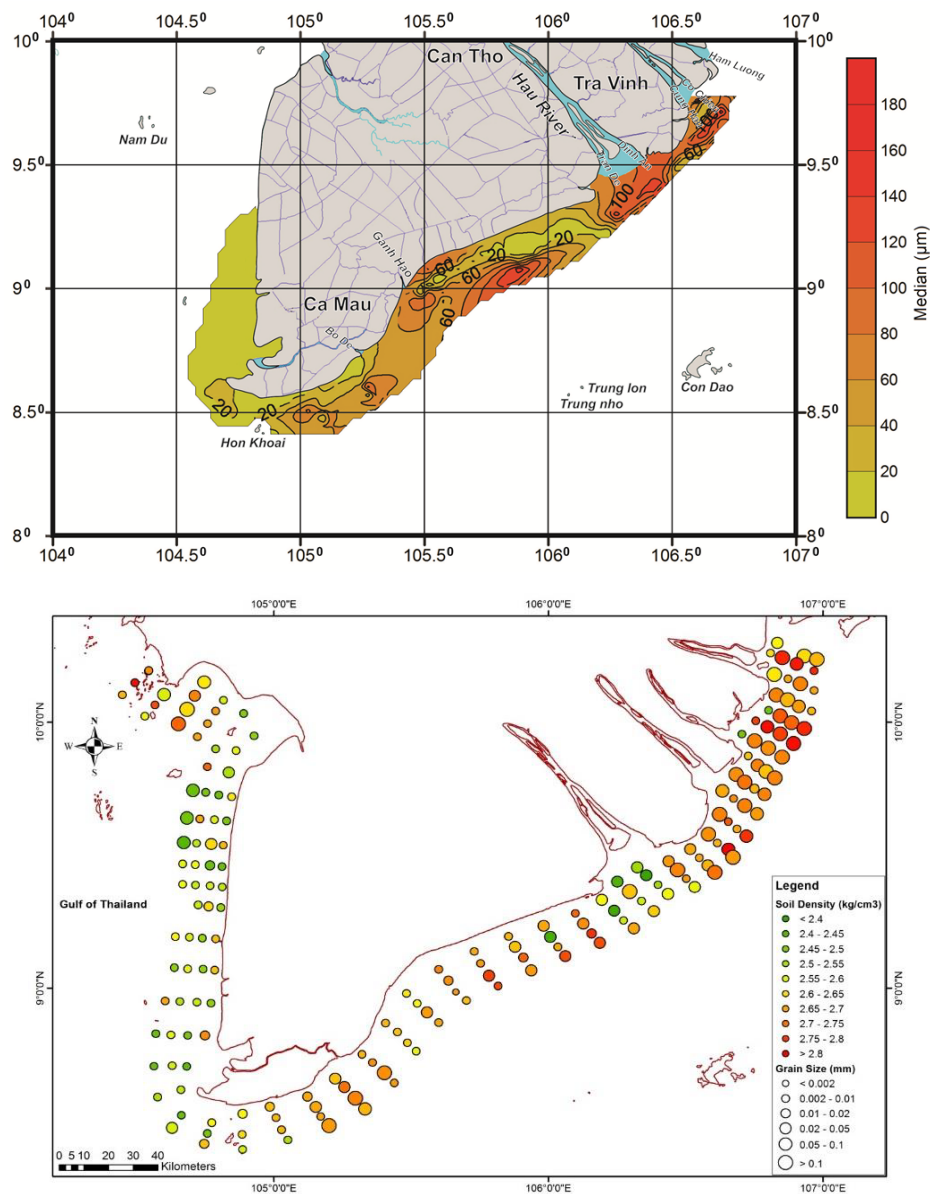
Figure 63 shows a geomorphological characterisation of the Mekong Delta. It is noted that Google Earth satellite images do not reveal these features anymore, likely because of current land use (urbanisation, agriculture and aquaculture). In particular, saltmarshes and sand ridges near the coast – which would have provided sustainable sea defences – have disappeared from the images. We expect the base of these ridges to be still present (sub-soil).



**Figure 63.** Geomorphological map of the Mekong delta (Minderhoud, 2019; based on Nguyen et al., 2000).

## Sedimentary structure of the MD clinoform

Figure 64 below present maps with the composition of the bed sediments on the clinoform. Coarser sand is only found in front of the river mouth, and at the edge of the clinoform in front of Bac Lieu. Elsewhere, along the east coast, mixtures of fine sand ( $D_{50} \approx 100 \mu\text{m}$ ) and mud are found.



**Figure 64.** Sediment composition of seabed from Nguyen Trung Thanh, et al. (2017) (upper panel) and from LMDCZ (2018) (lower panel)

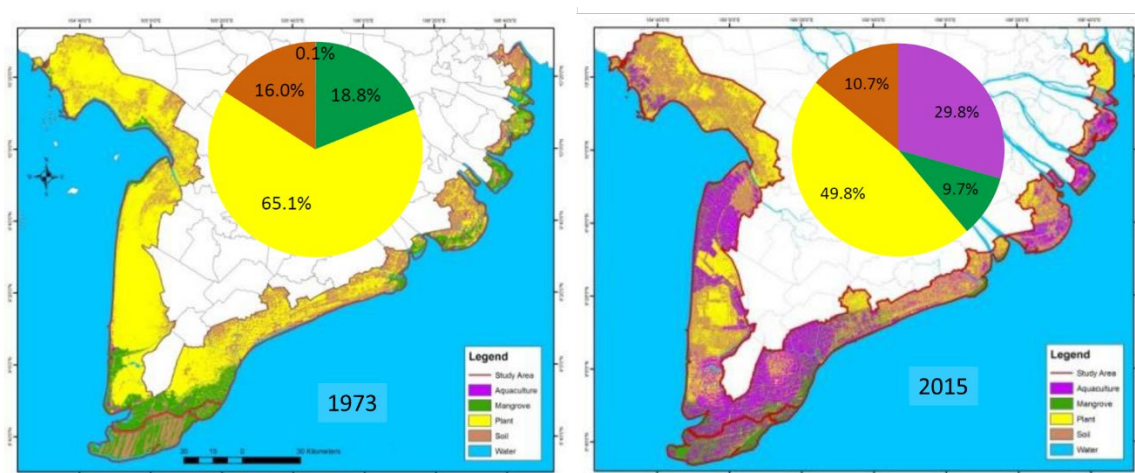
It is noted that all sediment deposited in the Mekong Delta stem from the Mekong River, though some deposits are 10,000s years old (Anthony et al., 2015; Liu et al., 2017; Tamura et al., 2020). The younger deposits are found closer to the estuary, and coast and clinoform are fed directly by sediments from the river. Further south, redistribution of relic sediments deposits by currents and waves is the dominant process.



### 4.3.2 Recent developments

#### Land use and coastal defence

Figure 65 presents a map (Phan, 2020; Phan and Stive, 2017) of the MD coastal area with land use in 1973 and 2015, distinguishing between five classes: aquaculture, mangroves, plants (rice crop, fruit trees, shrub, forest, etc.), soils (bare soil, urban, vegetable field with very low canopy cover, salt field, etc.) and water. This map shows a rapid increase in aquaculture area and a halving of the mangrove area (see 4.4.4).



**Figure 65.** Land use classification in 2015 by Phan (2020) and Phan and Stive (2017).

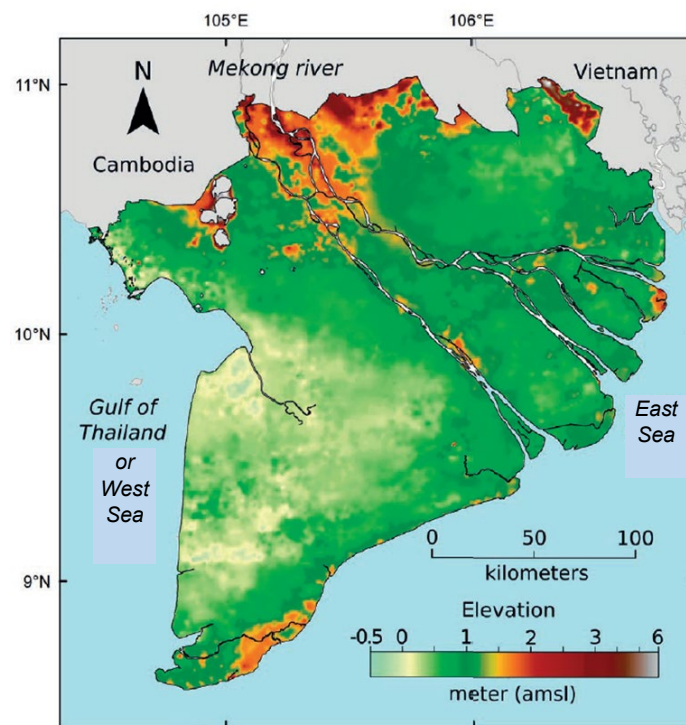
Figure 66 shows examples of the current shoreline with more or less solid bunds, seawalls and dykes. Though we have limited information at present, inspection of Google Earth satellite images suggests that large part of the shoreline is protected with these kind of structures. For instance, a major part of the Tra Vinh coastline is protected by either a massive seawall, or by coast-parallel detached breakwaters. This has profound consequences for the potential of mangrove recovery, as elaborated upon in Section 4.



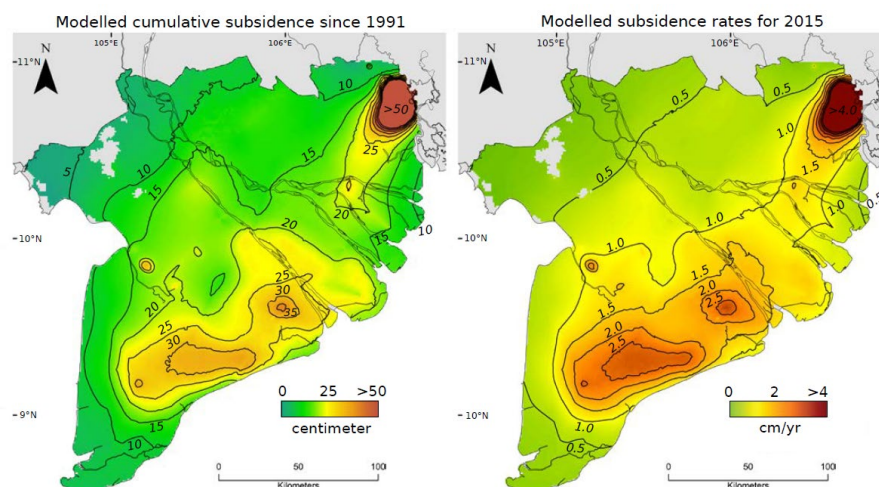
**Figure 66.** Examples of Mekong Delta shoreline at arbitrary locations along the east coast.

## Subsidence

The latest, and likely most accurate data on subsidence of the Mekong Delta is provided by Minderhoud et al. (2017, 2019, 2020) and Minderhoud (2019). Two methods were used, i.e. the so-called MERIT-DEM measurements (Multi-Error-Removed Improved-Terrain – Digital Elevation Model) and Topo DEM (Topographical Elevation Model DEM), which uses a large number of elevation points in the delta. Minderhoud (2019) used almost 20,000 points derived from a national digital topographical map by the Department of Survey and Mapping of Vietnam (2014). The results are shown in Figure 67, representing the situation in 2014.



**Figure 67.** New elevation model (Topo DEM) of the Vietnamese Mekong delta based on topographical elevation points of 2014 in meter above mean sea level (Minderhoud 2019).



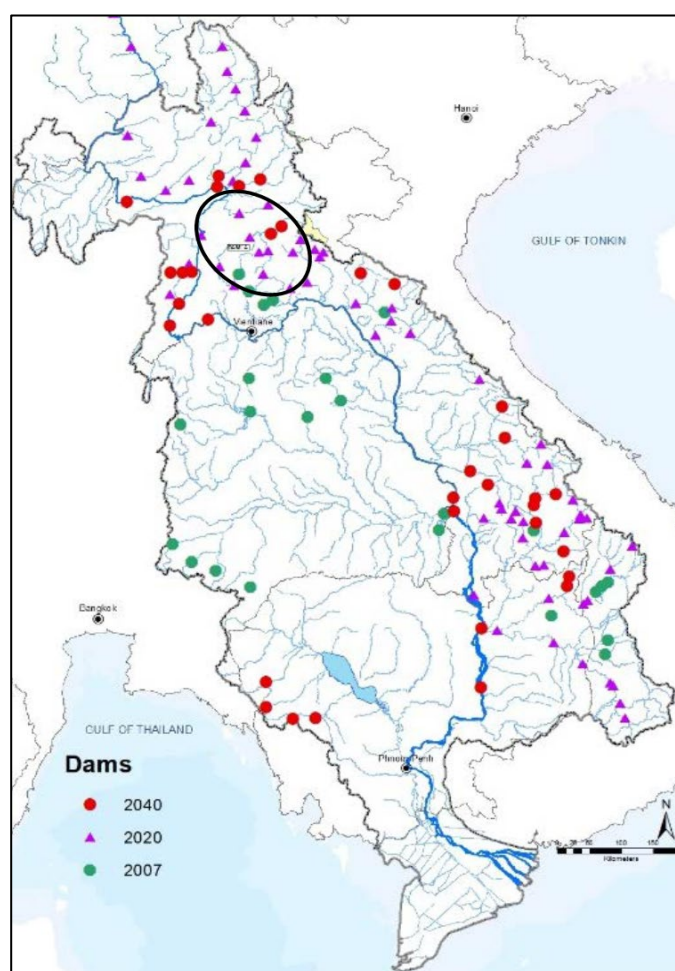
**Figure 68.** Subsidence rates due to groundwater extraction computed by Minderhoud et al. (2017) and Minderhoud (2019). A) cumulative subsidence from 1991 to 2016. B) annual subsidence rates for 2015.

Minderhoud et al. (2017) and Minderhoud (2019) developed a three-dimensional hydrogeological groundwater model with 15 vertical layers, representing 7 aquifers, based on 95 boreholes and literature values for the soil's compressibility. Groundwater extraction data from the ministries were used, indicating an annual growth in extraction of 2.5%, with current estimates of 0.7 – 2.5 Mm<sup>3</sup>/day. The results are presented in Figure 68 (above), showing subsidence rates well exceeding sea level rise.

The results indicate a MD-wide total subsidence since 1991 of 9 – 53 cm, with an average value of 18 cm, and hotspots up to 75 cm. Mean subsidence rates in the major part of the MD amount to about 1 cm/yr. The total subsidence in Ho Chi Minh City amounted to 90 – 150 cm, with an average of 115 cm, with present rates of 6.6 – 7.7 cm/yr.

