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ORIGINAL ARTICLE

Assessing potential impacts of sea level rise on mangrove ecosystems in the Mekong Delta, Vietnam

An T. N. Dang[1] · **Michael Reid**[1] · **Lalit Kumar**[2]

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Abstract

Sea level rise (SLR) due to global climate change negatively impacts coastal zones, in particular wetland and mangrove ecosystems. Mangroves in the Mekong Delta (MD) in Vietnam provide critical ecosystem services in the region; however, escalated relative SLR is likely to affect all ecosystems in the region, with mangroves probably more vulnerable than others. Given the fact that documented information and studies on SLR impacts on mangroves are limited for the region, this study aims to investigate potential changes in mangrove distribution in response to future SLR scenarios in the coastal area in the south of the MD using the Sea Level Affects Marshes Model (SLAMM). Wetland maps for 2013 derived from Landsat 8 OLI sensor, digital elevation model (DEM), and localized site-specific parameters (i.e., subsidence/accretion, erosion, historic trend of SLR, and over-wash) were used as input for the SLAMM to simulate spatial distribution of mangroves under different relative SLR scenarios (i.e., RCP2.6, RCP4.5, RCP8.5, more extreme SLR), and surface elevation change (i.e., subsidence, stable, and accretion) scenarios by the year 2100. Simulation results show that the average annual mangrove losses are likely to be 0.54% and 0.22% for subsidence and stable scenarios, respectively. The findings demonstrate the considerable impacts of SLR on MD mangrove ecosystems and the strong influence of subsidence processes. Inundation was also identified as a main driver responsible for the mangrove loss by the end of this century. Our results are in agreement with findings of other studies at global scales and observed data at regional scales. The results also demonstrate the potential of the approach developed herein for simulating mangrove dynamics under future relative SLR scenarios in the region with acceptable accuracy. The findings from the present study are useful sources for development of proper strategies for minimizing the impacts of SLR on mangrove ecosystems and their vital associated services, to protect and conserve the mangrove ecosystems in the region.

Keywords Coastal wetlands · Mangrove · Sea level rise · SLAMM · Mekong Delta, Vietnam, Climate change

Introduction

Mangrove forests are highly productive ecosystems in tropical and subtropical coastal regions, which benefit human society and ecosystems. Mangrove forests deliver critical ecosystem services to support the livelihood of millions of coastal communities worldwide, and annually contribute at least 1.6 billion (US dollar) to the global economy (Costanza et al. 1997; Hauser et al. 2017; George et al. 2019). Mangrove ecosystems not only protect coastal areas and associated populations from threats of natural hazards, i.e., floods, storms, and erosion (Giri et al. 2011; Ismail et al. 2012; Kathiresan 2012; Menéndez et al. 2020), but also importantly contribute to the enhancement of coastal water quality and biodiversity conservation by providing essential habitats for coastal flora and fauna (Giri et al. 2011; Kuenzer and Tuan 2013; Murchie 2015; McFadden et al. 2016). In

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An T. N. Dang
tdang6@myune.edu.au

Michael Reid
mreid24@une.edu.au

Lalit Kumar
lkumar9@hotmail.com

[1] Department of Geography and Planning, University of New England, Armidale, NSW 2351, Australia

[2] EastCoast Geospatial Consultants, Armidale, NSW 2350, Australia

addition, mangroves play a critical role in climate change mitigation by sequestering carbon (Tuan et al. 2015; Lovelock and Duarte 2019) and facilitating coastal accretion in response to sea level rise (SLR) (Krauss et al. 2017; Van De Lageweg and Slangen 2017).

