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Chapter 10

Mangrove–eco-geomorphic interactions modelling for nature-based solutions

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ABSTRACT

Mangrove ecosystems are widely recognized for having highly valued multiple ecosystem services. These services, however, are often overlooked because of the lack of understanding of mangrove's species-specific and associated eco-geomorphological dynamics. Therefore, it will lead to a limited quantification and valuation of mangrove's functional and structural attributes. A mangrove ecosystem model capable of mechanistically simulating the feedback loop between mangrove stands and physical–environmental drivers is essentially important, specifically in the strategy of integrating mangroves as nature-based solutions for climate change adaptation and mitigation. The main objectives of this chapter are to gain a better insight into complex mangrove ecosystem and eco-geomorphic interactions to predict their trajectories, the possibility of modelling those utilizing a process-based model, and explore the interactions of mangrove, mudflat, and physical–environmental drivers. Following that, this chapter introduces a new hybrid model, so-called DFMFON, achieved by coupling mangrove individual-based and landscape-scale hydro-morphodynamic models which is capable of reproducing mangrove forest dynamics and morphodynamic delta features. To conclude, the application, limitations, and future development of process-based mangrove modelling for nature-based solutions are discussed.

Keywords: Conservation, eco-hydro-morphological modelling, ecosystem services, mangrove coasts, mangrove dynamics, mangrove modelling, mangrove–eco-geomorphic, mechanistic mangrove–morphodynamic, nature-based solutions, restoration.

10.1 INTRODUCTION

10.1.1 Values and functions of mangrove forests

Mangrove forests, in the past, were often misunderstood to be associated with a wasteland (Dahdouh-Guebas *et al.*, 2020) with a smelly, rotten-egg-like environment and a swarm of mosquito-borne diseases (Yeo *et al.*, 2021). That undervalued perspective is somewhat of a justification for clearing

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mangrove swamps, primarily in favour of aquaculture and agriculture (Hagger *et al.*, 2022) or other developments. Between 2000 and 2016, a satellite remote-sensing analysis study revealed that 62% of global losses of mangrove area were human-driven through land conversion (Goldberg *et al.*, 2020). This decline mostly happened in developing countries, where the mangrove forests were cleared for other high economic value activities. Mangroves have been declining at an alarming rate, ∼1–2% annually, even faster than coral reefs or tropical rainforests (Alongi, 2002; Duke *et al.*, 2007). These factors make mangroves one of the threatened ecosystems.

In contrast to the previous justifications for mangrove forest deforestation, the loss of mangrove forests, in reality, deprives the population of the high intrinsic value of mangrove forests. The intrinsic values are derived from the high capacity of primary productivity, terrestrial ecosystem supports, and marine food webs (Dahdouh-Guebas *et al.*, 2022). The presence of mangrove forests has proven to be an effective buffer for storm surges (De Dominicis *et al.*, 2023). The effectivity of the wave attenuation function is non-linear, dependent on the width, density, and fragmentation of the forest (De Dominicis *et al.*, 2023; van Zelst *et al.*, 2021), where scientists have acknowledged wave height reduction between 13 and 66% over 100 m mangrove forest width (McIvor *et al.*, 2012). Another benefit can be correlated with how communities perceive their relation with mangrove ecosystems in cultural non-materialistic interrelationships. Here, mangrove forests serve as places for recreational or intangible spiritual purposes for people (Das *et al.*, 2022). These benefits have been recognized as the mangrove ecosystem services, which can be divided into habitat, regulating services (e.g., regulating climate and disaster risk reduction), provisioning services (e.g., source of food), and cultural services (Das *et al.*, 2022).

10.1.2 Mangrove forests under threat

Over-exploitation and loss of mangroves significantly impact biodiversity and the surrounding ecosystems (Carugati *et al.*, 2018). A dramatic impact of mangrove forest loss to biodiversity can be seen, for example, in Indonesia – home of the largest mangrove forests, ∼20% of the global mangrove area. The United Nations Environment Programme (UNEP) (2023) reports that large-scale mangrove land conversion to oil palm plantations, rice fields, and aquaculture is directly linked to habitat destruction and has endangered 41 bird species unique to mangrove forests. Winterwerp *et al.* (2020) report a direct relationship between mangrove loss and the severe erosion of the muddy coast on Java's north coast, comprising ∼44% of Java coastlines. This is an alarming situation for the community, whose lives to a major extent depend on mangrove ecosystems. In Demak, Central Java, aquaculture establishment close to the waterline and mangrove loss has resulted in self-accelerated coastal erosion (World Bank Group, 2018). Land subsidence and mismatch of infrastructure development aggravated the coastline retreat up to 1.5 km, where 70,000 people have been affected (Damastuti *et al.*, 2022). That caused significant income loss for almost 80% of the village inhabitants (Winterwerp *et al.*, 2016).

10.1.3 Impacts of climate change

The recent sixth assessment of the Intergovernmental Panel on Climate Change (IPCC) reports that human activities have caused 1°C global warming with already widespread and intensifying impacts (IPCC, 2022). The primary evidence of climate change impacts on coastal areas, among others, is sea-level rise (SLR) and more frequent and stronger storms. The current estimate shows that 23% of the world population has been exposed to flood inundation of over 0.15 m in a 100-year return period (Rentschler *et al.*, 2022). This figure may increase following the SLR scenario, where the frequency and intensity will become higher. When compounded with the changing patterns of precipitation in the hinterland, it increases the depth and duration of flooding in urbanized coastal cities worldwide, where more than 600 million people live in these low-lying regions (Magnan *et al.*, 2022). The exposure to coastal flooding will increase with the growing population and social-economic activities, where the destruction effect will be multiplied.

10.1.4 Mangroves for climate mitigation

Sustainable approaches to mitigation and adaptation measures are critical to address the impacts posed by climate change. The goal of mitigation is to reduce the amount of greenhouse gases in the atmosphere by reducing the sources or enhancing the carbon sink. As climate change is inevitable, followed by the global target of achieving net zero emissions by 2050 (Saintilan *et al.*, 2023), the high potential of ocean and coastal ecosystems referred to as blue carbon ecosystems (BCE) has been proposed as the natural climate solution (Macreadie *et al.*, 2021). BCE consists of tidal marshes, seagrass meadows, and includes mangroves.

Owing to its co-benefits of carbon sequestration and climate adaptation, the potential of BCE for climate mitigation has been the subject of extensive research (Kusumaningtyas *et al.*, 2022). BCE has a disproportionally large carbon storage, storing half of the carbon sequestration in the ocean while only occupying 0.5% of the sea floor (Macreadie *et al.*, 2021). Even though the sequestering capacity varies highly among sites (Sidik *et al.*, 2023), among the other ecosystems, mangrove forests have the highest rate of carbon sequestration (Kusumaningtyas *et al.*, 2022). The mangrove area, nevertheless, can shift the role from carbon sink to carbon source when deforested. It is estimated, with global annual deforestation of 0.4% (Hamilton & Casey, 2016), that the carbon emitted due to the loss of mangroves will contribute to 3–19% of the total carbon emission (Donato *et al.*, 2011; Pendleton *et al.*, 2012). Between 1996 and 2020, the global carbon stocks net reduction associated with mangrove loss was estimated to equal 139 megatonnes (Mt) (United Nations Environment Programme, 2023). This net reduction equals four times the carbon produced globally in 2018, originating from fossil fuels burning and cement manufacturing. On the contrary, when rehabilitation attempts to increase the mangrove extent can be achieved, the potential climate benefit can reach more than 424 Mt carbon by 2030 (Sasmito *et al.*, 2023).