### Dams and reservoirs

Figure 69 shows an overview of the major existing and planned hydropower dams in the Mekong River catchment (MRC, 2018). The black circle locates the so-called Lancang cascade, which would block major amounts of sediment fluxes towards the river (Van Bing et al., 2020).



**Figure 69.** Existing and planned hydropower dams in Mekong catchment (MRC, 2018) and the proximate location of the Lancang cascade (Van Bing et al., 2020).

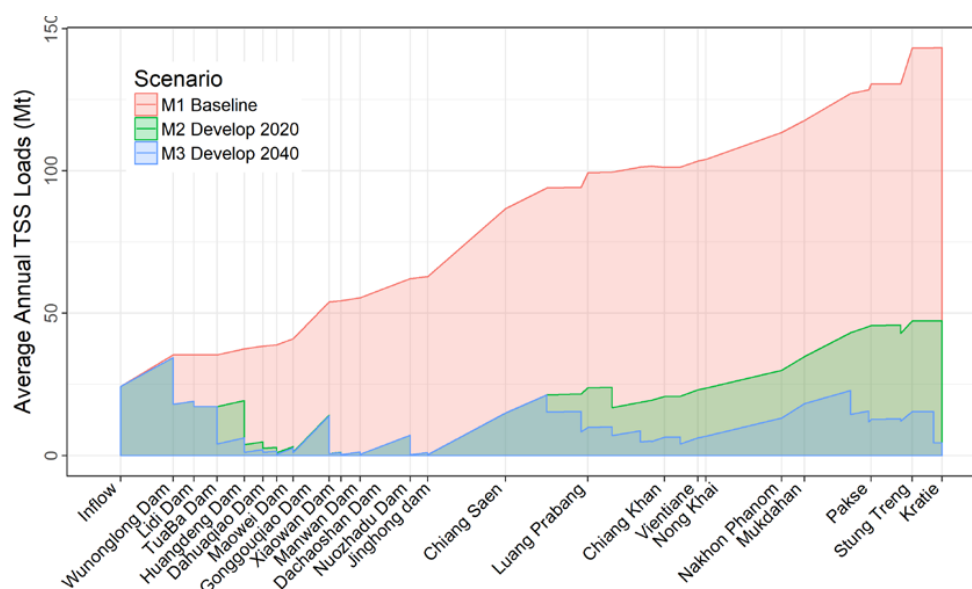


The Table below gives the capacity and impact of the six dams in the Lancang cascade. Their total capacity amounts to 41.8 km<sup>3</sup>, trapping 0.17 km<sup>3</sup> of sediment per year, or approximately 0.13×10<sup>9</sup> ton/yr (130 Mt/yr) of sediment.

**Table:** Capacity of the six Lancang cascade dams (Van Binh et al., 2020): MAR = mean annual runoff, MAS = mean annual sediment, CAP = reservoir total capacity

| Reservoir   | Filling (Year) | Catchment (km <sup>2</sup> ) | MAR (km <sup>3</sup> ) | MAS (km <sup>3</sup> ) | CAP (km <sup>3</sup> ) |
|-------------|----------------|------------------------------|------------------------|------------------------|------------------------|
| Nuozhadu    | Nov. 2011      | 144,700                      | 55.2                   | 0.0379                 | 22.37                  |
| Xiaowan     | Dec. 2008      | 113,300                      | 38.5                   | 0.0297                 | 15.13                  |
| Jinghong    | Apr. 2008      | 149,100                      | 58.0                   | 0.0391                 | 1.23                   |
| Manwan      | Mar. 1993      | 114,500                      | 38.8                   | 0.03                   | 1.06                   |
| Dachaoshan  | Nov. 2001      | 121,000                      | 42.3                   | 0.0317                 | 0.88                   |
| Gongguoqiao | Sep. 2011      | 97,300                       | 31.1                   | 0.0255                 | 0.51                   |

The impact of these dams on the sediment yield along the river into the estuary is depicted in Figure 70, where the baseline represents the conditions of 2007, the M2 scenario includes all dams planned to be operational in 2020, and the M3 scenario represents existing plans for 2040.



**Figure 70.** Longitudinal plot of sediment loads in the Mekong River from China (Lancang River to Kratie) (MRC, 2018).

Dams generally trap the coarser sediment, i.e. coarse sands. Most of the finer material (silts, “wash load”) can pass the dams undisturbed. Hence, the sediment arriving in the delta is expected to become finer. The coarser sediment is transported as bed load, which is a slow process. Hence it may take some time (decades?) before the impact of the dams becomes

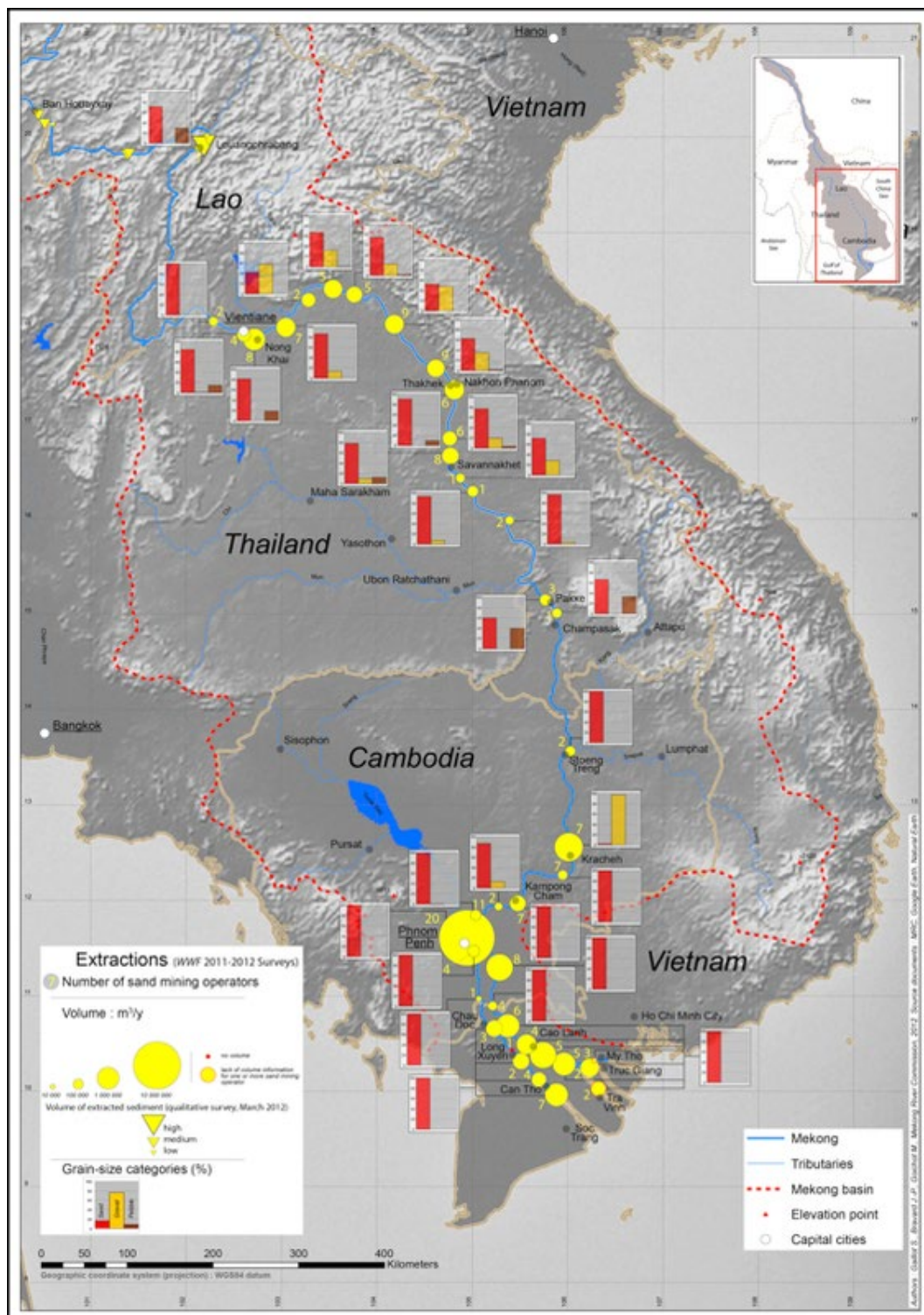


visible in the delta. Note that none of the available data and literature addresses these potential changes in sediment composition, nor the time scales upon which these would become manifest.

Tamura et al. (2020) analysed the coastal development of Ca Mau peninsula, the most southern tip of the Mekong Delta on the basis of (ancient) maps and borings. They concluded that the peninsula grew by 2 – 6 km<sup>2</sup>/yr in the first 1700 years AD. They argue that the dam constructions discussed above, nor fluvial sand mining, nor a local decline in mangroves can account for the current erosion of the eastern coast of Ca Mau, or for the correlative erosional features identified in the nearby subaqueous delta. The data reveal that the significant decrease in the sediment supply and subsequent erosion likely started between 1885 and 1940. It is likely that large scale irrigation and cultivation (24,000 km<sup>2</sup> by 1930) along the rivers in the delta sequestered large amounts of fine sediment, trapped in the many drainage/irrigation channels made and through overflow of the rice fields. The authors were not able to quantify the amount of sequestering, but it must be several 10s Mton/yr, given the sediment loads (see Figure above).

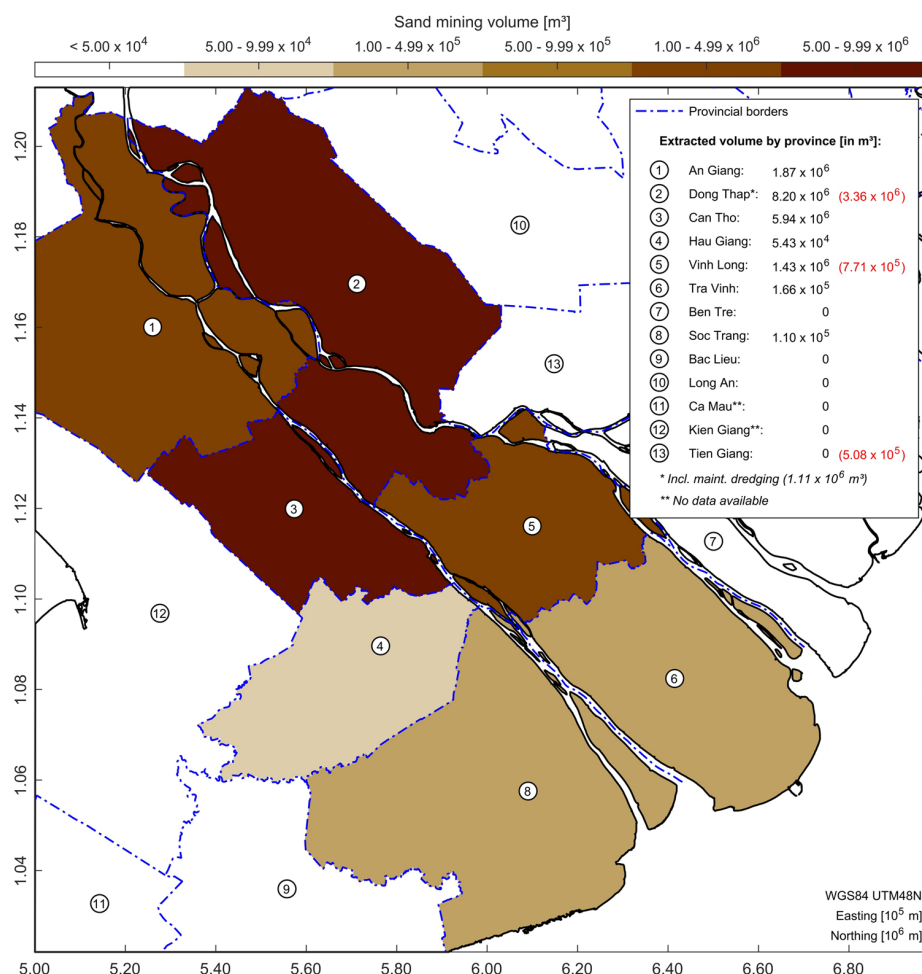
### **Sand mining**

Figure 71 shows that sand mining takes place along almost the entire river. Bravard et al. (2013) estimated that in 2011 a total volume of 34.5 Mm<sup>3</sup>, or 55.2 Mton of sediment was extracted from the Mekong main river in Laos, Thailand, Cambodia and Vietnam. The majority was sand, accounting for 90% of the extractions, i.e. about 31 Mm<sup>3</sup>, or 50 Mton. However, one may not relate these extractions directly to sediment losses towards the delta, as part of these extractions would be lost anyway behind the dams.



**Figure 71.** Map of extraction sites along the Mekong River, showing the extracted volumes and grain-size categories (Bravard et al., 2013).

However, sand extracted in the Lower Mekong River affects sand supply to the coastal zone directly. Figure 72 (below) shows over 25 Mm<sup>3</sup> sand extraction in 2018 from the Lower Mekong River (Jordan et al., 2019). These figures are the official numbers provided by Vietnamese authorities, hence extracted with concessions. Bravard et al. (2013) suggest that the actual volumes may be a factor 2 – 3 higher because of illegal mining activities. Comparison with Figure 70 shows that these volumes become of the same order of magnitude as the remaining sediment flux affected by trapping behind the dams. Indeed, Xing Li et al. (2017) suggest that the Lower Mekong River now imports sandy sediment from the coastal zone, thus aggravating coastal erosion.



**Figure 72.** Sand extraction concessions in the Lower Mekong River in 2018 (Jordan et al., 2019) and estimates (red numbers) from bathymetrical surveys.

Sand mining in the lower Mekong River induces severe bank erosion along the river (Figure 73), feeding the sediment transport capacity of the river (Bravard et al., 2013; Hackney et al., 2020).