Despite these critical functions and beneficial ecosystem services, mangroves have been degraded and deforested worldwide due to anthropogenic activities, climate change, and other human activities (Costanza et al. 2014; Richards and Friess 2016; Ghosh et al. 2019; Veettil et al. 2019b; Mafi-Gholami et al. 2020a, b). Globally, the total area of mangroves declined by around 35% in the 1980s (Lewis et al. 2016), while annual loss rates in Southeast Asia were double the global loss rate over the same period (Strong and Minnemeyer 2015). Importantly, climate change has been estimated to be responsible for 10–15% of mangrove loss, with corresponding declines in the ecosystem services they provide (Alongi 2008; Gilman et al. 2008). Due to being positioned in transition zones between ocean and lands, mangrove ecosystems are especially sensitive to SLR (Gilman et al. 2007; Ellison 2012). Therefore, among climate-associated factors, SLR is recognized as one of the major factors that adversely affect coastal wetlands and mangroves in the long term (LaFever et al. 2007; Kassakian et al. 2017), and so it controls the current and expected area and health of mangrove ecosystems globally (Lovelock et al. 2015, 2017). Previous studies concluded that global mangrove losses are inevitable unless mitigation and restoration measures in response to SLR are implemented (Gilman et al. 2006; Alongi 2008; Nicholls and Cazenave 2010; Ellison 2012). Loss of fringe mangroves is likely to cause adverse effects on ecological services, such as increases of shoreline erosion, exacerbation of impacts of storms, flood hazards, and tidal waves, and hence directly threatening coastal communities and their livelihoods (Barbier et al. 2008; Polidoro et al. 2010). Loss of mangroves can also lead to reduction in coastal water quality and biodiversity loss (Gilman et al. 2007). Hence, it is essential to plan appropriate mitigation and restoration strategies for the sustainability of mangrove ecosystems based on comprehensive and robust information on the potential distributions of mangroves under future climate scenarios.

Projecting responses of coastal wetlands to SLR has become a core research issue due to the complex and interacting physical, chemical, and biological processes that influence coastal wetland evolution, and the importance for natural resource management and policy making (Wiegert et al. 1981; Wu et al. 2015; Gopalakrishnan and Kumar 2020). Several models have been developed to project responses of coastal wetlands to SLR in response to this demand. These models include relatively simple models (Park et al. 1989; Doyle et al. 2003, 2010; Rogers et al. 2012; Strauss et al. 2012) as well as more complex ones (Costanza et al. 1990; Morris et al. 2002; Mudd et al. 2004, 2009; D'Alpaos et al. 2007; Kirwan and Murray 2007; Schile et al. 2014; Lovelock et al. 2015). While complex models take into account feedbacks and interactions between hydrology, sediment, sea level, and vegetation, and are thus likely to generate the most accurate and reliable projections, they require a lot of data and are computationally intensive. This means that they cannot be practically applied in many contexts, such as where limited data are available, or where projections beyond the local scale are sought (Martin et al. 2000; Wu et al. 2015). In these contexts, simpler models are required, which, while they do not incorporate feedbacks and interactions, have nonetheless proven to be reliable in validation exercises (Kirwan and Temmerman 2009; Wu et al. 2015). An example of such a simple model is the Sea Level Affecting Marshes Model (SLAMM).

SLAMM, a GIS-based model, is capable of simulating the dominant processes involved in coastal wetland change and shoreline alteration due to long-term SLR (Craft et al. 2009; Mcleod et al. 2010). The model focuses on analysis at landscape scales based on high spatial resolution data, which enable its predictions to be more realistic in comparison to other approaches (Craft et al. 2009; Clough et al. 2010). Other advantages of SLAMM are that it is open source, relatively easy to use, uses publicly available data, and can be used at a range of scales, from local to regional and subcontinental scales (Clough et al. 2010; Wu et al. 2015). Moreover, SLAMM has been steadily improved compared to its original version (Clough et al. 2010, 2016; Wu et al. 2015; Mogensen and Rogers 2018) and has been used to project the impacts of SLR on coastal wetlands, including mangroves and saltmarshes (Akumu et al. 2011; Li et al. 2015; Payo et al. 2016; Ekberg et al. 2017; Wu et al. 2017; Mogensen & Rogers 2018; Propato et al. 2018; Fernandez-Nunez et al. 2019; Raw et al. 2020; Wikramanayake et al. 2020). Although the model has been widely used in the USA, and some other areas, there is still a need to test its effectiveness in application as management tool in other parts of the world (Fernandez-Nunez et al. 2019).