10.1.5 Mangroves for climate adaptation

Climate adaptation involves measures to reduce the risk and vulnerability to the unavoidable climate change impacts due to the past and current emissions. Adaptation measures include building coastal defences, early warning systems, or setting the setback area. Traditionally, coastal defences are approached from an engineering perspective and comprise man-made or 'grey' structures. The grey structures are designed to interact and mitigate a specific part of the physical processes, for example, waves, sediment transport, or water level (Coastal Engineering Manual, 2002; Shore Protection Manual, 1973). Given the continuous physical environment actions, grey structures require careful operations and likely costly maintenance to fulfil their design function and reach the expected service life (Duarte *et al.*, 2013). In mangrove–muddy coastal systems, the solutions using conventional permanent structures could lead to fragmentation and disruption of ecological connectivity of mangrove ecosystem services (Borsje *et al.*, 2011) or can induce scour due to the reflective surface in muddy coasts (Winterwerp *et al.*, 2020). Several studies mentioned the benefits of incorporating vegetation or ecosystem-based approaches have been approved to be more sustainable and cost-effective compared to hard structure measures (Tiggeloven *et al.*, 2022; van Zelst *et al.*, 2021). The approaches utilizing natural elements for coastal hazard mitigation, such as mangroves are commonly termed nature-based solutions (Narayan *et al.*, 2016). Mangroves can aid climate adaptation with their coastal protection function (Temmerman *et al.*, 2023), with the highest economic value per hectare (Macreadie *et al.*, 2019). The role of mangroves as coastal defence is well appreciated. Many studies have shown mangrove functionality, such as a buffer from tsunamis (Dahdouh-Guebas *et al.*, 2005) or attenuating wind waves and storm surges (Marois & Mitsch, 2015; Montgomery *et al.*, 2018).

10.1.6 Coordinated actions

Acknowledging a wide range of ecosystem services, mangroves have been considered one of the high priorities with coordinated global communities for mangrove conservation (Friess *et al.*, 2020). Countries have committed to their National Determination Contribution to conserve mangrove forests as a derivation of the Paris Agreement. To date, countries and international organizations have been advocating green belt policies (Su *et al.*, 2021), for example the United Nations (UN) Framework Convention on Climate Change (UNFCCC. Secretariat, 2021), Australia (Kelleway *et al.*, 2017; Morris *et al.*, 2021), China (Fu *et al.*, 2021), Engineering With Nature in the USA (Bridges *et al.*, 2018; Nature-Based Solutions Resource Guide, 2022) or the US National Academy of Sciences agenda (Board and National Academies of Sciences, 2019), and Indonesia (Sasmito *et al.*, 2023). Globally, mangrove restoration and conservation are in urgent need as targeted by the UN Decades of Ecosystem Restoration 2021–2030, Sustainable Development Goals, and Aichi Biodiversity Targets. The global conservation community has committed to increasing the global mangrove cover by 20% in 2030 (Global Mangrove Alliance, 2021; Sasmito *et al.*, 2023). Given the close deadline of both targets in climate change mitigation–adaptation and mangrove conservation, prompt actions in recent years are critical. When it is successful, conserving the remaining and restoring degrading mangroves not only could mitigate climate change but also reduce the impacts to 296 million vulnerable people in tropical coastal regions (Sasmito *et al.*, 2023).

Altogether, recognizing mangroves as nature-based solutions will include conservation, restoration, and ecosystem creation (Temmerman *et al.*, 2023). Despite global commitment and optimism, conserving and rehabilitating mangroves should be ecologically sound. As an illustration, of the reported mangrove restoration projects, 80–90% experienced failures (Lewis & Brown, 2014). The reasons for failures are mainly a mismatch of mangrove species, lack of understanding of the eco-geomorphological characteristics, and the mangrove species-specific ecological requirements (Ellison *et al.*, 2020; Kodikara *et al.*, 2017; Lewis & Brown, 2014; Primavera & Esteban, 2008). Additionally, it requires more than a decade for mangrove forests to reach their optimal forest capacity for mitigation–adaptation purposes. To achieve that objective, mid-course adjustment is often required, emphasizing the need for operation and maintenance.

10.1.7 Mangrove restoration

Global targets have mandated mangrove restoration and conservation as one of the high priorities and are in urgent need (Friess *et al.*, 2020) – recognizing their multifunctionalities for climate mitigation and adaptation. Those ambitious targets have promoted a push for slowing down mangrove deforestation from an annual average loss of 0.21% during 1996–2010 to 0.04% during 2010–2020 (The State of the World's Mangroves 2022, 2022). Nevertheless, many past restoration attempts have failed due to the misunderstanding of the mangrove wetlands' ecological–physical processes (Lee *et al.*, 2019) and land (social) conflicts (Sasmito *et al.*, 2023). At first, ensuring mangrove restoration success requires a mechanistic understanding of the ecological–physical processes, where current knowledge is segregated.

Several guidelines exist, either in mangrove restoration (Lewis, 2005; SER (Society for Ecological Restoration Science and Policy Working Group), 2002; Zimmer *et al.*, 2022) or focusing on the practical design (engineering) and management (procedures to achieve) on the mangroves' integration in the coastal protection (Bridges *et al.*, 2021; Van Wesenbeeck *et al.*, 2021; World Bank, 2016). Those guidelines have resulted from ongoing learning-by-doing processes (Aerts *et al.*, 2014; Wilms *et al.*, 2021; Winterwerp *et al.*, 2020). Some guidelines remain quite site-specific, so that duplication of such approaches should be conducted carefully. However, given the dynamic nature of mangroves, it requires a system understanding from the tree, forest, and ecosystem levels. The system understanding requires field observation, periodic monitoring and assessment, and quantification approaches with modelling.

10.1.8 Need for prediction tools

Understanding the urgent needs mandated by global targets coupled with complexities and the state of the current knowledge mentioned above reassures us that advancing new knowledge in mechanistic understanding of mangrove–intertidal flat interactions is critical. Field observation and monitoring techniques have been experiencing rapid growth, for instance, the Global Mangrove Alliance initiative (www.mangrovealliance.org) with their mangrove restoration tracker and global mangrove watch. However, tools that can mechanistically assess and predict the process components and interactions of mangrove forest evolution are yet underdeveloped. Examples of these modelling tools resolving eco-geomorphological interactions are spatial and statistical models, followed by process-based and conceptual models (Rivera-Monroy *et al.*, 2022). All of these approaches express mangrove dynamics as a top-down hierarchy. In this hierarchy, the response of mangroves is averaged or pre-defined based on specific environmental conditions.

In light of climate change uncertainty, the pre-defined response of mangroves, as in a top-down approach, may not be valid under the changing physical–environmental stressors. Pretzsch (2009) suggests the system's understanding may be built upon integrating all known processes, components, and their interactions from the bottom-up. As in our work (Beselly, 2024), we could mechanistically model forest expansion, retreat, and colonization influenced by physical–environmental drivers. In this regard, this research study will contribute to obtaining more realistic projections of forest structure and eco-geomorphological change that account for the dynamic environmental conditions, for example driven by climate change.

10.2 INTERACTIONS BETWEEN MANGROVES AND THE ENVIRONMENT

10.2.1 Understanding mangrove dynamics

By definition, mangroves are plants thriving in the wedge of the intertidal zone (Spalding *et al.*, 2010). The global presence of mangroves is limited within the (sub)tropics and warm temperate climates, between 30°N and 37°S (Mukherjee *et al.*, 2014), and specifically located in a sheltered waterlogged environment (Krauss *et al.*, 2014; Sharma *et al.*, 2022). Mangroves can only thrive in specific eco-geomorphological characteristics determined by their species-specific tolerance. For instance, the latitudinal range of mangroves is limited by the tolerance of mangroves to cold temperatures due to the efficient temperature range of photosynthesis (Ball & Sobrado, 1999) and limited capability to assimilate CO_2 in cold temperatures (Ward *et al.*, 2016). Regional climate variability, such as precipitation and evapotranspiration, can affect mangrove zonation. Under high net evaporation conditions, dieback within the interior of a mangrove forest can occur due to an increase in salinity (Cortés, 2019). In contrast, during events of high precipitation, mangrove productivity increases (Gilman *et al.*, 2007), and subsequent increase in the mangrove area due to landward migration into salt marsh zones because of the decrease in porewater salinity and sulfate concentration (Ward *et al.*, 2016). More locally, hydrological forcings, for example, river discharge, tides, and waves, determine the lateral expansion of mangrove forest, whereas soil biogeochemistry determines the structure, for example, basal area and tree height (Jennerjahn *et al.*, 2017). The regional scale geophysical description of coastal environments guided the ecological classification of mangroves into fringe, basin, scrub, or riverine mangroves (Twilley *et al.*, 2017).