**Figure 73.** River bank erosion in the MD as a result of sand mining (photo Kackney et al., 2020)

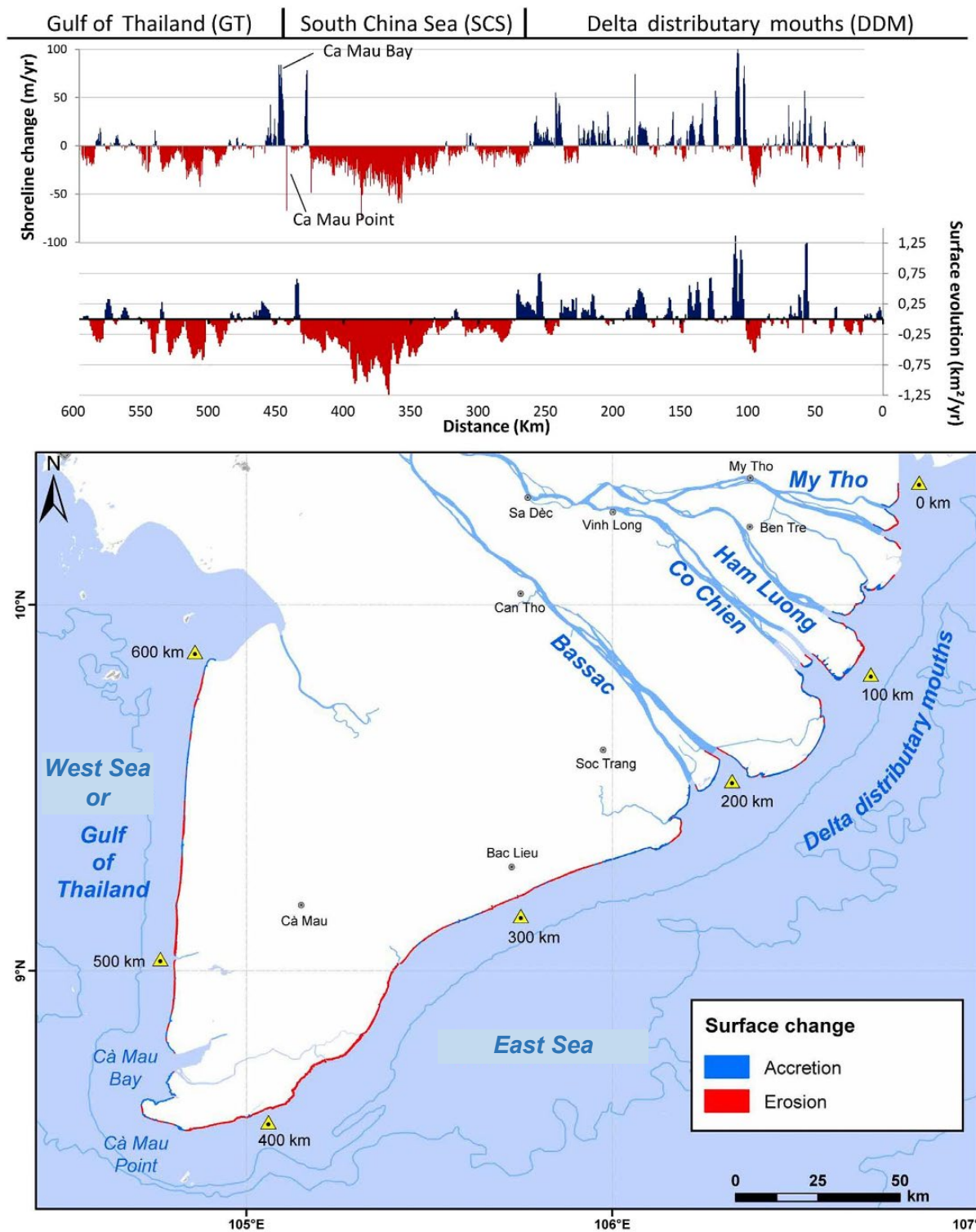
### **Coastal erosion**

Various studies on the coastal erosion of the Mekong Delta have been published. In the following, the work by Anthony et al. (2015), Besset et al. (2016, 2019), Li et al. (2017) and Phan (2020) is summarised. Figures 74 and 75 (below) present the shoreline development in the period 2003 – 2011/2012 as elaborated by Antony et al. (2015), based on satellite images, and by Li et al. (2017). The two figures are qualitatively similar, but the patterns differ in detail. However, the general conclusion can be drawn that shorelines which were still accreting about four decades ago, accrete less in recent years, and shorelines eroding four decades ago, erode more. These patterns are generally attributed to:

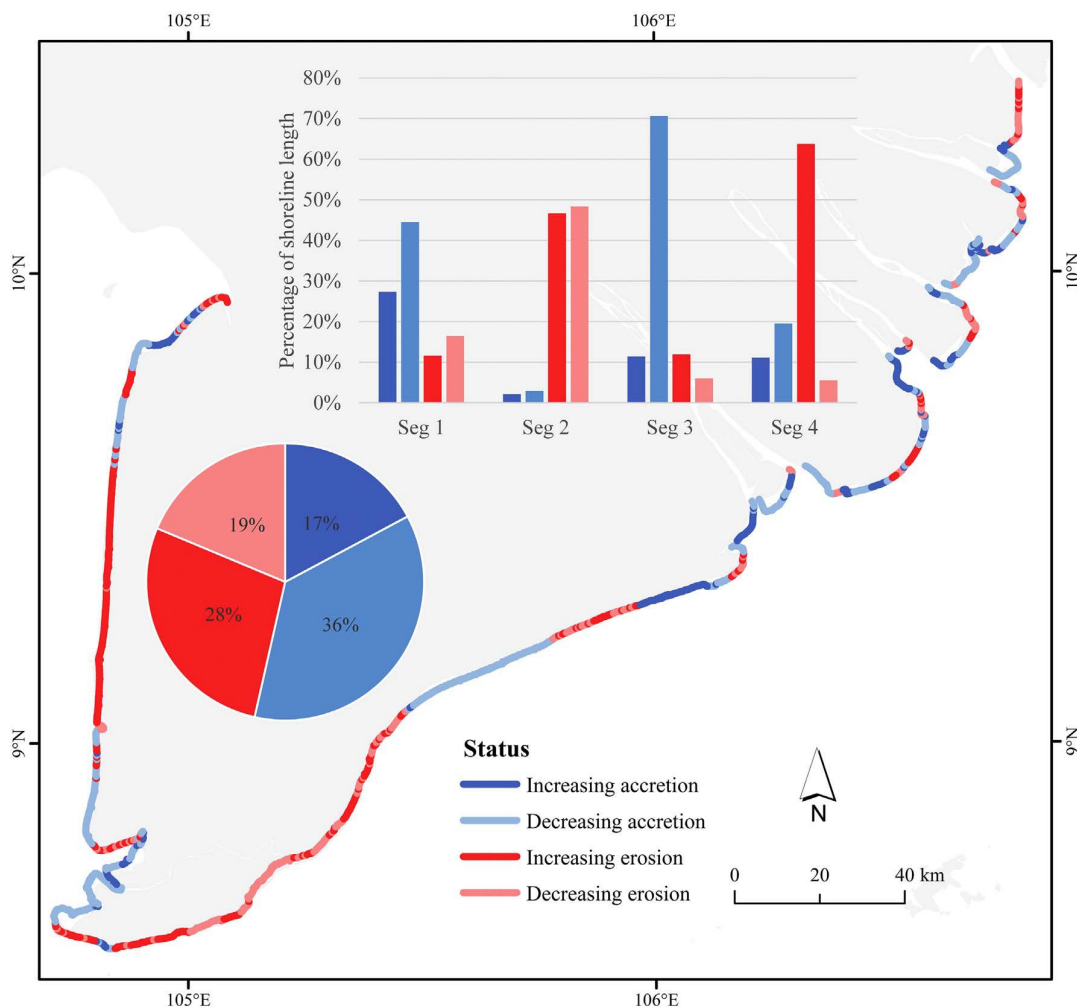
1. reductions in sediment input, both sand and fine sediment,
2. subsidence,
3. cutting mangroves.

We will argue in Section 4.3.4 that the current practice of coastal defence also has unfavourable effects on coastal accretion.





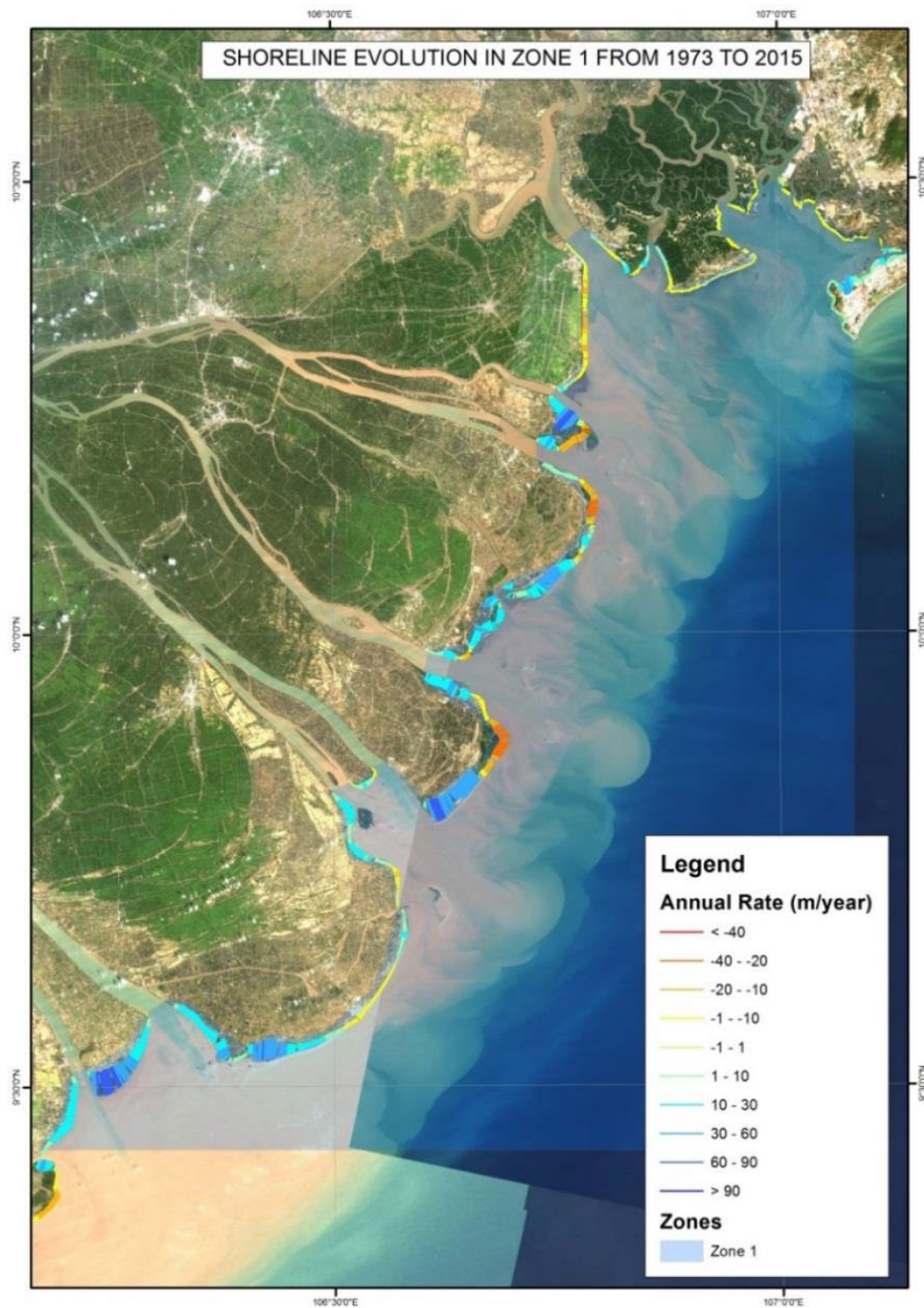
**Figure 74.** Coastal evolution between 2003 and 2011/2012 (after Anthony et al., 2015).



**Figure 75.** Comparison of shoreline evolution between 1973-1979 and 2010-2015, after Li et al. (2017). Segment 1: estuarine coast up to halfway Bac Lieu, Segment 2: Ca Mau east coast, Segment 3: Ca Mau west coast, Segment 4: Gulf of Thailand coast.

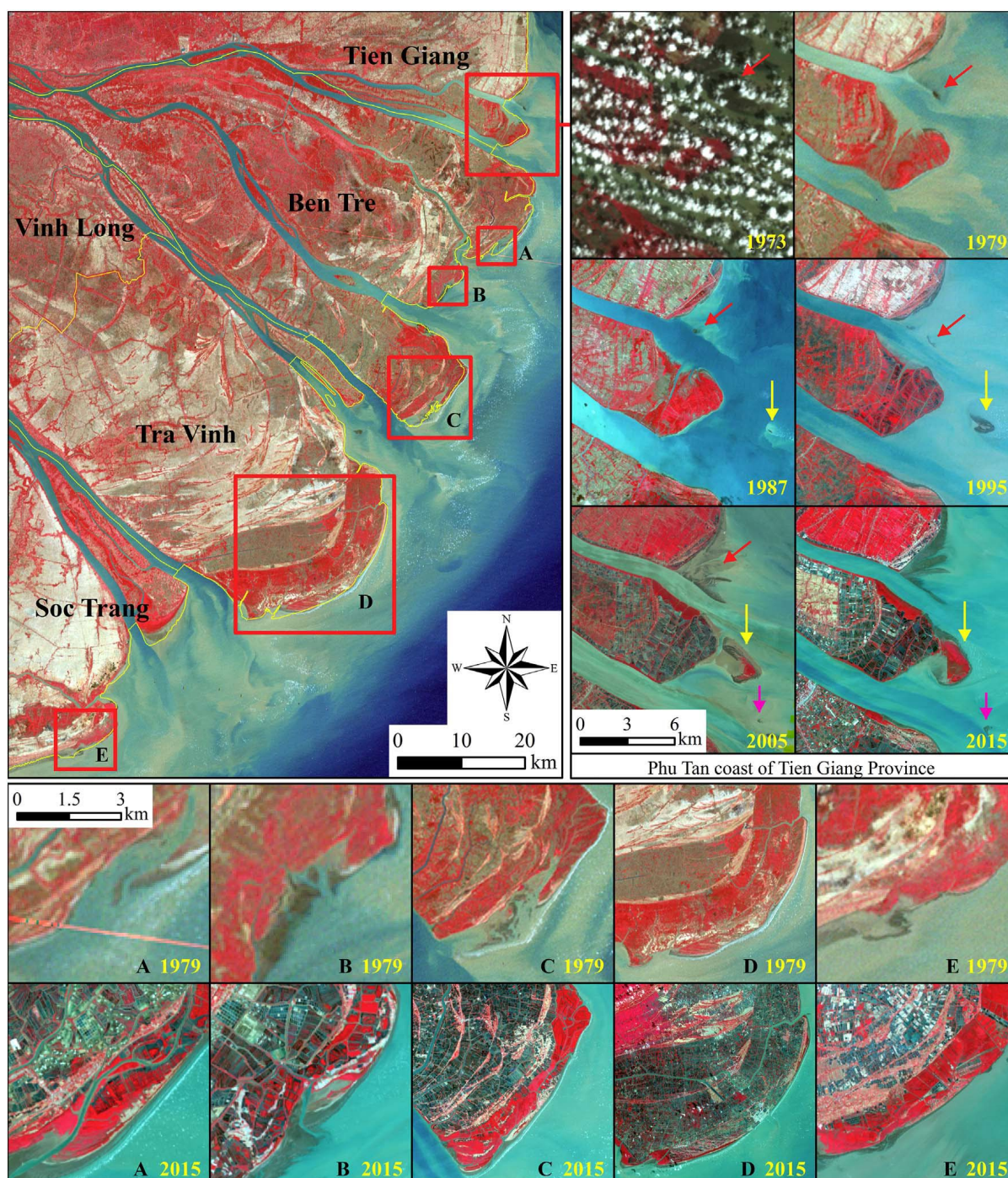
Figure 76 depicts the evolution of the shoreline around the Mekong estuary between 1973 and 2015 (Phan, 2020), showing that the land-building processes are still active. This is further elaborated in detail by Li et al. (2017) in Figure 77, showing how most of the islands in the estuary build out seawards. However, inspection of Google Earth satellite images show that large parts of newly formed land are rapidly converted into aquaculture ponds or agriculture fields, protected by bunds from the sea.