The Mekong Delta (MD) in Vietnam includes approximately 66,000 ha of mangrove forests (Stefan 2018). However, this area has been significantly reduced, particularly in coastal areas of Tra Vinh, Soc Trang, Ca Mau, and Kien Giang Provinces, due to SLR and local anthropogenic activities, such as conversion of mangroves to aquaculture and agriculture (SIWRP 2017; Oanh et al. 2020; Dang et al. 2021b). While SLR has been identified as one of the key factors threatening mangrove ecosystems in the long term in the region (Duyen et al. 2015; Lovelock et al. 2015; Smajgl et al. 2015; Veettil et al. 2019a; Oanh et al. 2020), long-term threats caused by SLR for mangroves in the MD remain understudied. At a global scale, Lovelock et al. (2015) developed a model to project the potential submergence of mangroves in Indo-Pacific regions due to SLR by the end of this century. The model predicted the potential for mangroves to persist within an appropriate inundation regime (between highest astronomical tide and mean sea level) based on the known ecophysiology of mangroves in response to SLR. However, the model did not consider critical processes, including tidal range, subsidence, and coastal erosion, which are also likely to affect mangrove coverage (Payo et al. 2016). For local to regional scales, some studies have used digital elevation models (DEMs) to project changes in mangrove coverage under different future SLR scenarios (Veettil et al. 2019a; Oanh et al. 2020); however, these studies did not consider potentially important feedback processes operating in mangrove ecosystems, such as self-adaptation of mangroves to SLR, land subsidence and uplift, sedimentation, and vegetation regression and succession (Stralberg et al. 2011; Ellison 2012).

In this context, this study aims to assess the potential impacts of SLR on mangrove ecosystems in Ca Mau province in the MD using SLAMM (Park et al. 1989). The main objectives of this study are to (1) simulate spatial distribution of mangroves under various future SLR scenarios by the year 2100, (2) identify potential changes in mangrove coverage due to impacts of SLR, and (3) discuss feasible mitigation measures for conservation of mangrove ecosystems in the MD. This work will pinpoint affected mangrove areas under future SLR scenarios, and identify those areas experiencing severe threats, and so contribute to the improvement of mangrove ecosystem management and conservation in the MD.

Materials and methods

Study area

The study focused on the southernmost region of Vietnam, including the coastal area of Ngoc Hien district and parts of Nam Can and Dam Doi districts of Ca Mau province in the MD. The study area is located between latitude 8°33′–8°50′N and longitude 104°43′–105°18′E, covering an area of around 1000 km^2 (Fig. 1). The area has tropical monsoon climate with two distinct seasons. The dry season spans from December to April, whereas the wet season lasts from May to November. The study area is mostly flat and positioned in low-lying region of the MD, and hence severely vulnerable to the SLR impacts (Nguyen and Woodroffe 2016).

The study area has a critical ecological role in the region due to its high biodiversity and the ecosystem services it provides (Hauser et al. 2017). Mangrove forests in Ngoc Hien district are the largest in the region and include the last reserve of old-growth mangrove forests in Vietnam. These forests are also internationally acknowledged as a Ramsar site and UNESCO Biosphere Reserve (Tue et al. 2014; Van et al. 2015). The fringe mangroves in the study area are likely to be limited at their landward side due to extensive aquacultural and agricultural land use in those areas, which will prevent the inland migration of mangroves and potentially squeeze the fringe mangrove area.

Methodology

There were three main stages to the approach used:

(1) Dataset preparation and processing for spatial modelling, including DEM and wetland map.
(2) SLAMM approach for projecting SLR impacts on mangroves.
(3) Change analysis for identifying changes in mangrove extent and important factors of the changes.

The particular tasks involved in these stages are presented in the flow chart in Fig. 2 and described in detail in the following sections.