Utilizing mangroves as a measure in the face of climate change uncertainty requires well-defined predictability of their functional capacity (Morris *et al.*, 2018) and the potential to persistently restore after disturbance (Temmerman *et al.*, 2023). However, mangrove capacity in attenuating waves, for example, varies over locations. It depends on the mangroves' biophysical characteristics (frontal area, density, and composition) and wave conditions (wave height and wave period), where the latter may benefit from highly detailed local observation or global hindcast. However, predicting the wave attenuation capacity, for instance, leads to a logistical challenge for forest inventory to carry out on the ground (Bispo *et al.*, 2019) as it is labour intensive. Studies have used different proxies to characterize the forest, for example, forest age (Maza *et al.*, 2021), characterization of the mangrove root structure (Mori *et al.*, 2022), and remote sensing (Beselly *et al.*, 2021). Attempts to approximate short-period wave propagation through vegetation fields have been formulated in the earliest original work by Dalrymple *et al.* (1984) and adapted in several recent works, for example Maza *et al.* (2019)

and Mendez and Losada (2004). The formulation requires an estimation of the vegetation bulk drag coefficient (C_D) where it is typically obtained through field measurements (Horstman *et al.*, 2021) or flume experiments (van Hespen *et al.*, 2021). Another approach is to estimate C_D as a function of mangrove projected area and volume (van Maanen *et al.*, 2015).

Mangroves have species-specific critical thresholds on the physical–environmental forcings that control their establishment or collapse. As an effect, these species-specific thresholds influence how we can predict the mangrove forest evolution and, therefore, measure persistence capacity. To add complexity, the critical thresholds are also varied over the life-stages (propagule, seedling, sapling, mature). The natural successful establishment would require propagule availability (Lewis, 2005), where the parental trees could be from the vicinity or adjacent as the propagules can float up to several months and be transported by hydrodynamic processes within short/long distances (Shih *et al.*, 2022). Successful seedling recruitment requires propagules to survive the window of opportunity (WoO), where external disturbances remain within their critical threshold. This WoO concept (Balke *et al.*, 2011) determines the set of species-specific thresholds on (1) the inundation-free phase, where a propagule can germinate and anchor the root within the low tide period, (2) a calm wave period, where waves remain low enough to prevent the seedling dislodgment, and (3) limited accretion–erosion, where the accretion will lead to burial and suffocate the seedling and erosion can lead to toppling and dislodgement (Balke *et al.*, 2013, 2014). Seedling establishment is an important factor in the forest's lateral (horizontal) expansion (Shih *et al.*, 2022; van Hespen *et al.*, 2022a). The species-specific optimum mangrove (vertical) growth is reduced by their local environmental conditions, that is, salinity, pH, and hydrogen sulfide concentration (Berger *et al.*, 2008; Chen & Twilley, 1998; Grueters *et al.*, 2014) in addition to the competition with the neighbouring trees. The interspecific competition to sources (space, nutrients, light, and by environmental forcings) limits the growth and, hence, leads to mortality.

10.2.2 Mangrove–mudflat interactions

Mangroves interact with the environment and provide feedback (Figure 10.1) dependent on the three factors in physical processes, that is, nutrients as the resources, salinity as the regulator, and sea level/hydroperiod (Grueters *et al.*, 2014; Krauss *et al.*, 2014; Wimmler *et al.*, 2021). These factors

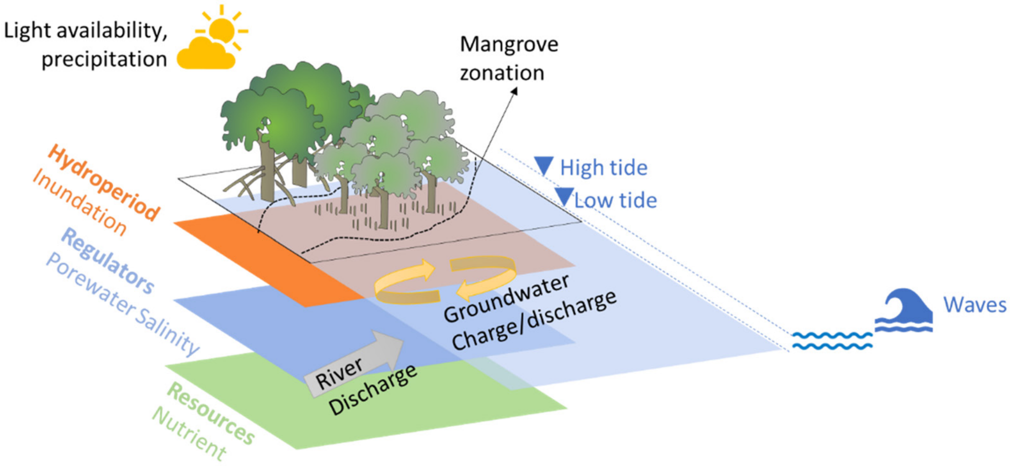

Figure 10.1 Schematic of hydrological factors (tides, waves, groundwater, and river discharge) and the regulating-resources factors along with the hydrological connectivity between mangrove forest and coastal waters (*source*: adapted from Rivera-Monroy *et al.*, 2022).

are intertwined and affect the growth and productivity of mangroves by regulating the zonation of mangrove forests as an influence on the porewater salinity distribution (Lovelock *et al.*, 2006; Piou *et al.*, 2006), limiting the growing capacity due to the availability of chemical/biochemical nutrients (Lovelock *et al.*, 2006; Reef *et al.*, 2010), and defining the spatial expansion dependent on the tidal flooding and duration (Balke *et al.*, 2015; Lovelock *et al.*, 2015). These factors determine the mangroves' growth, dieback, and seaward expansion (through tree recruitment).

Spatially explicit processes and feedback between mangroves and soil can be explained in three vertical soil layers, as shown in Figure 10.2. The relatively stable layer that comprises bedrock is located at the very bottom of the soil. In this layer, the movement is merely related to geologic processes. Thus, it is assumed to remain stable along the life of mangrove stands (Krauss *et al.*, 2014). Above the deep layer, subsurface layer processes are mainly related to the mangroves' rooting system (McIvor *et al.*, 2013). The root production contributes to the increase in soil volume and the subsurface elevation gain.

On the contrary, following the decomposition and compression of the roots after dieback, shallow subsidence may occur as root remnants take less space, decrease soil porosity, and promote soil collapse. Moreover, the shrink–swell of soil can be referred to as the increasing and decreasing soil water content due to the water absorption of the roots. In short, the subsurface processes comprise biological factors due to root production-decomposition and the physical factors in soil compaction and shrink/swell (Roskoden *et al.*, 2020). The top layer, as the most active, related to the surface processes that dynamically interact with the seawater. Here, the active processes are governed by hydrodynamics due to waves, tides, storm surges, and interactions with the mangrove's roots, stem, and canopy. This layer represents the transport and fate of materials in the form of sediment transport, nutrient exchange, salinity mixing, and mangroves' seedlings dispersal. The materials can be derived from outside of the mangrove forest (allochthonous) or inside the mangrove forest (autochthonous) (McIvor *et al.*, 2013).