These analyses are important for the understanding of the large-scale dynamics of the Mekong Delta, and for correlating these dynamics to the large-scale impact of dams, sand mining and subsidence. However, for designing interventions to rehabilitate coastal forests, more detailed analyses of the coastal system at much smaller scale are necessary, i.e. at the scale of several 10s to a few 100s of metres.



**Figure 76.** Shoreline evolution around the Mekong delta between 1973 and 2015 (after Phan, 2020).





**Figure 77.** The geomorphological characteristics in the estuarine area of the Mekong River. Asymmetric progradation and discontinuous shoreline shifts can be clearly observed from the satellite images. All images are NIR-Red–Green false colour composites. Vegetation is shown in red, mangroves are typically along the coast, and aquaculture is landward and is shown in a dark colour.



### **Summary of Mekong Delta geomorphology and its developments**

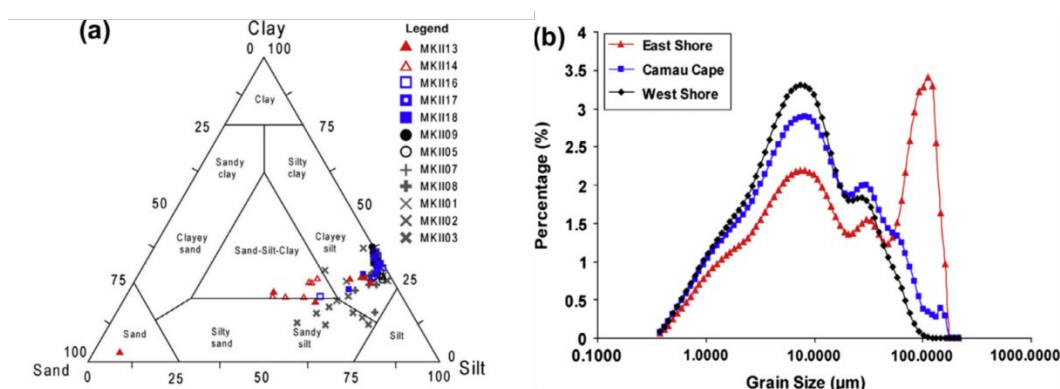
- The Mekong Delta consists of a shallow clinoform (sloping surface composed of deposited materials), with a 10 – 30 km wide foreshore at depth of 5 – 10 m;
- The delta is old, some parts over 10,000 years, formed from Mekong River deposits, which are highly stratified;
- The bed surface of the coastal waters consists of fine sand and mud;
- The Mekong Delta has been cultivated largely, with agriculture fields and fishponds close to the shoreline;
- The Mekong Delta subsides rapidly by natural compaction, groundwater extraction and/or tectonic processes, at some places at rates of 4 cm/year;
- The many dams and reservoirs constructed in the Mekong River catchment have reduced natural sediment loads in the river by 60 – 70%;
- Sand mining in the lower parts of the river (i.e. downstream of the dams) reduces sediment loads further, causing bank erosion – there is anecdotal information that the river now imports sand from the coastal zone;
- Fine sediment is also trapped in the lower estuary in a web of drainage channels, and when inundating/irrigating agricultural fields (rice fields);
- Large parts of the coast are eroding, erosion rates have increased, and accretion rates have decreased during the last decade, attributed to a decrease in sediment input by the river, subsidence and cutting of mangroves.

### **4.3.3 Large-scale coastal sediment dynamics**

This section summarises the natural processes driving the transport of fine sediment along the coast of the Mekong Delta at the scale of this delta. In the section, the interaction of these processes with the present coastal configurations are discussed.

#### **Sediment composition**

Section 4.3.1 summarises the composition of the bed sediment on the clinoform. The grain size distribution of bed samples was further elaborated by Xue et al. (2014), see Figure 78 below. The sandier samples were taken along the east coast. The fine sediment has a mean diameter of about 10  $\mu\text{m}$ . Though the northern part of the east coast was not sampled, it may be presumed that the mean diameter of the fine sediment here also measures about 10  $\mu\text{m}$ . The sand content is expected to increase towards the Mekong estuary, as the rivers bring the sand to the foreshore.



**Figure 78.** Grain size distribution of sediment along west coast and east coast of Ca Mau and Bac Lieu (Xue et al., 2014).

Illite is the abundant mineral of the fine sediment, which also contain fractions of kaolinite and smectites (Xue et al. (2014). It is therefore expected that the fine sediments are well flocculated, and effective particle sizes of the fine sediment are much larger than the 10  $\mu\text{m}$  of the Figure above, as reported by Wolanski et al (1996), who found floc sizes of about 40  $\mu\text{m}$  and larger.

### Estuarine processes

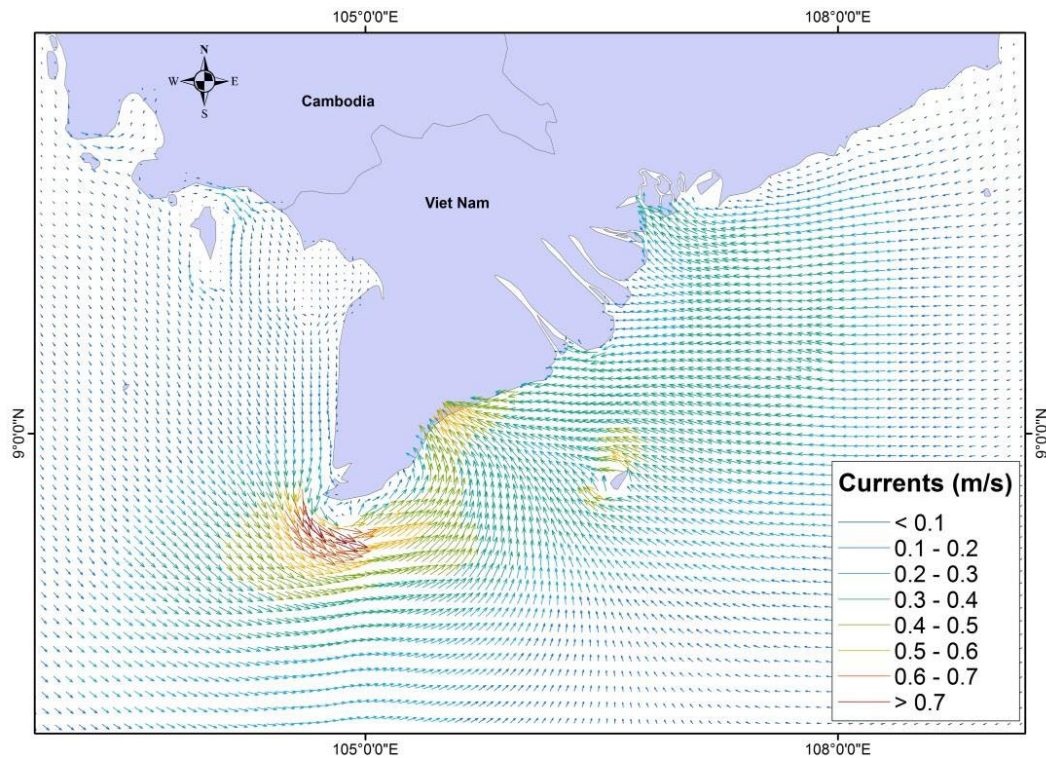
As discussed in Section 4.2.3, fresh Mekong River water entering the East Sea<sup>2</sup> is dispersed rapidly, as there are no processes which keep the fresh water close to the coast. The effects of salinity-induced gravitational circulation<sup>3)</sup> are expected to be important only along the coast of the estuarine provinces Tien Giang, Ben Tre, Tra Vinh and the northern part of Soc Trang, while small along most the coastline of the provinces Bac Lieu and the southern part of Soc Trang (Thanh et al., 2017; Marchesiello et al., 2019). Of course, also within the estuary, gravitational circulation is one of the dominant mechanisms, responsible for the formation of the estuarine turbidity maximum in the river, e.g. Ha et al. (2018).

### Longshore sediment transport mechanisms

The longshore sediment transport mechanisms can be best understood from the numerical model studies by Marchesiello et al. (2019) and Phan (2020; see also Phan et al., 2019, 2020). With these model studies the individual effects of tide, waves (stirring and currents) and wind-driven currents were studied. Though the results of these modelling studies differ in detail, the general picture is consistent. Instantaneous current velocities are moderate and do not exceed 0.5 m/s, generally a few dm/s only. Owing to the small depth over the clinoform, the currents have a profound onshore component during rising water, as depicted in Figure 79. In the absence of gravitational current effects, this is the main agent bringing (fine) sediment to the coast and therefore crucial for the coastal sediment balance.

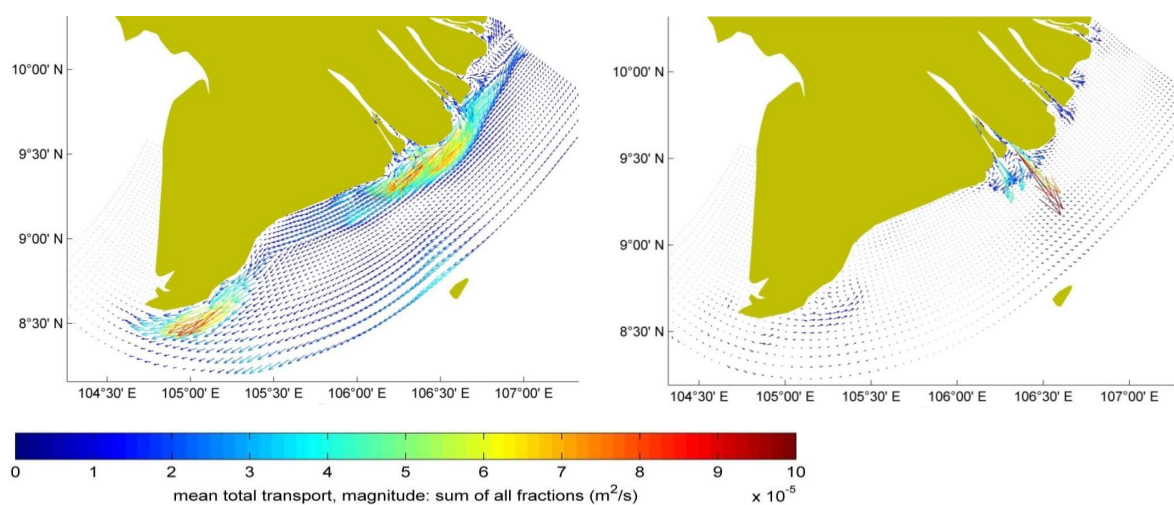
<sup>2</sup> In international literature the sea east of Vietnam is referred to as South China Sea, in Vietnam it is East Sea

<sup>3)</sup> gravitational circulation induces a residual near-bed onshore current, induced by salinity gradients perpendicular to the coast – this is generally a powerful mechanism to transport fine sediment, and keep it close to the coast.



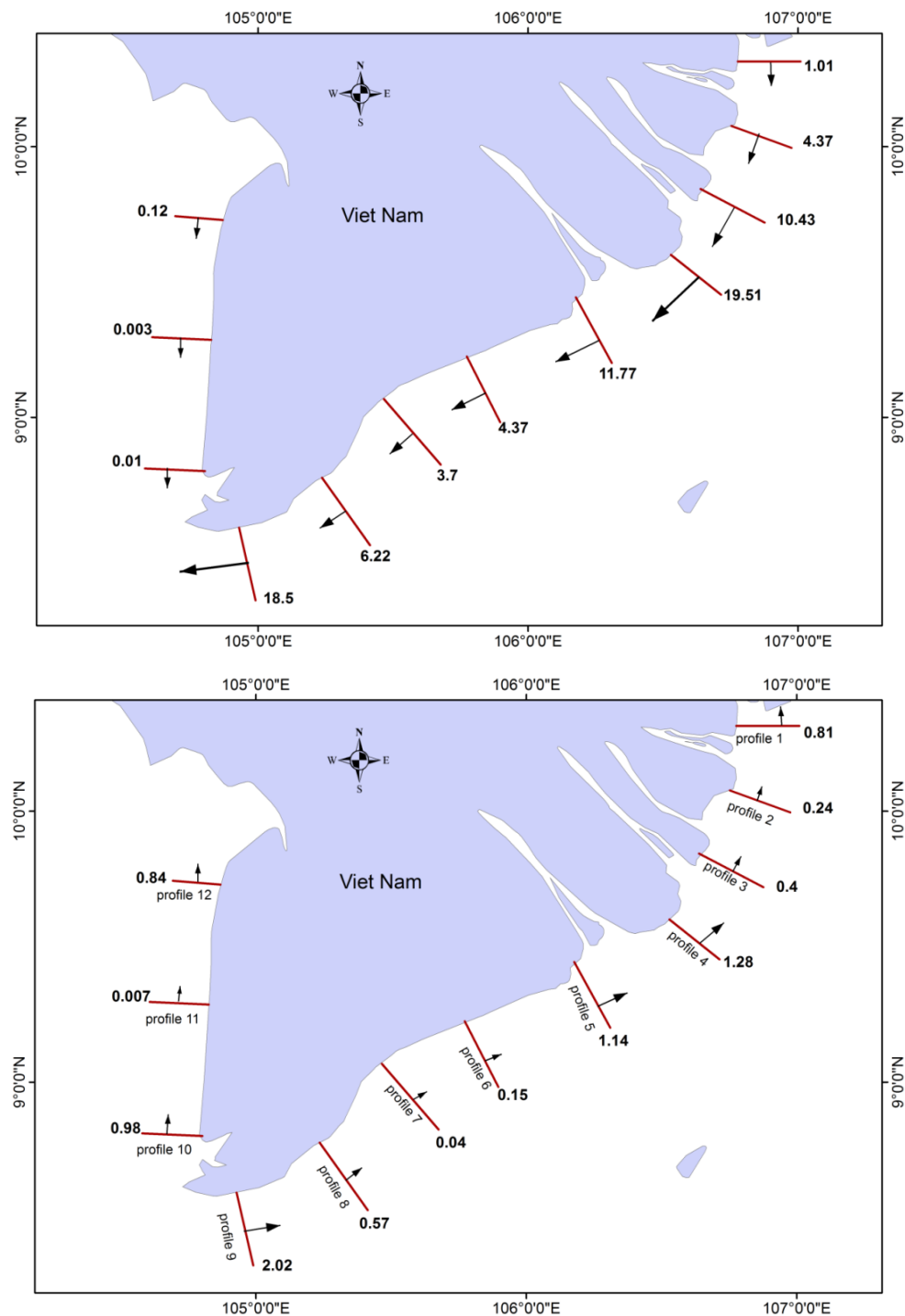
**Figure 79.** Computed instantaneous current pattern during rising tide (Phan, 2020).