Dataset preparation and processing

Digital elevation model (DEM) and slope generation

In the SLAMM, elevation is considered as the most important factor influencing model accuracy as conversions between wetland classes are mainly driven by elevation (Clough et al. 2016; Fernandez-Nunez et al. 2019). Elevation data with high vertical resolution is essential for this kind of study since small changes in elevation are potentially important to the low-lying landscape (Ghosh et al. 2019). The present study used the high-resolution (i.e., 5-m spatial resolution, and better than 1-m vertical accuracy) DEM data, which was obtained from the Ministry of Natural Resources and Environment (MONRE), Vietnam. The DEM was constructed in 2008 using (1) survey points, and (2) the elevation points and contour lines of the topographical maps, which were acquired from photogrammetric data and geodetic survey (Trần et al. 2016; Minderhoud et al. 2019). The data is characterized by coordinate reference system of WGS_1984_UTM_Zone_48N, and vertical reference of the Vietnam's geodetic Hon Dau datum. The datum experiences elevation origin at mean sea level of the tide gauge at Hon Dau, an island offshore of Hai Phong province in Vietnam (Minderhoud et al. 2019). The DEM, afterward, was resampled to a spatial resolution of 15 m for further analysis due to the limitation of computer capacity and was clipped based on the boundary of the study area.