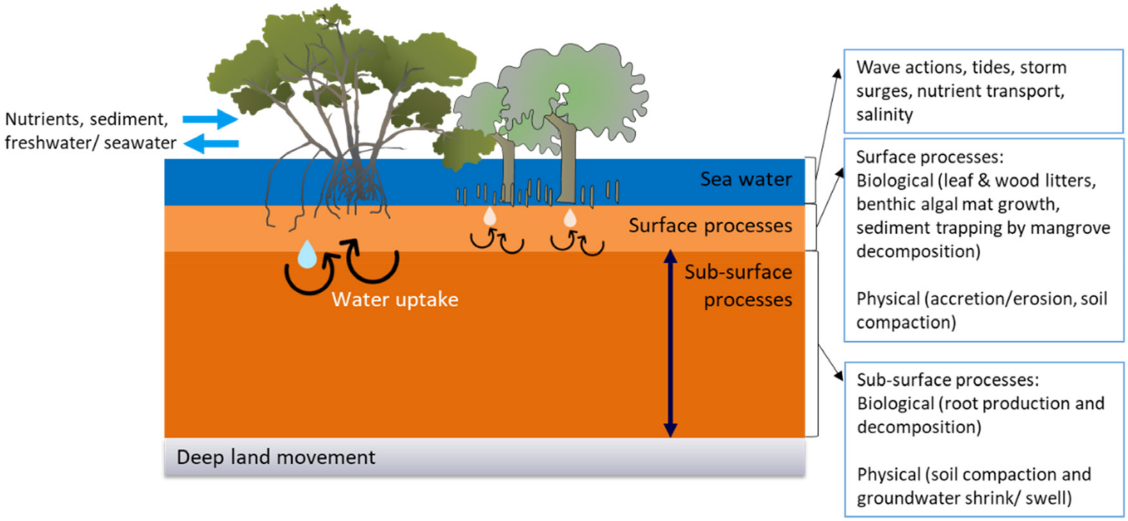

Figure 10.2 Illustration of the conceptual model mangrove–soil interactions. The interactions in soils are presented in three layers: the static and stable deep land movement layer, the subsurface layer related with root growth and decay processes, and the active layer associated with the hydro-morphodynamic processes. Mangrove trees respond to growth, dieback, and tree recruitment, thus providing new environmental feedback (*source*: adapted from Krauss *et al.*, 2014).

10.2.3 Modelling mangrove–mudflat interactions

10.2.3.1 Eco-geomorphic hierarchical approach

Due to the high variety of mangrove ecosystems, hierarchical approaches based on the biophysical characteristics describing their abiotic controls (hydroperiod, resources, and regulators, see Section 10.2.2) have been adopted and recommended (Krauss *et al.*, 2008; Rivera-Monroy *et al.*, 2004). This approach can be adopted, for example, at the geomorphological level, such as assessing mangrove vulnerability due to SLR (Lovelock & Ellison, 2007), at the forest level such as in ecological mangrove restoration techniques (Lewis & Brown, 2014) that provide structured restoration attempts by considering wetland's geomorphology and hydrology, and a detailed tree-level investigation on nutrition competition and hydroperiod effect on growth and change in community structure (Rivera-Monroy *et al.*, 2004). It shows how mangrove ecosystems interact and operate on different spatiotemporal and life-stage scales. Any disturbance on each hierarchy would alter ecosystem processes and may cascade into the higher hierarchical level and, in turn, affect the interactions and traits of the abiotic–biotic component at the lower levels. For example, if the change in sea level exceeds the sediment build-up, it may lead to the collapse and reorganization of mangrove zonation. To this end, no single model is capable of representing and simulating the processes encompassing all levels; they were designed to explain processes occurring at their own level (Rivera-Monroy *et al.*, 2022).

10.2.3.2 Model classifications

A mangrove ecosystem model is essential to assess and predict forest structure development. Still, ecological models need to parameterize interactions and processes in mangrove wetlands because it is impossible to cover the entire spectrum of spatial scales and time scales involved in mangrove biocomplexity. In other words, a model should be complex enough to pay tribute to the complexity it aims to describe but simple enough to understand what is going on. Mangrove ecosystem models can be classified into conceptual-, statistical-, spatial-, and process-based (Rivera-Monroy *et al.*, 2022).

10.2.3.2.1 Conceptual models

Conceptual models draw the intercorrelation causal effect of biocomplex responses with the change of stressors and the related attributes varying from the habitat unit to the global scale (Davis *et al.*, 2005; Day *et al.*, 2008). An example of a conceptual model is the windows of opportunity during calm periods on the variability of tides and winds to determine the recovery of salt marsh, mangrove, and floodplain vegetation (Balke *et al.*, 2014). Balke *et al.* (2014) used a time-series analysis of the water level and calculated how long the low water level lasted with respect to bed level, defined as an undisturbed period. The undisturbed period is then compared with the threshold of the inundation-free period required for each ecosystem to determine whether such an ecosystem is capable of establishing. The same holds for windspeed, where the threshold of a low wind velocity period should prevail to estimate the establishment of the vegetation.

10.2.3.2.2 Statistical and spatial models

These are the most widely used models, accounting for 60% of the papers reviewed (Rivera-Monroy *et al.*, 2022). It is partly because of the increasing (open) data availability, such as satellite constellation Sentinel (European Space Agency, 2015) by the European Space Agency and Landsat (USGS, 2013) by the NASA/USGS mission and the improvement in satellite sensor resolution and monitoring frequency. The development of open tools and processing methods, especially attributed to the cloud computing analysis, for instance Google Earth Engine (Gorelick *et al.*, 2017), allows for global and near-daily to weekly analysis of satellite imagery. The dominant usage of those big spatial datasets is to update mangrove inventory maps (United Nations Environment Programme, 2023). When complemented with field-based observations and statistical models, these maps can be used, for example, to evaluate mangrove spatial distribution (Bunting *et al.*, 2022) and assess carbon stocks (Murdiyarso *et al.*, 2015). The statistical models utilize the curated information of either remotely sensed or field-based datasets to infer the relationship between environmental variables and mangrove biophysical characteristics. For instance, an estimate of the global mangrove aboveground biomass based on the climate, assuming mangrove biomass corresponds to temperature and precipitation (Hutchison *et al.*, 2014). Although the application of statistical and spatial models has played an important role in explicitly explaining, for example, the drivers of mangrove loss and gain (Hagger *et al.*, 2022) or impacts of climate change on mangrove carbon stocks and fluxes (Alongi, 2022), what empowers this approach is also the major limiting factor, which is the datasets. The available dataset is rarely long enough or lacks consistency (Macreadie *et al.*, 2019; Sharma *et al.*, 2022; Zimmer *et al.*, 2022). The information gap may add complication in validating the prediction of the fate of the mangrove ecosystems in the climate uncertainty scenarios.

10.2.3.2.3 Process-based models

Process-based models, sometimes known as mechanistic models (Cox *et al.*, 2006), are numerical representations based on explicit causal mechanisms or processes on how the systems work, grounded in proven scientific knowledge (Cuddington *et al.*, 2013; Roelvink & Reniers, 2012). The explicit nature of process-based models in representing reality and its transparency on the assumptions of the processes provide more confidence in understanding phenomena either in hindcast or forecast simulations (Best *et al.*, 2018). In this chapter, we define process-based for two subclasses, that is, hydro-morphodynamic and vegetation dynamic models. Process-based hydro-morphodynamic models have long been used in wetland analysis, for example, modelling estuarine hydrodynamics (Thanh *et al.*, 2017), wave attenuation (Yoshikai *et al.*, 2023), tidal circulation (Horstman *et al.*, 2015), and sediment dynamics (Willemsen *et al.*, 2016). Another subclass, vegetation dynamic models, aims to simulate the mangrove forest trajectory.