As tidal currents are weak, the monsoon winds and waves have a large effect on the current directions and on the direction of the residual currents and sediment transport. As discussed in Section 4.1.5, residual currents are weak, a few cm/s at most. Figure 80 shows computed residual sediment transport patterns in winter (NE-monsoon) and summer (SW-monsoon). The southward sediment flux in winter is much larger than in summer, which is mainly to be attributed to the higher waves in winter, which stir up more sediment, in combination with larger wave- and wind-driven currents.



**Figure 80.** Computed residual sediment transport during winter (NE-monsoon, left panel) and during summer (SW-monsoon, right panel) (Phan, 2020).

It is of course this seasonal pattern which has built the Mekong Delta over time. This is further illustrated in Figure 81, showing the computed residual longshore sediment fluxes in winter and summer. In the estuarine section, the fluxes increase in southward direction, as sediment is supplied by the river. Then the fluxes decrease, indicating sedimentation along the coast. In the Bac Lieu foreshore, the flux increases again, indicating coastal erosion, as observed.



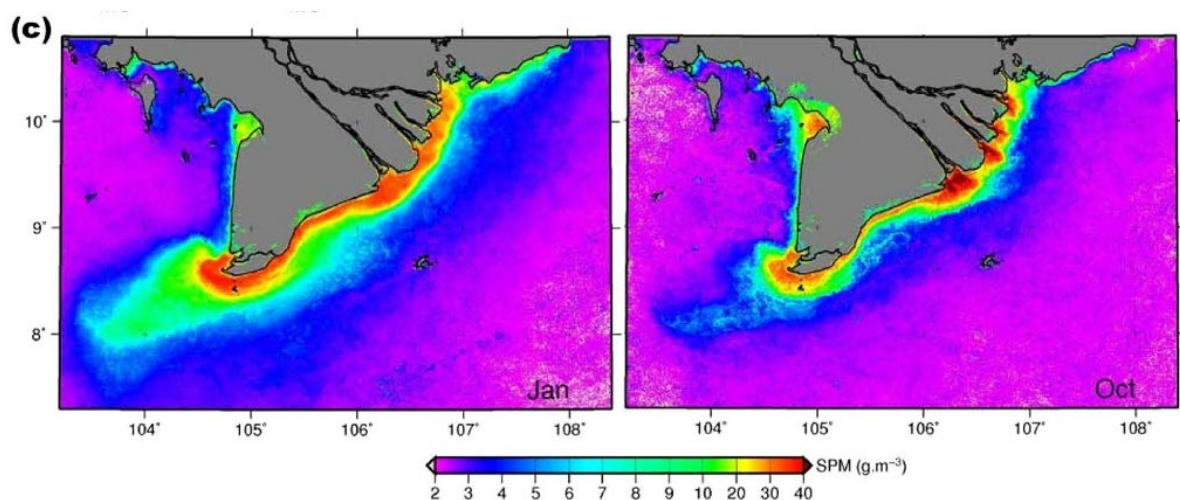
**Figure 81.** Computed residual sediment flux during winter (NE-monsoon, upper panel) and during summer (SE-monsoon, lower panel) in Mton/yr (after Phan, 2020).



## Suspended sediment concentrations

Though not resolved in the numerical model simulations, the spatial and temporal variability in the actual fluxes must be large, as also affected by the local conditions of the shoreline. In particular this is the case for the onshore (and offshore) sediment fluxes. For designing interventions for reforestation, the local hydrodynamics must be analysed at small spatial scales. The local onshore sediment flux can then be assessed in conjunction with the suspended sediment concentration, which generally exhibits much smaller spatial variations, owing to the fineness of the sediment. Temporal variations in suspended sediment concentration may be large, though, as affected strongly by waves.

Figure 82 depicts near-surface suspended sediment concentrations in January and in October derived from satellite images by Anthony et al. (2015). As argued above, suspended sediment concentrations along a major part of the MD shoreline in winter are larger than in summer. However, near the mouth of the estuary, an opposite trend is observed, reflecting the larger discharges of sediment-laden river flows in summer.



**Figure 82.** Observed near-surface suspended sediment concentrations in January (NE-monsoon, left panel) and in October (SW-monsoon, right panel), after Anthony et al. (2015).

Near-surface suspended sediment concentrations do not exceed about 40 mg/L. However, satellite images during storm are generally not available due to cloud cover. Hence, the values in Figure 82 must be considered as good-weather values. Moreover, near-bed suspended sediment concentrations are larger than near-surface values. Hence, the actual suspended sediment concentrations are expected to be larger, of the order of (a few) 100 mg/L, relevant for coastal accretion.

Note that the numerical modelling studies by Phan (2020) and Marchesiello et al. (2019) yield similar suspended sediment concentrations. However, no information could be found on vertical sediment concentration gradients, nor on values under storm conditions.

#### Summary of large-scale Mekong Delta coastal sediment dynamics

- The mud fraction of the sediment is very fine ( $D_{50} \approx 10 \mu\text{m}$ ), but the mud is flocculated ( $D_{50} > 40 \mu\text{m}$ ), as illite is a major mineral component;
- Longshore sediment transport is driven by the tide, winds and waves, and their interaction;
- Longshore sediment transport exhibits a profound seasonal variation, with the larger southward directed flux during the winter NE-monsoon and the smaller northward flux during the summer SW-monsoon;
- Near-surface suspended sediment concentrations measure about 40 mg/L – characteristic values are like of the order of (a few) 100 mg/L;
- Onshore, coast-forming sediment transport along the provinces Bac Lieu and the southern part of Soc Trang is driven mainly by tidal filling, i.e. onshore fluxes during rising tide;
- Onshore, coast-forming sediment transport along the provinces Tien Giang, Ben Tre, Tra Vinh and the northern part of Soc Trang is driven also by gravitational circulation, induced by cross-shore salinity gradients in/around the estuary.

#### 4.3.4 Small-scale coastal sediment dynamics

At smaller scales, coastal sediment dynamics are strongly governed by the morphological state of the coastline. This is in particular the case along the provinces Bac Lieu and the southern part of Soc Trang, as the local onshore sediment transport is mainly driven by (local) tidal filling. Possibly this explains the larger erosion rates along these provinces. Along the provinces Tien Giang, Ben Tre, Tra Vinh and the northern part of Soc Trang, the larger scale process of gravitational circulation is important as well, as a results of which the coastal development along these provinces is less affected by the local coastal morphology.

Assessment of these local processes is key in designing reforestation measures. This section therefore describes how the dynamics of the natural, mangrove-covered Mekong Delta coast are affected by the effects of coastal infrastructure erected over time along this coast.

All mangrove-mud coasts are dynamic. The time scale of these dynamics is determined by local conditions, such as the hydrodynamics (tide and waves), the availability of sediment and propagules, the age of the mangrove forest, etc. Currently, we have no data on the time scales of the mangrove-mud coastal dynamics in Vietnam. These dynamics become manifest through retreat and accretion of the shoreline and through development of density and biodiversity of the forest. Coastal retreat is attributed to wind and waves. Waves erode (fine) sediment from the foreshore and from in between the mangrove stands. Foreshore erosion may lead to cliff erosion, which is generally accompanied by large erosion rates (Figure 83).



**Figure 83.** Cliff erosion in Suriname.

As the roots of mangrove trees penetrate the soil only by about 0.5 m, 10 cm of erosion of soil from in between the roots would reduce mangrove anchoring strengths by 20%. The trees then become vulnerable to wind stresses. This is a major cause of loss of mangrove trees along the pristine mangrove-mud coast of e.g. Suriname (Figure 84). Thus, wind and waves are destructive forces for a mangrove forest.

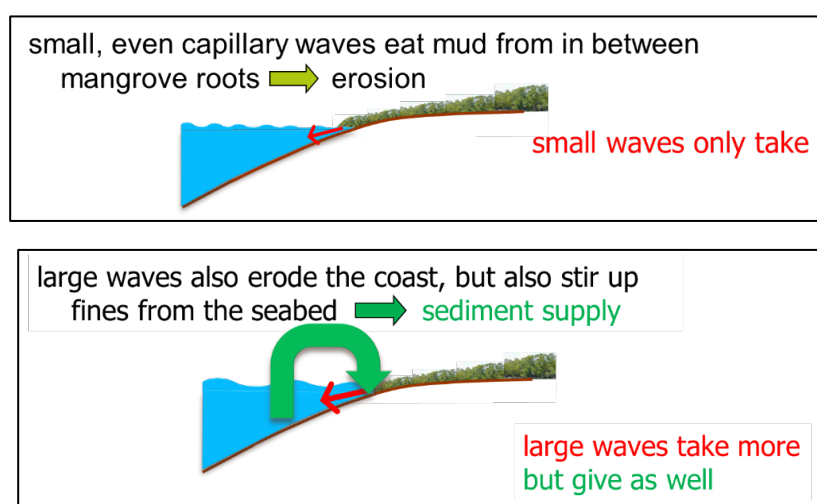


**Figure 84.** Loss of mangrove trees in Suriname by wind stresses.

Mangrove-coasts accrete through sedimentation on the intertidal mudflats in front of the trees, forming new mangrove habitat. The lower fringe of mangrove habitat is found between mean sea level (MSL) and mean high water (MHW), depending on the hydrodynamic conditions, while the upper fringe is found at high-high water spring (HHWS). At higher elevations, mangroves are outcompeted by other species. When abundant propagules are available, mangrove colonisation of the mudflats can be fast.

The sediment that is deposited on mudflats, forming mangrove habitat, is mobilised by waves from the foreshore and carried by the tide onshore. Such mobilisation is only possible by the larger waves; small waves cannot stir the seabed but are able to eat sediment from in between the mangrove roots. Note that on short time-scales, the direct contribution of

sediment supply from the river is small – sediment generally first settles on the foreshore and then returns onshore. This leads to the somewhat counter-intuitive situation that small waves only erode the coast, while larger waves, of course, erode more sediment from the coastline, but also feed the coast with new sediment. In other words: big waves give and take, while small waves only take, as sketched in Figure 85. Thus, tide and larger waves form the constructive forces for a mangrove forest.



**Figure 85.** Schematic diagram of the role of waves in coastal accretion.

Mangroves have important ecosystem services (Rossi et al., 2013). From a coastal management point of view, mangroves provide two services:

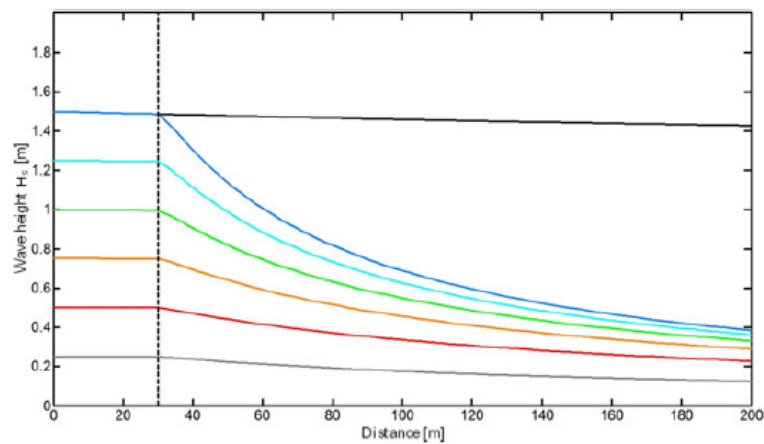
1. Mangrove-mud coasts can keep pace with (moderate) sea level rise,
2. Mangroves fringes dissipate wave energy, thus protecting the hinterland.

Within a mangrove forest, flow velocities are very small, and all sediment entering a forest during high water or via the various drainage channels will be trapped. Hence, if sufficient sediment is available, mangrove forests can keep pace with relative sea level rise, i.e. the rise of the mean sea level in response to climate change superposed on subsidence because of consolidation, ground water extraction, tectonic processes, etc. Mangroves can accommodate up to 7 cm relative sea level rise per year (Saintilan et al., 2020), but their tolerance may be compromised under conditions of sediment shortage or subsidence (Ward et al., 2016). At higher rates of sea level rise, mangroves will drown or suffocate from too much sedimentation. Mangroves thus can engineer their own environment and are therefore referred to as “eco-engineers” (Jones et al., 2014).

Mangroves can damp waves through dissipation of turbulent kinetic energy in the wakes of their stems, roots and branches. The rate of damping depends strongly on the configuration of the mangrove forest (species, age of trees, density, height) and of the abiotic conditions (bed slope, water levels, wave height and period). Damping rates are highly non-linear, as exemplified in Figure 86, with the larger damping rates in the first part of the mangrove forest. McIvor et al. (2012, 2013) carried out an extensive literature study and concluded that moderate wind and swell waves can be damped by 50 – 100% over the first 500 m of a mangrove forest. During storm surges, damping rates are much smaller, with values between



5 and 50 cm per kilometre mangrove forest. Data on higher waves and during storm surge are rare, and no definite conclusions can be drawn at present.



**Figure 86.** Numerical model results by Van Wesenbeeck et al. (2017) illustrating the attenuation of waves through emerged vegetation, starting at 30 m. Significant wave heights are indicated on the y-axis and distance within the vegetation is depicted on the x-axis. The black line represents the reference situation without vegetation. The different colors indicate different initial (offshore) wave heights.