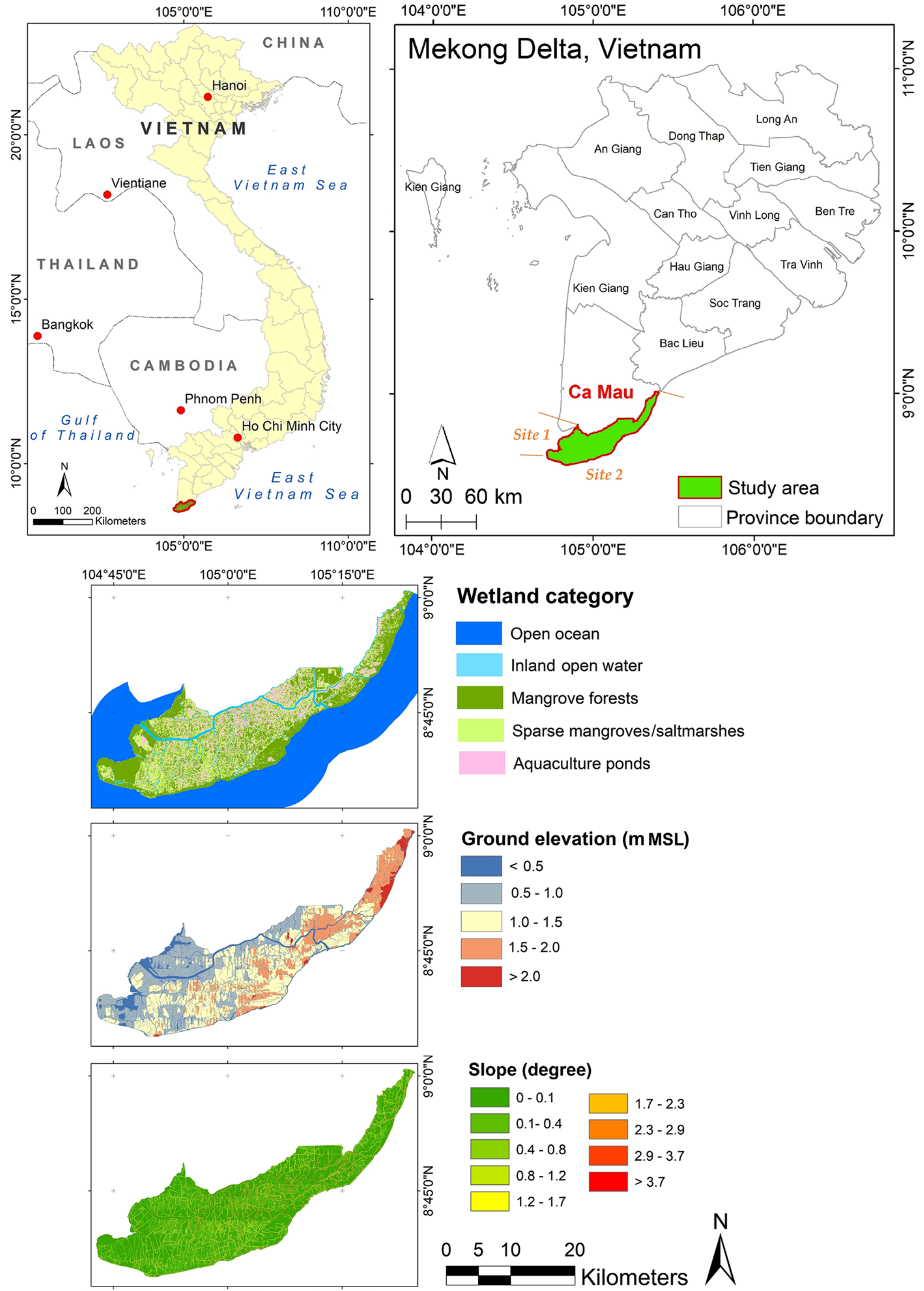

Fig. 1 Study area location, main wetland classes, digital elevation model (DEM), and slope gradients of the study area; MSL: mean sea level

Slope data is another essential input for SLAMM. The slope data (in degrees) was calculated from the DEM using spatial analysis tool in ArcGIS 10.4.1. The calculated slope data returned a mean value of 0.23° with the largest slopes ranging from 2.61 to 6.82° at the transition from lands along rivers and canals to mangroves (Fig. 1). The DEM and slope data in raster format were subsequently converted to ASCII Text format utilizing the data conversion tool in ArcGIS 10.4.1 for further analysis in SLAMM.

Wetland map

The present study used a wetland map, which was derived from Landsat 8 OLI images in 2013. The Landsat image was pre-processed, utilizing atmospheric correction, radiometric correction, seamless mosaicking, and sub-setting and enhancement of spatial resolution by the image fusion technique. The Combined Mangrove Recognition Index (CMRI) (Gupta