10.2.3.2.3.1 Process-based: hydro-morphodynamics A process-based hydro-morphodynamic model is identical to simulating abiotic processes, for example, hydro-morphodynamics (Roelvink & Reniers, 2012). We can find examples of hydro-morphodynamic models in the application of mangrove wetlands, such as the investigation of tidal circulation within the wetland (Horstman *et al.*, 2015), wave attenuation function of mangrove forests (De Dominicis *et al.*, 2023; Pelckmans *et al.*, 2023), and effect on hydrodynamic and sediment exchange (Bryan *et al.*, 2017; Nardin *et al.*, 2016). The simulation exercises in the references mentioned previously assumed that physical processes occur in a static vegetation state, ignoring forest structure changes. Those studies commonly include detailed spatially varying hydro-morphodynamic–ecology interactions in a relatively limited vegetation time scale (seconds to weeks) where vegetation development within that period can be assumed to not significantly affect hydrodynamics (Bryan *et al.*, 2017; Friess *et al.*, 2012; Nardin *et al.*, 2016). Therefore, the model structure does not allow the inclusion of the feedback loop mechanisms of environmental (abiotic) changes to vegetation growth (biotic).

10.2.3.2.3.2 Process-based: vegetation dynamics The detailed physical processes involved in hydro-morphodynamic models have proven to be capable of elucidating mangroves' functional capacity on the physical drivers (van Hespen *et al.*, 2022a). However, the persistence or capacity to recover in mangrove forests is reflected in their ecological processes, which occur on a longer time scale, and encompass life stages from seedling to mature trees (Wang *et al.*, 2014). Over time, mangrove forest structure and composition will change in response to environmental conditions. Evidence can be obtained from past studies observing the vegetation dynamics based on satellite imagery or aerial photographs, which take years to decades of observation. For example, a study by Kleinhans *et al.* (2019) shows the migration and succession of riparian vegetation due to the river meandering process and the impacts of mangrove extent variation to estuarine hydro-morphodynamics in Waikaraka Estuary, New Zealand (Glover *et al.*, 2022). With this in mind, introducing the co-evolution of morphology and vegetation has an important role in providing a process understanding of the interactions of vegetation and physical drivers. We describe vegetation dynamic models in two types: population dynamic and individual-based models (IBMs).

(a) Vegetation dynamics – population dynamic
One approach to simulate vegetation dynamics is a population dynamic model, assuming the composition of individuals who share similar traits (e.g., species and biophysical properties) are grouped on a grid/plot and behave under a set of rules (Cappuccino, 1995). This grid-based population model can change simultaneously depending on the endogenous (e.g., density, above-belowground biomass) and exogenous (e.g., inundation frequency, wave energy) factors. To our knowledge, many studies focused more on the dynamics of mangrove primary productivity, for example, the phenology of *Rhizophora apiculata* on seasonal climate variation (Christensen & Wium-Andersen, 1977), the effect of nutrient availability on *Rhizophora mangle* (Onuf *et al*., 1977), biomass production of *R. apiculata* (Christensen, 1978), and effect of nutrient and irradiance to seedling growth of *Ceriops australis* and *Ceriops decandra* (Ball, 2002). Exception is in Thi Ha *et al.* (2003), where they investigate the seasonality of seedling growth and production rates of *Kandelia candel*. The main issue in population dynamic models, specifically in mangroves, is not being able to treat propagule dispersal. Instead, considered individuals within populations can follow homogenous behaviour without individual variation (Jørgensen & Fath, 2011).

(b) Vegetation *dynamics – individual-based*
Another approach is the IBM, where populations and communities are composed of discrete individuals and emerge by following the feedback loop of individual interactions with their environment (DeAngelis & Grimm, 2014). Individual organisms (biotic) are considered explicitly with their variability, local interactions, and specific adaptive behaviour in their physical environment (Grimm & Railsback, 2005), where they distinguish themselves from each other. Individuals can have different growth, survival probability, or reproduction capacities due to their relative position with other individuals (competition/endogenous factors) and with regards to the environmental conditions (exogenous factor). In this approach, individuals have self-directed motivation and adapt or modify their environment through their actions (Jørgensen & Fath, 2011). The main challenges in IBM modelling are their extensive computational requirement to run in large spatial scales (>1 km^2) and the need for long-term species-specific biophysical characteristics datasets. With the growing number of field datasets available and advances in IBM architecture, there is a possibility to improve IBM performance.

10.2.3.2.3.3 Integration of vegetation dynamics and hydro-morphodynamics In recent years, few attempts have been made to integrate two-way couplings of biotic–abiotic processes. In wetland modelling, several studies have attempted to include complex interactions on ecological, hydrodynamic, and morphological changes. This eco-morphodynamic modelling ranges from riparian (van Oorschot *et al*., 2016) and salt marshes (Best *et al*., 2018; Brückner *et al*., 2019) to mangroves (van Maanen *et al*., 2015; Xie *et al*., 2020). Those models were based on well-appreciated knowledge of hydro-morphodynamic processes. They assumed an abundant supply of seedlings to fill the numerical grid when the inundation threshold allowed, indicating colonization. Meanwhile, the numerical grid area is a proxy to explicitly define above-ground competition for resources (maximum density or biomass capacity), which drives growth and mortality. Eventually, the models do not consider the dispersal mechanism in a smaller time scale (Friess *et al*., 2012; Piercy *et al*., 2023) that is closely correlated with hydrodynamic processes (Duke *et al*., 1998; Shih *et al*., 2022), which determines vegetation establishment and lateral expansion. On the contrary, the development of IBMs has incorporated abiotic interactions in a somewhat limited way. One example is MANGA (Bathmann *et al*., 2020), which includes mangrove's effect on groundwater salinity by coupling IBMs with OpenGeoSys.

Integrating IBMs with abiotic processes is still mainly occurring in plot scale and over a short period. Despite already including the complex abiotic–biotic interactions, all integrated models discussed still lack representation of feedback loop processes at each life stage. Thus, it can be a challenge when we want to assess the wetland responses on non-stationary drivers such as variation in wave climate, sediment supply, and multispecies vegetation dynamics.

10.3 RESEARCH GAPS IN MODELLING MANGROVE ECO-HYDRO-MORPHODYNAMICS

10.3.1 Detailed mangrove dynamics

Despite the increasing number of numerical models evaluating the physical–ecological processes, those models still focus on the detailed abiotic interactions and use 'prescribed' ecological interactions, and thus may not reflect the actual mangrove dynamics, losing the important interactions at the lower level. Examples are models that consider interactions of the wave, tidal, and sediment trapping efficiency in mangrove forests but do not include dynamic ecological processes (Willemsen *et al.*, 2016), simulation of morphological evolution in sandy tidal embayments with aggregated mangrove feedback (van Maanen *et al.*, 2015), and mangrove response to SLR with pre-defined species composition concerning inundation depth (Buffington *et al.*, 2021; Rodríguez *et al.*, 2017; Xie *et al.*, 2022). Mechanistic interactions should consider the full life stages of the mangrove from propagule, seedling, and sapling to mature tree. An important note is given by Zainol *et al.* (2022) on the role of propagule dispersion in determining colonization and regeneration of mangrove forests, which is affected by hydrodynamic processes (Shih *et al.*, 2022).