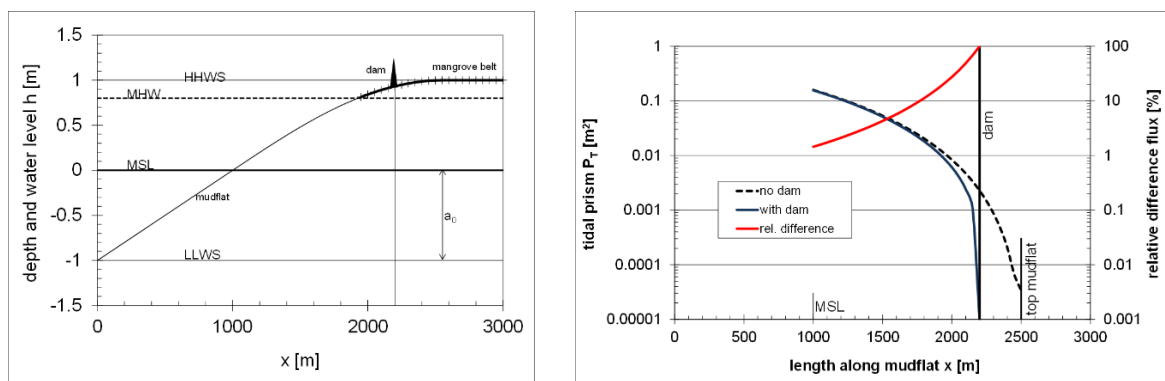
**Figure 87.** Example of a concrete seawall in Tra Vinh province (upper panel) and of a detached breakwater at Nha Mat (Bac Lieu province, lower panel).

The natural processes above are disturbed by infrastructure along the shoreline and in the foreshore, such as reflecting seawalls and detached breakwaters, examples of which along the Vietnamese coastline are shown in Figure 87 (above). This disturbance is due to:

1. Reduction in coastal accommodation space for fine sediment, reducing sedimentation rates,
2. Increasing wave heights by wave reflection, increasing erosion rates.

Reduced sedimentation rates and increased erosion rates modify the (intertidal) mudflat profile, thereby affecting the mangrove habitat. In extreme cases, reforestation becomes impossible. Below, these mechanisms are further elaborated at a conceptual level.

Intertidal mudflats are characterised by gentle slopes, thus exhibiting large tidal volumes. Onshore velocities during rising water are therefore relatively large, bringing sediment onshore. Such intertidal mudflats have therefore a convex-up cross-shore profile, as observed in nature, but which also follows from theoretical and numerical modelling studies. Figure 88 quantifies the reduction in intertidal area, thus accommodation space by the erection of a dam at  $x = 2200$  m on a convex-up tidal flat, about 10 cm below high-high water spring (HHWS), left panel. The right panel shows the reduction in tidal prism, i.e. the reduction in the volume of water that could be accommodated beyond the dam. Close to the dam, this reduction is of course 100%. However, even at 400 m further seaward, which is more or less the location of mean high water (MHW) this reduction is still 10%. This implies that the mangrove habitat receives 10% less sediment-laden water, thus at least 10% less sediment. Winterwerp et al. (2005, 2013) reason that this reduction may flip the evolution of a tidal mudflat from accreting to erosive, and its cross-shore profile from convex-up to concave-up.



**Figure 88.** Reduction in tidal volume and accommodation space for sedimentation in response to erection of seawall or fishpond bund (from Winterwerp et al., 2013).

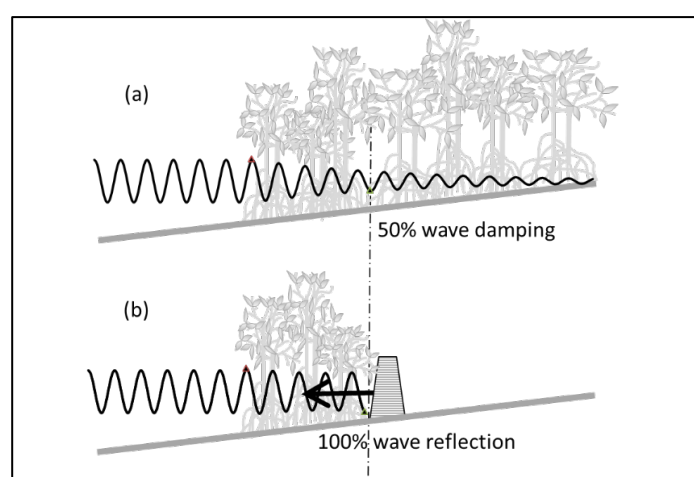
The dam in Figure 88 not only reduces the accommodation space on the intertidal, but also induces wave reflection. The degree of reflection is given by the reflection coefficient  $K_R$ .  $K_R = 0.5$  implies that 50% of the wave height is reflected. The reflected wave interacts with the incoming wave, and a standing wave is formed with wave height 1.5 times that of the incoming wave (for  $K_R = 0.5$ ). As the bed shear stress scales with wave height squared, this example would imply an increase in bed shear stress by more than a factor two, generally inducing scour in front of the dam. This is of course the reason that bed protection is often

required in front of coastal structures. Note that  $K_R$  is strongly dependent on the shape and finishing of the dams/seawalls, as summarised in the Table below:

**Table:** Wave reflection coefficients (Goda, 1985).

| Structure type                                             | Reflection coefficient ( $K_R$ ) |
|------------------------------------------------------------|----------------------------------|
| Vertical wall, crown above water                           | 0.7-1.0                          |
| Vertical wall, submerged crown                             | 0.5-0.7                          |
| Revetment consisting of rubble stones                      | 0.3-0.6                          |
| Revetment consisting of energy dissipating concrete blocks | 0.3-0.5                          |
| Permeable (energy dissipating) vertical structure          | 0.3-0.8                          |
| Natural beach                                              | 0.05-0.2                         |

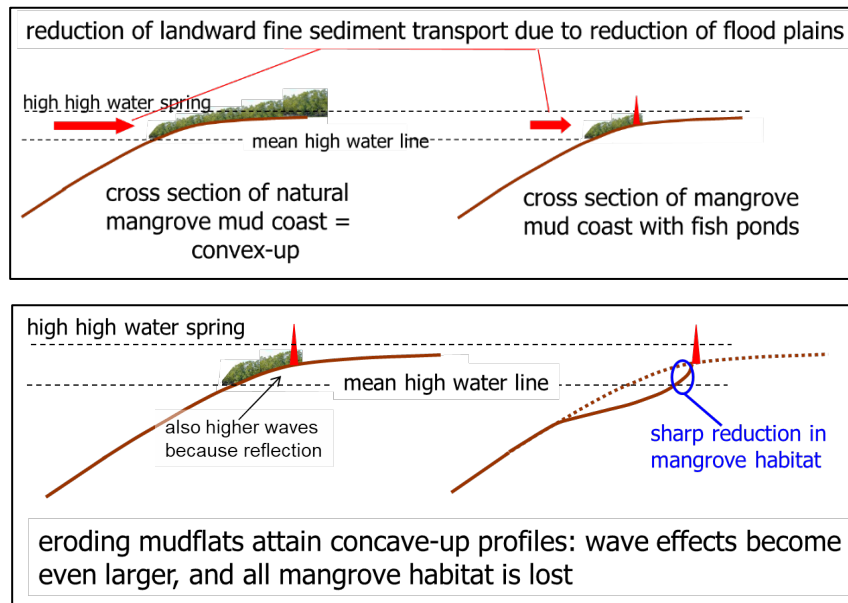
Figure 89 sketches the potential effect of erecting a dam/bund in a mangrove forest. The upper panel depicts a wide mangrove forest in which all wave energy is damped, hence likely wider that 500 – 1000 m (see above). Suppose that a dam is erected at a location where 50% of the incoming wave height is damped, as in the lower panel of this Figure. Assuming  $K_R = 1$ , this implies that at the dam, the wave height is doubled, thus equal to that of the incoming wave. The reflected wave, traveling in offshore direction is damped as well, of course. But while traveling offshore, it interacts with the incoming wave, which, for this particular example, results in a constant wave height through the mangrove forest. The trees in the back are now not only attacked by incoming waves, but also by their reflections. The situation worsens if a dam would be erected closer to the waterline. Then wave stresses on the mangroves in the back may even exceed those at the waterline. This explains why vegetation in front of a seawall often starts to collapse in the back, near that seawall.



**Figure 89.** Schematic diagram of increasing wave heights by wave reflection (from Winterwerp et al., 2020).

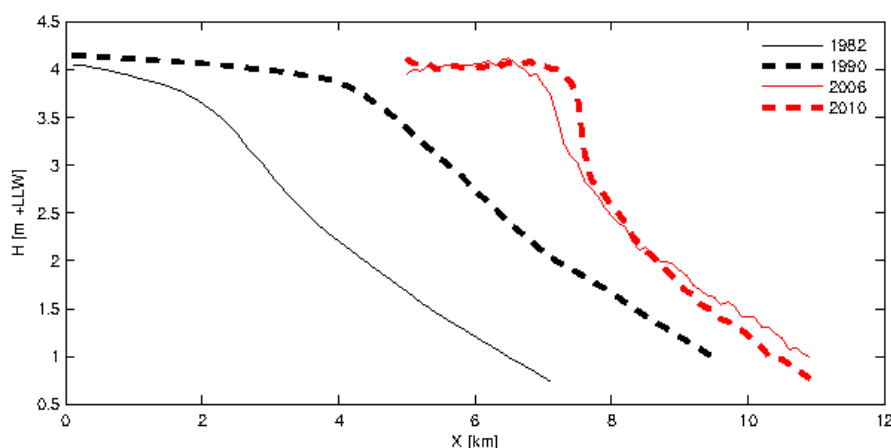
The combination of a reduction in accommodation space and an increase in wave height in response to the erection of a dam on the intertidal is sketched in Figure 90. Constructive processes (sedimentation) decrease and destructive processes (erosion) increase. The cross-shore profile of the intertidal evolves to concave-up, reducing the mangrove habitat further,

and further increases wave height and wave-induced stresses because of larger water depths. This positive feed-back process worsens the situation even further.



**Figure 90.** Schematic diagram of impact seawall on mangrove habitat. Upper panel: reduction in accommodation space for sedimentation; lower panel: increase in wave-induced erosion. Over time, the mudflat cross-shore profile evolves from convex-up to concave-up (lower right figure)

This sequence of events explains the observed transition of the cross-shore mudflat profile of Chong Ming Island (China), depicted in Figure 91. Where the mudflat accreted rapidly by 3 km between 1982 and 1999, in 2010 accretion of the supratidal stopped, while the lower intertidal even started to erode. This response is attributed to an aggressive policy of land reclamation on the island, erecting massive dykes (Van Maren et al., 2013).

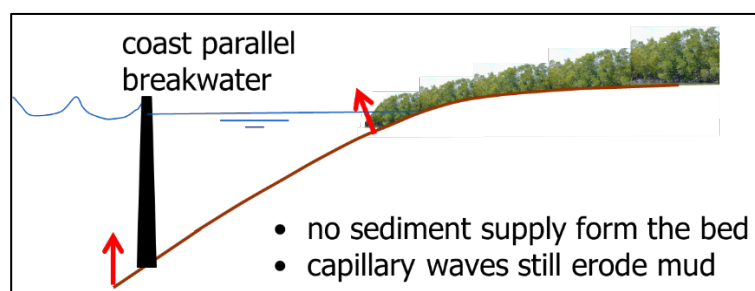


**Figure 91.** Measured evolution of cross-shore profile of Chong Ming Island (Yangtze estuary, China);  $H = 0$  reflects low-low water spring (Van Maren et al., 2013).

The constructive processes for mudflat building are further reduced by the erection of coast-parallel detached breakwaters (see Figure 92). These breakwaters block all waves behind, but also block the onshore transport of sediment-laden water. Indeed, erosive forces decrease, as the larger sea-borne waves can no longer approach the shore, but the smaller, locally generated waves still can. Thus, erosive forces decrease, but do not vanish entirely, while the



constructive processes do vanish entirely – the remaining small erosion rates are no longer compensated by newly deposited sediment. As a result, the mudflat, i.e. the mangrove habitat continues to erode, though at a slower pace.



**Figure 92.** Schematic diagram of impact of coast-parallel detached breakwater on onshore sediment supply.

As the Mekong Delta hinterland is shallow, sea dykes are necessary at many places to protect people and their possessions. This necessity is likely to increase with climate change. This development was recognised by the World Bank (2017), who advocated the concept of green-grey coastal infrastructure. The grey part consists of the dykes to protect the hinterland against flooding. The green part consists of a (mangrove) greenbelt in front of the dyke (Figure 93).



**Figure 93.** Schematic diagram of green-grey coastal infrastructure (adjusted from World Bank, 2017).

Damping of the waves offers potential cost reduction of sea-dike construction: lighter or no hard dike revetment, a lower and thus smaller dike.

This greenbelt has three functions:

1. It damps waves, reducing the mean loads on the dyke, which therefore require potentially less dike revetment and less maintenance, thus saving costs,
2. By damping waves, it reduces wave run-up and overtopping, which allows for lower crest heights of the dyke, thus a lighter construction, saving costs,
3. It restores part of the mangrove ecosystem services, amongst which carbon sequestration, water purification, habitat and nursery of fish and the ability to grow with sea level rise.

Further to the discussion above, these functions can only be fulfilled if the greenbelt is wide enough. As we have no data for Vietnam, a first estimate would be that greenbelt widths of at least 500 m are required.

### **Summary of small-scale Mekong Delta coastal sediment dynamics**

- Mangroves damp waves (50 – 100% of the height of moderate waves over 500 m of forest) and can grow with moderate (relative) sea level rise;
- Mangrove-mud coasts are dynamic; the time scale of the dynamics depends on the mangrove configuration and abiotic conditions;
- A mangrove-mud coast is built up by the interaction of tide and larger waves; it is destructed by larger and smaller waves and by wind stresses;
- A sea dyke too close to the shoreline has the following impacts:
  - sediment accommodation space reduces, thus sedimentation rates reduce, and mudflat formation is slowed;
  - wave reflection on the sea dyke can increase erosive stresses and attack mangroves from the back;
  - the stable convex-up cross-shore mudflat profile may become concave-up, setting in motion a positive feed-back, enhancing erosion rates further;
- Coast-parallel detached breakwaters slow down erosion rates, but do not stop erosion as all constructive onshore sediment supply is killed;
- Green-grey coastal infrastructure may be a compromise to protect low-lying hinterland, while benefiting from mangrove ecosystem services.