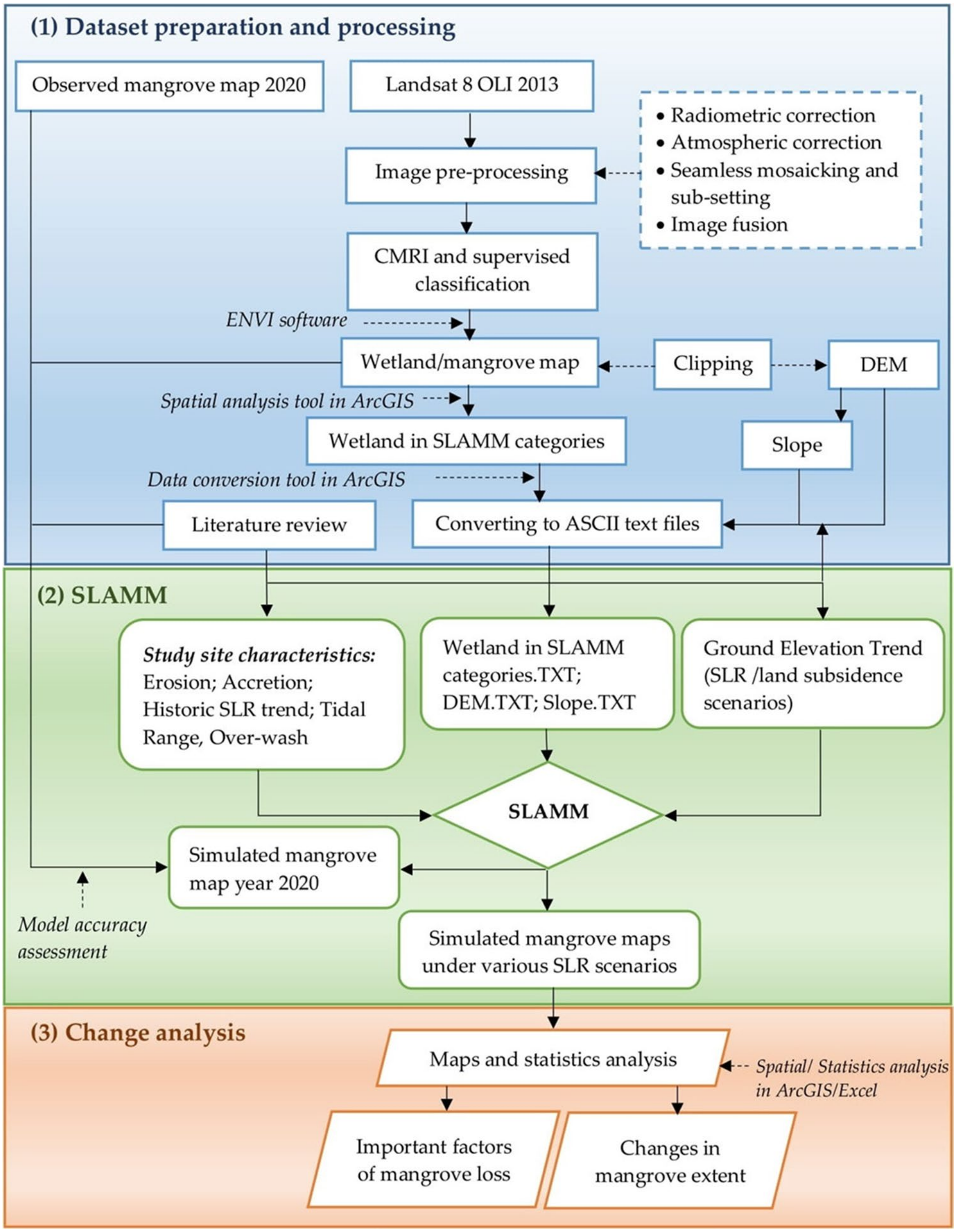

Fig. 2 Method flowchart. DEM: digital elevation model; SLR: sea level rise; SLAMM: Sea Level Affecting Marshes Model; CMRI: Combined Mangrove Recognition Index

et al. 2018) and supervised classification approaches were used in combination to identify wetland categories. The overall accuracy and kappa coefficient of the classified wetland map were 90.67% and 0.89, respectively. See Dang et al. (2021b) for complete details of the classification approach of the wetland map. Although wetland map for the year 2020 in the study area is available, the wetland in 2013 was selected for the analysis in SLAMM since observation data for the period 2013/2020 was subsequently used to validate the model.

The classified wetland map with 15-m spatial resolution included four categories: open water, inland water, mangrove forests, and sparse mangroves. The wetland map imported into ArcGIS 10.4.1 was extracted using the 15-m spatial resolution DEM to attain the same cell size and extent. The wetland map was subsequently reclassified into corresponding SLAMM categories (Table 1) using the spatial analysis tool in ArcGIS 10.4.1, based on knowledge, experience, and ecological definitions described in the SLAMM 6.2 technical document (Clough et al. 2012). The wetland map in SLAMM categories was afterward converted to ASCII Text format for the modelling exercise in SLAMM.

Sea Level Affects Marshes Model (SLAMM) approach

Sea Level Affects Marshes Model conceptualization

Changes in wetland areas within the study area in response to SLR were simulated using SLAMM version 6.2 beta

Table 1 Conversion of coastal mangrove habitats into SLAMM categories in the study area in the Mekong Delta, Vietnam

Wetland classes	Description	SLAMM categories	SLAMM code
Marine water	Open permanent salt-water along the coast	Open ocean	19
Inland water	Large or small fresh, brackish or saline water, including rivers, lakes, ponds	Inland open water	15
Mangrove forests	Dense mangrove forests in coastal zones with a minimum of 30% canopy cover, and dominated by mangrove species, i.e., *Avicennia alba and Rhizophora apiculata*	Mangrove	9
Sparse mangroves/saltmarshes	Includes vegetated mangrove and saltmarsh areas, normally in aquaculture ponds with crown cover of less than 30%	Mangrove	9
Aquaculture ponds	Artificial water bodies with regular geometric boundary for aquaculture practices		15

obtained from http://warrenpinnacle.com. The SLAMM approach assumes that wetland habitats correspond to a range of vertical elevations as a function of tidal range. The SLAMM considers five principal processes, erosion, inundation, accretion, overwash, and saturation; however, the current work only focussed on the influence of erosion, inundation, accretion, and overwash. These four processes are outlined below. Saturation, which simulates migration of fresh marshes and coastal swamps onto adjacent uplands as a response of the fresh water table rising to sea level (Clough et al. 2012), was excluded. This is because fresh marsh and coastal swamp categories are absent from the study area, and the study zone is surrounded by intensive aquaculture and agriculture lands (Liu et al. 2020; Dang et al. 2021b), which limit the inland migration of mangroves (Payo et al. 2016). It is also worth noting that SLAMM focuses broadly on wetland categories rather than mangrove ecosystems specifically. However, for tropical coastal systems, mangroves are the only wetland category included in SLAMM and are identified as lands with 0.50% or more mangrove cover (Clough et al. 2012; Payo et al. 2016).