10.3.2 Optimizing restoration strategies

There have been explicit approaches in mangrove restoration attempts; examples are ecological restoration (Lewis & Brown, 2014) and assisted rehabilitation or planting (Primavera *et al.*, 2011). Even though these approaches have considered the hydro-morphology and ecological system in the planning, they lack mechanistic understanding as time progresses. Therefore, restoration practice is still site-specific and results from long trial-and-error processes (Primavera & Esteban, 2008; Wilms *et al.*, 2021). Although the (long-term) prediction of abiotic systems has been well understood, for example, in Roelvink *et al.* (2020) and van der Wegen *et al.* (2011), when a biotic system is involved, either the spatiotemporal scale may be reduced to include details (Bryan *et al.*, 2017) or such simplification in vegetation dynamics should be required (Brückner *et al.*, 2020). The lack of spatial-explicit understanding of biotic–abiotic systems encompassing local to ecosystem scales may hinder successful restoration attempts. When looking at the global targets, the approaching deadline to cut carbon emissions and climate actions requires prompt solutions in developing restoration strategies. Therefore, a full life stage process-based model may be a helping hand to understand the interactions and the important factors determining a successful restoration.

10.4 MANGROVE–MUDFLAT PROCESS-BASED ECO-HYDRO-MORPHODYNAMIC MODELLING

10.4.1 Simulating mangrove–mudflat dynamics with a hybrid eco-hydro-morphodynamic model

Mechanistic mangrove–mudflat interactions are modelled with the two-way coupling of biotic–abiotic processes. It is approached by enabling the feedback loop of landscape-scale process-based hydro-morphodynamics with an individual-based mangrove dynamic model. DFMFON model (Beselly *et al.*, 2023) approached this by integrating the hydro-morphodynamic Delft3D-Flexible Mesh (DFM) and individual-based MesoFON mangrove model (MFON). The coupling of the two model paradigms allows for resolving the feedback loop of both short-term variation in local scale abiotic factors (flow, waves, sediment availability, and salinity), including the propagule dispersals, including the long-term bio-geomorphic mangrove forest dynamics.

As shown in the model schematic (Figure 10.3), the feedback loop between mangrove and the physical–environmental drivers is represented in two loops, that is, the inner and outer loops. The simulation is initialized by parameterizing the mangrove stand properties, that is species, diameter at breast height at 130 cm above the ground (D_{130}), vegetation density, and vegetation height in each DFM numerical cell. DFM applies the modified Baptist equation (Baptist *et al.*, 2007) to include mangrove-induced resistance λ (Equation (10.1)) and bare bed roughness C momentum equation (Equation (10.2)) by considering vegetation height h_v (m), density n (number of trees/m²), bulk drag coefficient C_D and unvegetated bed roughness C_b to water depth h, κ is the dimensionless von Kármán constant, and D is the stem diameter (m). In DFMFON, we predict C_D (Equation (10.3)) as the sum of the total drag coefficient of the bare surface $C_{D,no}$ and the ratio between the constant e and vegetation length L (Equation (10.4)). L is defined by considering the species-specific mangrove root and stem volume V_M varying over the water depth, by comparing it with the volume of water in a numerical cell (V).

$$\lambda = \frac{1}{2} C_D n D \frac{h_v}{h} \frac{C_b^2}{C^2} \tag{10.1}$$

$$C = C_b + \frac{\sqrt{g}}{\kappa} \ln\left(\frac{h}{h_v}\right) \sqrt{1 + \frac{C_D n D h_v C_b^2}{2g}} \tag{10.2}$$

$$C_D = C_{D,no} + \frac{e}{L} \tag{10.3}$$

$$L = \frac{V - V_M}{A} \tag{10.4}$$

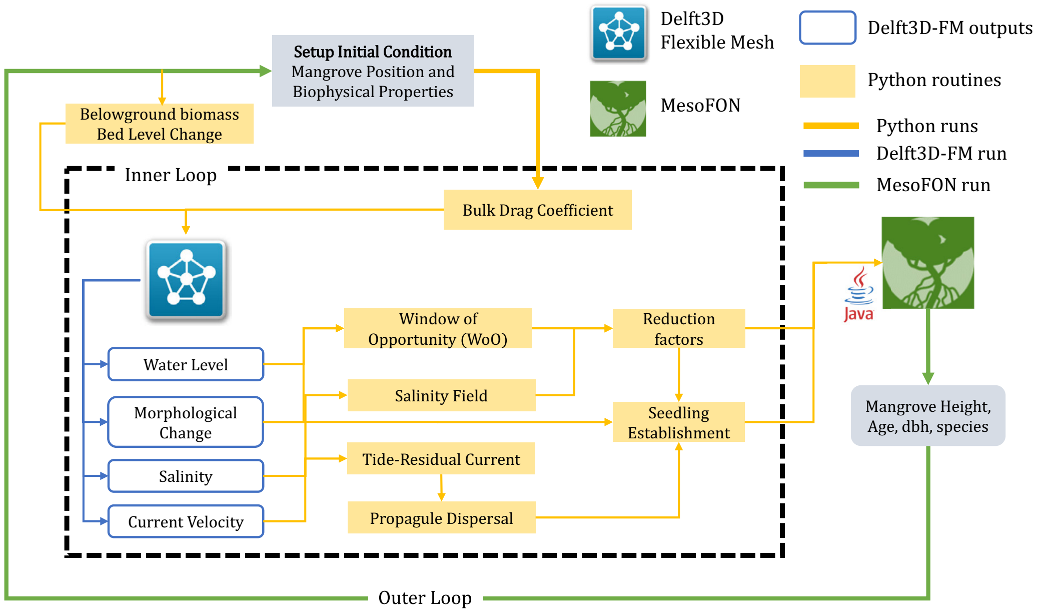

Figure 10.3 Schematic of the DFMFON model (*source*: adapted from Besselly *et al.*, 2023).

In the inner loop, DFM uses the vegetation information to simulate the hydro-morphodynamics and salinity for 90 morphological days. The reason for choosing this period is to allow the model to consider the seasonal fluctuation of the local climate condition and incorporate the mangrove's reproduction, that is, flowering, fruiting, and seedling production. The resulting DFM variables (water level, salinity, and bed level) are then converted as a WoO (Balke *et al.*, 2011) map, salinity distribution, and the tide-residual current. The WoO map is applied to evaluate the seedling establishment. In the model, we define a minimum of 3 inundation-free days and 5 inundation-free days for *Avicennia* sp. and *Rhizophora* sp., respectively. When the cell is inundation-free on the minimum required days, the propagules are allowed to establish. We assume that the propagules are buoyant and able to float irrespective of the salinity value. The magnitude and direction of the average tidal current over 2 weeks determined the final location of the propagules' spatial distribution over the numerical cells. In addition to the establishment due to inundation frequency, the effect of sediment disturbance on propagules, that is, burial and uprooting due to erosion is also considered. Eventually, the surviving propagules are allowed to grow into seedlings and afterwards into saplings when reaching 2 years of age. With this technique, the model closes the life cycle by considering the propagules production, seedlings establishment, saplings, and adult trees. DFMFON with this regard explicitly considers a successful seedling establishment as the factor driving the forest expansion.

Retrieving the information from the 90 morphological days inner loop simulations, the MFON considers the physical–environmental drivers that influence the mangrove dynamics. The salinity field simulated in the inner loop affects the growing capacity of mangroves and the number of propagule production. The surviving seedlings will be transferred into the outer loop once they have reached the sapling stage. MFON utilizes the field of neighbourhood (FON) approach (Berger & Hildenbrandt, 2000) to represent the strength of influence one tree exerts on another. In addition to the distance, the FON strength depends on their diameter and is subject to calibration for each species. The level of competition with the neighbouring trees is sensed via the FON value both for the above- and below-ground. With this, the mangrove growth is governed by the competition for resources and salinity field. The mortality occurs when the mangrove is indicated to not grow for 5 years sequentially (Grueters *et al.*, 2014). Currently, the model only considers the mortality as a result of the tree-to-tree competition for resources in the sapling-to-adult stage and burial-uprooting in the propagule stage. At the end of the outer loop processes, the simulated distribution and mangrove biophysical properties are fed to the inner loop as input, where the below-ground biomass contributes to the positive bed-level development.