## 4.4 Aquaculture & other land-use in the Mekong Delta

### 4.4.1 History

The Mekong Delta has always been a deeply woven estuary with intricate bodies of marshlands, sand ridges, creeks and rivers. After the construction of a network of waterways from 1820 onward, human settlement expanded beyond the sandy heights parallel to the coast, where the Khmer fishers and traders had settled already (Cooke & Li, 2004).

Most farmers and fishers living in and around the Mekong Delta have to live with an annual monsoon flood that lasts between 2 and 6 months, and may inundate the land by up to 3 m, depending on the particular year and location (Xuan & Matsui, 1998). After the introduction of steam-powered dredging in the 1880s, the total area put under rice cultivation rose to over 2 million hectares, and the delta's population rose from about half a million in 1860 to over 4 million in 1930 [Inspection des Travaux Publics, 1930, p20].

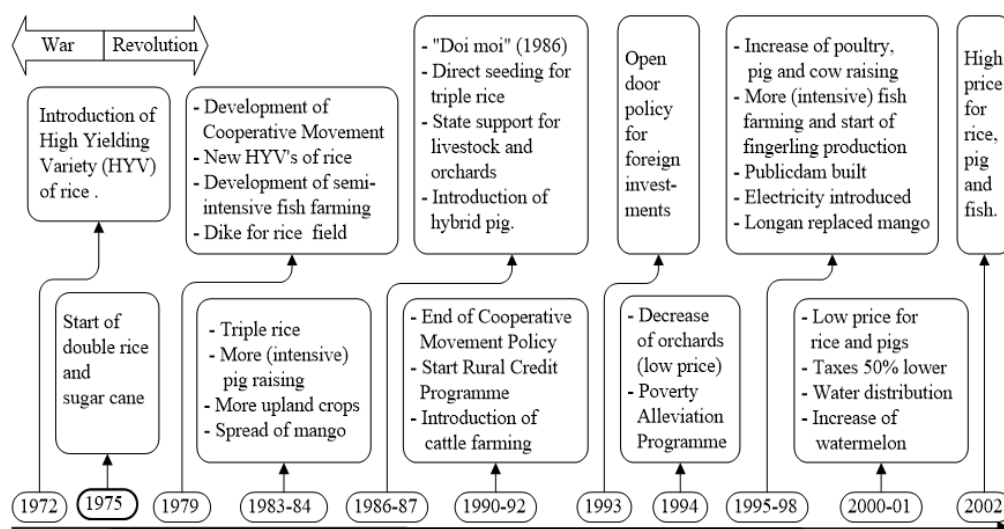
Gradually, the French colonial administration also improved road access. Waves of Thai, Khmer, Viet, Lao, Chinese, Malay and European immigrants intermixed (Cooke & Li, 2004) and constructed houses along the roads and the raised borders of the waterways, thus starting the characteristic linear settlements (Sanh et al., 1998). People fished, lumbered, traded, and cropped sugar palm and both floating and rainfed rice. In the fields with floating rice, farmers harvested fish and other aquatic organisms that recruited naturally with the inflowing flood water. Others created a homestead by digging a hole in which fish and shrimp recruited naturally, thus starting pond aquaculture. All that time rural life in and around the waterways remained dominated by the diurnal tides from the East Sea (Bosma et al. 2006a).

Due to decolonisation wars, few new canals were dug in the 1930s. Between 1945 and 1976 the rural population followed a survival strategy, but continued dredging of waterways. Before 1976 the regional Mekong Committee enabled delta-wide master-planning with internal support, and the reunified Vietnamese government engaged in land reclamation projects, creating polder landscapes and seasonal flooding zones. Then, during the first stages of the collectivisation period of state-controlled monoculture of rice for the cities, in zones with favorable agro-ecology, farmers developed various mixed cropping strategies by building bunds for trees and ponds for fish and shrimp in some parts of their rice fields. Thus, from systems producing mainly rice in monoculture, farmers shifted to a variety of rice-based integrated systems with many degrees of crop-fish-livestock integration and of market orientation (Phong, et al., 2007; Prein, 2007; Sanh et al., 1998).

### 4.4.2 Land tenure reform

After the Vietnamese government privatised agriculture in 1986 and issued the new land policy in 1992, the innovations above supported the shift from mainly self-sufficiency to dominantly market-orientation. Farmers intensified both the agriculture and aquaculture, and specialised more on specific products (Figure 94). Thus began the farmers' adoption of new water management practises, of rice varieties and rice technologies that allowed two to three crops per year, improved food security, decreased market price for rice, and land made

available for other uses. Farmers became less fixated on growing rice to ensure household food security (Bosma et al., 2006a).



**Figure 94.** Examples of trends in agriculture from 1972 – 2002 for Cai Bè, Tam Binh and O-Mon districts in the Mekong Delta (Bosma et al, 2006b).

After 1992, land tenure was individualised and the land, designated either for agriculture by awarding a ‘red certificate’ or a ‘green certificate’ for forestry or perennial crops. These certificates gave farmers the right to use the land as collateral and brought land use within a regulatory framework. Farmers with a green certificate for forest plots, on hills and in wetlands, were obligated to align their land use practises with regulations, e.g., maintaining permanent (tree) crops to protect land against erosion. Gradually, as the land market liberalised, land prices rose, and access to land became dependent on the market. Within 30 years, alongside an improved on-land infrastructure network, the Mekong Delta became one of the world’s most productive zones for rice and aquaculture, providing in the beginning, foreign currency for investments in economic diversification (see below).

Since 1986, the Vietnamese government has decentralised the cost of maintaining canals and other infrastructure to provincial governments and private landowners. This decision has challenged local authorities to find models that can sustain and improve the transport, irrigation and drainage systems. Moreover, to widen the consideration of water resources from the focus on irrigation, the government shifted the control over the waterways and irrigation from the single domain of the Ministry of Agriculture and Rural Development (MARD) to shared responsibility with the Ministry of Natural Resources and the Environment (MoNRE) (Molle & Hoanh, 2008).

#### 4.4.3 Crops, livestock, forest and aquaculture zones

The Mekong Delta divides into seven major agro-ecological zones (Sanh et al, 1998). In these zones, the Vietnam government policies are strongly supportive of extension, training and other support measures such as subsidised credits. These policies are guided by the agro-ecological opportunities and may follow bottom-up initiatives of farmers, development agents and scientists. Crops, livestock, forest and aquaculture production are designated in



specific regions, e.g.: fish and shrimp in the lowlands, and goats and cattle in the hilly districts of the Mekong Delta. Livestock–fish–crop systems and their specialisations are found mostly in the freshwater alluvial zone, and to a lesser extent in the hills and uplands zones where rain-fed agriculture and forestry predominate over irrigated cropping. Development of crops is a constraint in the two zones with strong acid-sulphate soils. Crops as well as livestock are hardly developed in the two zones with saline water or having high and persistent floods during monsoon. In those flood-prone zones in the upper-delta, farmers crop rice in the dry season and practice communal aquaculture in the flood season; while in the lower delta, they crop rice in the wet season when water salinity is low, and culture shrimp or fish in the dry season. In tidal forestry zones, rather than planting mangrove species, both private and public parties prefer to plant the afore-mentioned *Melaleuca* that has a high demand and can be harvested after 4 years instead of 12 years or more.

#### 4.4.4 Rice, shrimp and mangrove loss

During the colonial period and due to defoliations in the second Indo-China war, the cover of mangrove forests had been reduced to make space for rice (Luu, 2000). After independence, with support from several donors, the government implemented reforestation with accompanying resettlement schemes along the coast of the Mekong Delta. Then, the protected forests along the coast were distinguished in two zones: the 300m strict protection zone and the 300m semi-protected zone (Vien, 2008). However, due to population growth, weak control on (semi-) protected forests (Truong and Do, 2018) and poverty, the mangrove cover was reduced again to a little more than the protected national forests along the coast in favor of developing (bio-)industry.

In the coastal zone, the push to produce shrimp has led to a huge conversion of mangrove into shrimp farms between 1980 and 1995; e.g., in Bac Lieu and Ca Mau about 66,250 ha was converted (Buu & Phuong, 1999 cited by Tran et al., 2012a). In 2000, when the demand and price of rice decreased, the government allowed re-entry of seawater to the formerly salinity-controlled areas to increase shrimp farming (Kâkönen, 2008). Thus, within some years, farmers, thereafter, shifted about 200,000 ha from rice to shrimp and mud-crab culture in Ca Mau, Bac Lieu, Kien Giang and to a lesser extent in Tra Vinh provinces; while *Pangasius* catfish culture boomed in the upper and central provinces of An Giang, Dong Thap, Can Tho and Hau Giang (VNMC, 2003; Van Brakel et al., 2011). This shift from rice to shrimp drove the (near to) landless population, depending on labour demand, to other livelihoods, who among others, engaged in the collection of mud-crab, elongated goby fry, and juvenile clam and cockles in mangroves and on mudflats (Joffre and Schmitt, 2010). Alongside, local governments promoted horticulture with irrigation of groundwater on coastal sand ridges. In several provinces, the latter has been, along with demands for aquaculture and industry, the cause of massive evacuations due to the lack of drinking water for the population in 2020/21.

In the semi-protected coastal mangrove zone, the semi-public Forest Companies can engage farmers in contracts for mangrove-aquaculture (silvo-fisheries), mostly in plots of 0.5 to 2 ha where they have to maintain 50 to 70% mangrove cover, and respect various rules which often limit their options to manage the pond properly (Tran et al., 2014; Bosma et al. 2014).

Part of this mixed or integrated mangrove-aquaculture has been certified by a variety of labels to fetch higher prices at export markets, but the premium was either too low or didn't reach the farmers to make a difference (Tran et al, 2012a and 2012b). However, the average income from the shrimp component, 2,700 USD/ha/yr found by Truong and Do (2018) was almost double than the 1,400 USD/ha/yr found by Tran et al. (2014).

#### 4.4.5 Aquaculture earnings, risks, export and sustainability

Aquaculture in the Mekong Delta has a broad cultural background related to various immigrants practicing floating rice and cage cultures, and recruiting fish and others in the rice-fields and homestead ponds. To satisfy international and local consumers, farmers have cultured carp for a start, recruited catfish and local shrimp and collected mud crab and scallops. At present, they raise a wide range of species in brackish water for the local market to meet typical local culture demand, e.g., the elongated goby (*Ctenogobius sagittula*), snail (*Cerithidea obtusa*) and mud-skipper (*Pseudapocryptes elongatus*) (Truong et al., 2010), and some for export, e.g. mud-crab, lobster, tiger shrimp and white legged shrimp.

Vietnam's export on aquaculture started after the boom of global prices of shrimp during the 1980s. Transfer of know-how on feeding and on new species for aquaculture was through government's extension services, radio and later television; moreover, those who travelled, sometimes for government (military) service, picked up ideas and acquired specific knowledge (Bosma et al., 2006a) that spread through social networks (Bosma et al. 2019). To counteract the impact of pesticides used for rice-culture, the researchers developed convergent rice-fish systems where both fish and fresh water shrimp are cultured in ditches around the rice-field (Bosma et al., 2012). To handle changes in salinity, farmers in the intermittent brackish/fresh water zones continue to crop rice in the wet season and shrimp in the dry season (Huy, 2019). The species and systems in aquaculture are continuously evolving, as is shown for shrimp (Joffre and Bosma, 2009). The culture systems vary widely, e.g., raising tilapia in cages within boats, fattening *Pangasius* catfish in deep ponds from which litter is pumped regularly, collecting mud-crab in mangrove-shrimp farms and/or caging them individually in ponds, and growing shrimp extensively in ponds or intensively indoors. The area of mixed and integrated mangrove-aquaculture, however, remains confined only to the semi-protected forests due to constraining contracts, fear of losing land after planting trees and limited area (Bosma et al., 2014). Insufficient knowledge and awareness on risks of intensive shrimp farming further restrict opportunities for integrated aquaculture (Joffre et al., 2016; Bosma et al. 2019).

At present, Vietnam's Mekong Delta is among the global top producers of shrimp and *Pangasius* catfish (a fresh water species). In 2019, Vietnam earned over 3.9 billion USD from exporting frozen, processed (frozen) and fresh shrimp, or about 1.4% of Vietnam's total export value (OEC, 2020). In addition, exported fresh, filleted and otherwise processed fish garnered 3.4 billion USD (1.2%), and that of molluscs, 0.27 billion\$ (0.1%). We may assume that most of these three species come from the Mekong Delta. Meanwhile, export value of rice is less than 0.9%; furniture and seats, 3.3%; and electronics, 30%. Hong et al. (2017) report that the Vietnamese fisheries and aquaculture sectors have reached their export goals

through many unsustainable practices, and may fall down at the expense of environment health. In the future, high export levels can only be achieved if (1) government were to strictly enforce environment-friendly policies, and (2) industry were to recognise the urgency of addressing the issue; otherwise, the latter would undermine its own future.

To get an idea on the risk in shrimp farming -- from 2012 to 2015, a new disease called Acute Hepato-Pancreatic Necrosis Disease (AHPND) wiped out 100,000 jobs and over US\$ five billion of income (Trang and Bondad-Reantaso, 2019). This loss represents about one-third of Vietnam's potential annual export value (See below). New diseases are inevitable in intensive mono-culture production systems, particularly with poor water management like that in the Mekong Delta where water management systems are designed to irrigate rice, but not to prevent disease contamination between aquaculture farms. In Vietnam, the outdoor RAS systems have gradually gained popularity, in particular for shrimp culture, as RAS increases bio-security, and saves space and water (Nguyen 2021, oral communication). In RAS, farmers can produce 30-40 ton/ha of water area. When the RAS is managed as bio-floc system, the bio-floc-filled water can be fed to mud-skipper, before the water is recycled in a pond with tilapia and seaweed. Such combined systems are almost zero-waste. However, RAS needs lined ponds, while for sustainable aquaculture the contact between water and soil is important, and helps to achieve more efficient resource use (Joffre and Verdegem, 2019). Thus, concrete lining of the pond-bunds is practical, but lining of the bottom, either with concrete or plastic, is not recommended for good soils.

#### **4.4.6 Mangrove conservation or local livelihoods: benefit sharing**

A recent study demonstrated that local farmer's decision-making with regard to mangrove conversion for shrimp farming in the Mekong Delta has been primarily determined by economic considerations since the income from mangroves is very low compared to that from shrimp farming (Ha et al., 2012). The decision-making by local farmers was also influenced by the way in which benefit sharing policies are implemented by the state-owned forestry companies and management boards. However, attitudes by farmers towards mangrove plantation and protection were far from negative (Ha et al., 2012). The study supported the claim that shrimp farmers may well be able to plant, protect and manage mangroves if they have more rights and responsibilities over forests and are able to benefit more from the production of mangroves. In this way more sustainable management of mangrove forests may be promoted (Ha et al., 2012).