10.4.2 Biocomplex interactions of the individual mangroves and physical–environmental stressors

The advantages of the DFMFON model are the two-dimensional horizontal characterization of the physical–environmental drivers and the capability of representing multiple species of mangroves and their individual interactions. This approach makes it possible to investigate forest expansion, retreat, and colonization, including the feedback loop with varying physical–environmental drivers. The model is applied with a case study in the Porong Delta, East Java Province, Indonesia. We investigated the model's capability and limitation in reproducing the mangrove extent development, the mangrove age–height relationship, and the morphodynamic delta features.

Mangrove forests in a mixed fluvial-deltaic with predominantly muddy sediment of the Porong Delta have seen a rapid expansion due to the unprecedented sediment load from the hinterland. The high sediment load originated from the mud volcanic eruption, which since the first burst in 2006 still delivers mud from the surface, albeit has declined almost to one-third of the first eruption (Karyadi *et al.*, 2012). This unprecedented flow forced thousands of residents to evacuate (Mazzini, 2018). To reduce the damage to the community and environment, the discharge of mud is contained in a reservoir. During the wet season, the mud is diluted and pumped into the river, increasing sediment load and concentration in the Porong River. The Porong River is regulated by a series of barrages and floodgates upstream, during the dry season the river flow is reduced to almost zero as it is diverted into the nearby metropolitan, in contrast to the high flow as the barrage is opened to prevent flooding in the city. As a result, within 11 years of the operation, a study by Beselly *et al.* (2021) shows the rapid delta expansion and seasonal mangrove dynamics in the estuary. The study combines multiple sources of satellite imageries to derive a 6-to-3 monthly mangrove extent map; which is more frequent than the globally available annual mangrove map. The study unravels the seasonal characteristic of the mangrove dynamic, where the mangrove forests tend to have successional patterns on the transition during the wet to dry season and dieback during the dry to wet season. The wet season correlates with the fruiting season and high freshwater discharge with additional mud disposal operation from the reservoir. The propagules are then dispersed and established on the newly deposited mud towards seawards. A relatively low saline condition and soft elevated mud layer increase the establishment opportunities. In the middle-to-end of the dry season, the increasing competition for resources and space leads to the dieback. More specifically for the Porong Delta, the dieback is pronounced likely because the massive sediment load is deposited further in the margin of the forest. Thus, it is transformed into the basin mangrove type in certain areas. The seedlings that are more sensitive to salt and drought might die under this condition in the basin-type areas by the end of the dry season.

The unique characteristics of the Porong Delta and the associated mangrove dynamics are simulated in DFMFON by creating a schematized funnel-shaped estuary with scenarios representing the seasonal high–low discharge and sediment load operation. In total, eight scenarios were tested to investigate the response of the system on variation in salinity (high, medium, low) on the condition of high, medium, and poor sediment availability (see the detailed scenarios in Beselly *et al.*, 2023). The model is schematized into a 1500 m length × 500 m width funnel-shaped estuary with a delta lobe in the middle. It starts with 140 m width of the river channel bifurcates into two branches encircling the 500 m × 640 m delta lobe at the upstream and the river branches are connected at the downstream. The mangrove population is characterized by consisting of only a single species of *Avicennia marina* that is dominating the Porong Delta. The mangroves are initialized as saplings at the age of 1-year-old with uniform biophysical properties and randomly placed in the upstream part of the delta lobe with a density of $0.03/\text{m}^2$, similar to the density found in the study area. The model is forced with two open boundaries, that is, fresh river discharge that also supplies sediment and tidal water level in the seaward boundary, and is simulated for 60 years.

The simulation results (Figure 10.4) show the realistic spatiotemporal mangrove–mudflat dynamics in prograding delta settings. The schematized model reproduces the expansion pattern of mangrove forest with the elongated shape towards downstream similar to the observation. However, the model could not reproduce the expansion rate similar to the case study. This has been expected considering the source of propagules in the model is originating from one source and one species, that is, *A. marina* in the delta lobes. Meanwhile, the case study has multiple sources from mangrove forests in the vicinity of the delta. In addition, the fruiting season for most of the mangrove species occurs during the wet season, where in this case, the river discharge and local precipitation could transport propagules over longer distances. The canopy height development shows a good comparison with the mean absolute error of 1.105 m (Figure 10.4d). The simulation realistically represents the cyclical seasonal pattern (Figure 10.4c) with the expansion during the wet season and retreat during the dry season.

The findings demonstrate the dependency of the mangrove ecosystem on the sediment supply and the gradual shift of mangroves from colonizers into ecosystem engineers. Understanding these interactions and functions is relevant, especially the latter, considering the function of mangroves as nature-based solutions. The capacity of mangrove stands in influencing hydrodynamics is varied over space and time. We observed three stages on the interplays of mangrove and mudflat, indicated by the mangrove canopy area as a proxy. The first stage occurs during the first 15 years of development indicating mangroves as colonizers, the seedlings establish and fill the allowable space until reaching the physical limit (mean sea level elevation). The high sediment supply and river discharge assisted in dispersing the propagules. As the established mangroves are relatively sparse, the effect on hydrodynamics tends to be limited and mostly dominated by local topography. Thus,

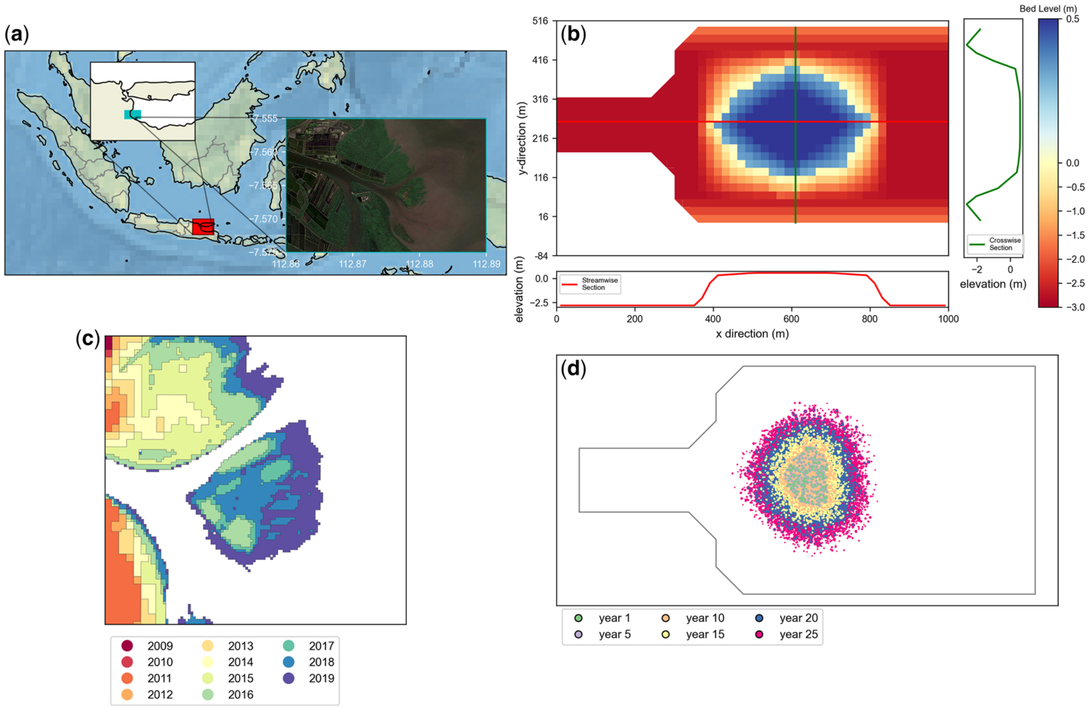

Figure 10.4 Depiction of real case study located in (a) Porong Delta, East Java Province, Indonesia, with focus on the northern delta lobe, (b) in which schematized as funnel-shaped delta with initial delta lobe in the centre, the schematized model is run for six different scenarios by varying the sediment concentration and fluvial discharge, here (c) a snapshot of simulated mangrove forest after 60 years for the base scenario (scenario A: high salinity and rich sediment concentration condition) is depicted, and (d) comparison of the simulated and observed mangrove canopy height (*source*: adapted from Beselly *et al.*, 2023).

there is not so much difference in morphology between no- and with-mangrove scenarios. In the second stage representing years 15 to 30, the expansion rate is increasing, given the initial population has become adult and produces more propagules. Here, we see the gradual distribution of mangrove stands with the older trees in the interior and younger trees at the fringe. The allowable space in the mudflat platform is seen to be rapidly filled with saplings. In this stage, we start to see the effect of mangroves in hydrodynamics, particularly by the large and dense older stands in the interior near the upstream of delta lobes. Their presence alters the flow and settles the sediment downstream, increasing the establishment probability. Hence, we would argue mangroves during the second stage act as ecosystem engineers, although somewhat limited and dependent on the sediment supply. After 30 years of simulation, the majority of the population has reached the reproductive age, and adding the number of propagules in the system. Here, in stage three, the propagules have been dispersed outside the delta and colonized the mudflat along the northern and southern channels.

The modelling experiment unravels the benefit of including full life-cycle mangrove dynamics and the feedback loops with the physical–environmental drivers. DFMFON accommodates the possibility of representing mangrove establishment and mangrove dispersal by considering hydrodynamic conditions. The capability to include this process is important since the establishment and dispersal are critical in determining the persistence of mangrove forests (Van der Stocken *et al.*, 2019).

The present development of the DFMFON model (Beselly *et al.*, 2023), however, considers the hydrodynamics as the main physical driver where mangroves act as obstacles. The extra obstacles can attenuate the waves, alter the flow, and eventually affect the soil surface development. Salinity is parameterized to control the mangrove growth, however, without the feedback from mangroves on the salinity distribution. The physical drivers determine the lateral expansion of the mangrove stands, whereas the environmental driver regulates the mangrove tree diameter, mangrove height, and propagule production. The tidal water level, wave force, and erosion-sedimentation determine whether a propagule can establish into a sapling. The established seedlings and adult trees will compete for the resources (space and nutrients), as a result, the high competition would lead to mortality. Currently, DFMFON considers extreme physical drivers (e.g., sedimentation and erosion) into the seedlings' failure (burial, toppling, and dislodgement). Estimation of the extreme physical drivers of young-mature trees such as stem breakage or uprooting remains a challenge, for instance how to estimate the mortality threshold following the breakage.

10.5 FUTURE DEVELOPMENTS OF MANGROVE MODELLING

Our modelling attempts have contributed to an ongoing exploration on the important parameters of mudflat–mangrove system dynamics. Currently, our knowledge of important trait-based parameters is limited to a few mangrove tree species. Extending it to other species and different regions is deemed necessary. An example is the WoO approach to describe seedling establishment that is valid for *Avicennia alba* in Singaporean mangroves (Balke *et al.*, 2011) and *A. marina* in Firth of Thames, New Zealand (Balke *et al.*, 2015). Interest in investigating this parameter has increased, for instance, as shown in the study of hydro-morphodynamic factors in seedling establishment for eight mangrove species in China (van Hespen *et al.*, 2022b). Replicating such a study for an extended database of mangroves and regions is possible. Current growth functions such as the one in Berger and Hildenbrandt (2000) initiates the growth parameter when the mangroves have reached the sapling stage, equal to breast height (137 cm). It is worth investigating the mangrove's growth after a successful seedling establishment into the sapling stage.

The presence of mangroves has important impacts on hydrodynamics, for example, attenuating waves and altering the flow. This is a relevant research topic, considering mangroves function as sustainable solution in disaster risk reduction. However, the mangrove capacity in attenuating waves, for example, varies between locations. It depends on the mangroves' biophysical characteristics (frontal area, density, and composition) and wave conditions (wave height and wave period). To approximate wave propagation, formulations require a function of mangrove projected area and volume on different water depth. Idealized vegetation configurations are commonly applied, such as for *Avicennia* sp. (van Maanen *et al.*, 2015) and *Rhizophora* sp. (Ohira *et al.*, 2013). All of these assume water depth will inundate the whole mangrove structure below the canopy. The hydrodynamic estimation will benefit most if such a database describing the function of (prop, pencil, cone, buttress) root, stem, and canopy exists. The correlations on species-specific root dimension would be better to be built based on the commonly used key parameters such as trunk diameter at breast height.

Although the chapter focuses more on the primary production and surface contributions, the contribution of below ground biomass to bed level is noticed, especially in the capability of mangrove areas to adjust to SLR (Krauss *et al.*, 2014). Similar to that, the role of, for example, mangrove litter and detritus to sediment accumulation in the forest interior is considerable (Ding & Fu, 2021). However, the challenge remains how to estimate their contribution and the relationship to determine the magnitude in a dynamic way in the model.

Mortality in the model is controlled by the competition between sapling-mature trees and during seedling establishment stage. Limited literature is available investigating mechanical tree damage, for instance due to the exposure to extreme wind and waves. This will be relevant, especially to understand the role of mangrove as coastal protection (Morris *et al.*, 2019; World Bank, 2017). Such a mechanistic limit on the extreme conditions leading to tree mortality, once it is known, will be useful to investigate mangroves' persistence.

The process-based mangrove hydro-morphodynamic model presented in this chapter is validated, focusing on Porong Delta, Indonesia. As of any other model, for site- or species-specific use, this coupled model requires to be validated with field datasets, with the emphasis on mangrove attributes trajectories, particularly, the growth parameter and seedlings production. More efforts are needed to gather local- to forest-scale datasets in mangrove wetlands, as these available datasets will be useful to further validate the model. There are increasing remotely sensed monitoring efforts in mangrove ecosystems, such as Global Mangrove Watch (Bunting *et al.*, 2022), and several field-based observations, for example, in Guyana (Best *et al.*, 2022) and New Zealand (Gijsman *et al.*, 2023). Although remotely sensed observations can provide a larger extent in understanding the dynamics of mangrove forests in their area development or vegetation phenology, a field-based dataset is limited to its temporal period and location. Additionally, as mentioned previously, the field dataset collected in the study area requires state-of-the-art equipment that needs trained experts to operate and can be costly – considering the remoteness of the mangrove forests. Off-the-shelf tools such as unoccupied aerial vehicles demonstrated in Beselly *et al.* (2021) can be a promising solution to provide very high-resolution imagery to complement satellite-based tools with lower resolution (10–30 m). Remotely sensed datasets in this way can only detect changes above the canopy and, hence, estimate the dynamics with canopy height or leaf index as a proxy; observing processes occurring under the canopy remains a challenge to investigate. Innovative low cost and autonomous monitoring tools are being developed, such as laser-based surface elevation dynamics sensors (Hu *et al.*, 2020) to monitor daily bed level changes or Mini Buoy (Balke *et al.*, 2021) to observe water level and current velocity in mangrove wetlands. Such low-cost observation equipment can potentially provide an extended period and a larger extent of the study.

There are roughly two types of process-based, eco-hydro-morphodynamic models (see Section 10.1). Each type of model serves the specific needs of the simulations. The question remains, how much complexity should we include in the models? Such as to what extent does including the effect of light availability, precipitation, or including interactions with subsurface hydrodynamics in the model affect the accuracy. Applying these tools requires more considerations than just adding physical and ecological processes. Although more datasets will provide opportunities to improve and include more processes in the model, the type of model applied depends also on the study objectives and research questions. In future research, a systematic comparison between individual-based and population dynamic vegetation models coupled with hydro-morphodynamics can be conducted, considering, for example, geomorphic and sedimentary settings, single or multiple tree species, or based on short- or long-term physical–environmental changes.

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