#### **4.4.7 Future context of aquaculture and mangroves in the Mekong Delta**

Five drivers, four socio-economic and one environmental, may notably influence the context of aquaculture and mangroves in the Mekong Delta. The socio-economic drivers are: 1. Successful national policies on poverty reduction and family-size restrictions, 2. Farm households' priority on education of their children so that they can escape from the risky life of farming, 3. Due a net rural-urban migration of (skilled) labour (about 14% for the five provinces; GSO, 2020) the number of elderly couples with no children to take over farming is increasing; they may either shift from rice farming to the labour-saving fruit and fish farming

if conditions allow, or stop and do occupations requiring low technical skills, 4. At present, in the zones with strong acid-sulphate soils and saline water influence, many of the ponds classified as used for extensive aquaculture belong to (absentee) owners having a salaried job, 5. Fast land-inward progressing salinity intrusion limits options for freshwater crops and rice-shrimp systems farming. These drivers give an opportunity for new more voluntary land reform policies that can create space for water and mangrove in a more resilient re-parcelled landscape in the estuary with (seasonal) salinity intrusion (see Sections 2.9 and 3.2).

There is a need for widespread ecologically diversified mangrove forests in the estuaries with islands and peninsulas in the Mekong Delta (Figure 95). These mangroves help to protect dykes, catch sediments, clean effluents, provide seafood (re)production, and store brackish water. In such landscapes, behind the mangroves along the waterway, a cascade of aquaculture farms can be created (Figure 36). Further inland shrimp-rice farms can be planned, as long as fresh water is available. The cascade can improve water management, and thus, reduce disease incidence and losses in shrimp by using a more diverse brackish aquaculture with e.g., seaweed, seabass and saline tolerant tilapia, making the shrimp less vulnerable to diseases and making at least aquaculture more sustainable.



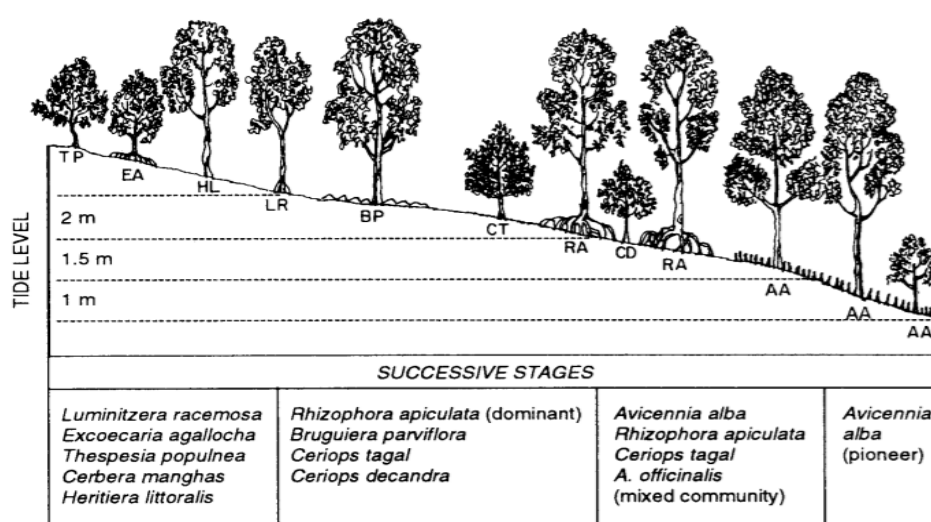
**Figure 95.** Farmers giving up part of their pond areas (bottom right) to widen the green belt along a river bank at Demak, Indonesia (Wetlands International, 2021).

#### **4.4.8 New mangrove greenbelts along the banks of rivers, creeks and canals**

New mangrove greenbelts should surround the redesigned landscape with cascades of ponds between separate canals for inlet and outlet water (see Sections 2.9 and 3.2). The nearer to



the coast, the wider the greenbelt should be; the larger the hinterland of e.g. cascade aquaculture, the more space can be reserved for greenbelts. New mangrove greenbelt spaces should have a favorable slope built over 20 to 50m along the banks of creeks, rivers and canals, providing the conditions for natural recovery of a biodiverse succession of mangrove, as described previously for Ca Mau province by Hong and San (1993) (Figure 96). The appropriate species for mangrove planting adjacent to aquaculture ponds can be identified through a combination of scientific knowledge and participatory methods (Khandkar-Siddikur et al. 2020).



**Figure 96.** The zonation and succession of mangrove vegetation at Ca Mau Cape, Mekong Delta (Hong and San, 1993)

### Aquaculture & other land-use in the Mekong Delta

#### Special considerations:

- Except for the national protection forests, most mangroves in the Mekong Delta have been replaced, first by rice and later by shrimp farms and infrastructure.
- At present, most aquaculture, such as intensive shrimp farms, can't invest in sustainable practices due to scale and high risk of disease spread because of poor water management options, i.e. no separate canals for inlet and outlet water.
- Freshwater provision is threatened in the coastal districts.

Tools: Reorient land-use from crops to brackish water aquaculture and mangroves.

Challenges: Align industry with sustainability goals.

Key: Enforce environment friendly policies

## 5 CONCLUSIONS

### 5.1 Coastal sediment dynamics considerations

- The Mekong Delta **subsides** rapidly by natural compaction, groundwater extraction and/or tectonic processes, at some places at rates of 4 cm/yr.
- The many dams and reservoirs constructed in the Mekong River catchment have reduced natural **sediment loads** in the river by 60 - 70%.
- **Sand mining** in the lower parts of the river (i.e. downstream of the dams) reduces sediment loads further, causing bank erosion - there is anecdotal information that the river now imports sand from the coastal zone.
- **Fine sediment** is also trapped in the lower estuary in a web of drainage/irrigation channels, and when inundating/irrigating agricultural fields (rice fields).
- Large parts of the coast are eroding, **erosion** rates have increased, and accretion rates have decreased during the last decade, attributed to a decrease in sediment input by the river, subsidence and cutting of mangroves.
- **Mangroves damp waves** (50 – 100% of the height of moderate waves over 500 m forest) and can grow with moderate (relative) sea level rise.
- Mangrove-mud coasts are **dynamic**; the time scale of the dynamics depends on the mangrove configuration and abiotic conditions.
- A mangrove-mud coast is built up by the **interaction of tide and larger waves**; the sediment is taken away and the coast is destroyed by smaller waves and by larger waves during wind stresses, respectively.
- A **sea dyke** too close to the shoreline has the following impacts:
  - sediment accommodation space is reduced, thus sedimentation rates reduce, and mudflat formation is slowed;
  - **wave reflection** on the sea dyke can increase erosive stresses and attack mangroves from the back;
  - the stable convex-up cross-shore mudflat profile may become concave-up, setting in motion a positive feed-back, enhancing **erosion** rates further.
- Coast-parallel **detached breakwaters** slow down erosion rates but do not stop erosion as all constructive onshore sediment supply is prevented, causing any efforts to rehabilitate mangroves between the dyke and such breakwaters to be unsuccessful (lost money).
- **Green-grey coastal infrastructure** may be a compromise to protect low-lying hinterland, while benefiting from mangrove ecosystem services, such as coastal protection, fisheries and production.

## 5.2 Mangrove restoration considerations

- **Do not use the same approach or method everywhere** when trying to restore/rehabilitate mangrove, as the success depends on site conditions & history. What works in one area might not in another area.
- Some of the most important factors to be taken into consideration for successful mangrove rehabilitation include:
  - **Sediment supply, sediment stability and sediment transport pathways**
  - **Tidal hydrology** (duration and frequency of tidal inundation)
  - **Propagule supply** (recruitment)
  - **Community participation** (sustainability)
  - **Livelihood** / land tenure considerations.
- Mangrove restoration only works in former mangrove areas; planting in **new locations** that never had mangroves before typically requires engineering interventions to modify site conditions, which can be labour-intensive and increase costs.
- Mangrove **planting is often not required**, except if a site is clearly recruitment-limited, but planting can help to speed up, diversify and assist natural recovery, and to promote community participation.
- Where planting is deemed helpful, it is critical to ensure sufficient quantities of **quality seedlings**, i.e. local (co-managed) nurseries, for the mangrove (re)planting program to achieve large-scale results.
- First develop a **system understanding** of physical site conditions (esp. sediment supply & stability, tidal hydrology), propagule dispersal and seedling establishment, factors determining resilience, and role of existing mangroves in coastal protection.
- **Ecological Mangrove Restoration** (EMR) is considered the best practice approach to mangrove restoration (including 7 or 8 steps/principles that focus on system understanding, to undo what was modified, facilitate natural recovery, plant only if necessary, ensure community participation and include socio-economic considerations).
- The **resilience** of mangroves depends on the size/extent of intact habitat, its species/genetic diversity, the level of recruitment (propagule supply) and the degree of connectivity with other (adjacent) mangrove areas.
- Adaptive solutions to **coastal squeeze** should allow space for a landward retreat of the mangrove vegetation in response to the effect of sea level rise (in many locations exacerbated by subsidence).
- The effectiveness of the role of mangroves in coastal protection depends on the mangrove species, **density and height** of the vegetation, and the width of the greenbelt.
- Recommended minimum **widths of mangrove greenbelts** along shorelines for coastal protection are in the order 500 m (or more), while the recommended minimum widths along riverbanks, waterways and channels are in the order of 20 to 50 m.
- There is need for clear **planning and opportunity mapping of the mangrove restoration approach**, to be guided by an appropriate **decision-making framework** (including consideration of past failures, proper site selection, clear objectives, achievable success criteria and practical monitoring requirements).

- **Community participation** is critical for empowerment and ‘ownership’ and ensures that the sustainability of the outcome of the mangrove restoration efforts is enhanced through co-management by the local population. Participation is essential to manage conflicting priorities and balance between short-term economic benefits and long-term environmental sustainability and resilience.
- When planning for mangrove restoration, give due consideration to ongoing **human activities** in the area, such as water management practices for aquaculture, sustainability of land-use (e.g. shrimp farming), land tenure arrangements and the potential need for re-parcelling/land-use reforms and development of alternative livelihood options.
- **Mekong-specific landscape considerations for mangrove restoration/afforestation:** Consider a combination of multiple approaches, including:
  - A. Conservation of existing mangroves and interventions to increase the resilience of degraded stands;
  - B. Seaward expansion of coastal mangroves along the seafront (afforestation outside the outer dykes) in combination with engineering interventions to increase sediment supply and stability;
  - C. Permanent landward rehabilitation of coastal mangroves in (former) shrimp pond areas<sup>4</sup> to restore (or widen) a protective greenbelt (where needed) and to allow for natural retreat in response to the effects of sea level rise (exacerbated by subsidence), by building a new dyke 1 to 1,5 km land-inward;<sup>5</sup>
  - D. Temporary, 30-50 year, landward rehabilitation by building a new dyke inland of an existing dyke where part of the land can be used again once a self-sustaining greenbelt and mudflat have recovered; and
  - E. Expansion of the width of mangroves along the banks of rivers, channels and creeks behind the inner dykes across a mosaic landscape of cascade aquaculture and/or silvo-aquaculture.
- Large-scale and **long-term changes**, including subsidence, salt intrusion, sediment shortage, sea level rise and coastal squeeze in the Mekong Delta need to be taken into consideration when planning mangrove restoration/afforestation initiatives.
- There is an urgent need to **balance between production and protection** (socio-economy/livelihoods versus coastal resilience) in the future management of mangroves and aquaculture in the Mekong Delta.

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<sup>4</sup> Such retreat may be partly facilitated by the ongoing process of salinisation in the Mekong Delta, which is already contributing to environmental and agricultural shifts (switching from rice- to shrimp farming) and migration of farmers (Dun, 2012).

<sup>5</sup> When exploring such ‘dyke retreat’, consideration could be given to incremental approaches, non-permanent interventions, temporary set-backs, only partial (localised) setbacks in strategic locations, and/or the establishment of a second dyke line further inland in certain areas to win time and allow for the transformation of vulnerable coastal pond areas into protective mangrove greenbelt zones.



### 5.3 Mangrove-Aquaculture (silvo-fisheries) considerations

- In coastal districts of the Mekong Delta, where salinisation progresses and the provision of **fresh water** is threatened, **land-use** should be reoriented from crops to brackish water aquaculture and mangroves.
- **Aquaculture** within mangroves disturbs the ecosystem, does not contribute to the expansion of the mangroves, and illegal practices will be hard to control. However:
- **Silvo-fisheries** with mangroves inside the ponds, or mangroves on the dykes, do not contribute to coastal protection or marine biodiversity, but shrimp farmers recognise the importance of mangrove greenbelts for disease prevention.
- Along riverbanks within the estuary, rehabilitation of a riparian **mangrove greenbelt** is easier than in front of the coastline, and protects the dykes, cleans the water for among others aquaculture, and provides space for water storage to reduce flood risks. This is currently jeopardised by sand mining yielding severe bank erosion
- At present, most aquaculture, such as intensive shrimp farms, can't invest in sustainable practices due to scale and high **risk of disease** spread because of poor **water management** options, i.e. no separate canals for inlet and outlet water.
- Within an estuary the mangrove greenbelts can be combined with a sustainable shrimp aquaculture landscape having **separated canals** for inlet and outlet water, and a **cascade of ponds** with different species (carnivores - tilapia - shrimp - seaweed / tilapia - clams) to reduce risks of disease outbreaks and environmental pollution. The mangroves, the treatment reservoirs and the settling ponds can be **managed as Joined Stock Company**.
- To become **sustainable**, an **aquaculture landscape** with many small farms needs rebuilding of water management infrastructure, re-parcelling of farm-land and compensation of drop-outs.
- Progressive salinisation, along with the increasing numbers of extensively managed ponds by absentee owners due to population dynamics, i.e. smaller households and net rural out-migration, and continuous bankruptcies of intensive shrimp farms, created a favourable socio-economic environment for the **re-parcelling** of farm-land, and resettling of households who have to move landward to allow for natural coast recovery.

Concluding, please note that the current severity and complexity of the environmental challenges in the Mekong Delta are the result of >100 years of historic developments, which cannot be solved within the short timeframe of a single ADB project, but requires a long-term integrated environmental management and development plan for the Mekong Delta. To create insight in, and to elucidate decision- and policymakers on the pro's, con's, and short and long term impacts, we recommend to conduct Visualisation Workshops and Social-Cost-Benefit-Analysis of different scenarios (including 'business as usual' and the various proposed actions and interventions) with a time horizon up to 2100.



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