Inundation The increase of water levels and salt boundary is tracked by lowering elevations of each cell due to SLR, hence maintaining the mean tide level constant at 0. Impacts of land subsidence vary spatially and are incorporated in these elevation calculations. The influence on each cell is determined based on the minimum elevation as well as the slope of that cell (Clough et al. 2012).

Erosion Erosion is triggered according to a maximum fetch threshold (9 km) and the proximity of the wetland to open ocean or estuarine water. If such conditions are satisfied, horizontal erosion will occur with a rate defined by site-specific parameters. Within a specific site or sub-site, erosion parameters for swamps, marshes, and tidal flats could be specified. Erosion rates of tidal flats pertain to both estuarine beaches and tidal flats if the beach experiences sufficient fetch to cause erosion. Erosion parameters for the tidal flats additionally apply to ocean beaches if the specific module of beach erosion is inactivated (Clough et al. 2012).

Accretion SLR is offset based on vertical accretion and sedimentation utilizing mean or site-specific values for each wetland category. Accretion rates could vary spatially within a specified model domain (Clough et al. 2012).

Overwash SLAMM assumes overwash to occur in barrier islands of under 500-m width during each 25-year time step due to storms. Sediment transportation and beach migration are also considered (Clough et al. 2012).

Model setup and site parameters

The wetland map in SLAMM categories, DEM, and slope files in ASCII Text format were used as primary data inputs for modelling in SLAMM. There are also important model parameters, such as relative SLR scenarios and site parameters (Table 2) which were required for SLAMM simulations. In SLAMM, wetland classes are in quasi-equilibrium with SLR, and each wetland class is able to convert into another category only once for each time step. The present study used a 5-year time step for simulating wetland changes for the period 2020/2100. Table 2 presents the list of data inputs and SLAMM parameters utilized for predictions of wetland/mangrove changes in the study area under various SLR scenarios.

Relative SLR scenarios Relative SLR (i.e., the balance of Eustatic SLR, subsidence, and sedimentation) was used in this study. SLR was predicted to have discernible effects on coastal mangrove ecosystems at the end of the century. According to our testing results and previous studies (Lovelock et al. 2015; Payo et al. 2016; Ghosh et al. 2019; Wikramanayake et al. 2020), the present study therefore considered different SLR scenarios by the year 2100. The four different SLR scenarios by 2100 are 0.45 m, 0.55 m, 0.75 m, and 1.40 m for the RCP2.6, RCP4.5, RCP8.5, and more extreme

Table 2 SLAMM data inputs and site parameters

Model inputs			Source
NWI photo date (year)	2013	2013	(Dang et al. 2021a, b)
DEM date (year)	2008	2008	MONRE, Vietnam
Slope	2008	2008	
Cell size (m)	15	15	
Model/site parameters	Site 1	Site 2	
Direction offshore of DEM	West	East	
Historic trend (mm/year)	5.37	5.37	(Lovelock et al. 2015)
Great diurnal tide range (m)	0.18	1.80	(Hak et al. 2016)
Tidal flat erosion (horz. m/year)	0.00	12.79	(Liu et al. 2017)
Overwash	25	25	(Clough et al. 2012; Danh 2015)
Mangrove accretion (mm/year)	4.00	4.00	(Saintilan et al. 2020)
Use of elev pre-processor [True,False]	False	False	(Clough et al. 2012)
Ground elevation trend (mm/year)	−6.50, 0.00, +6.50	−6.50, 0.00, +6.50	(Fujihara et al. 2016; Minderhoud et al. 2018)
Time steps (years)	5	5	
SLR by 2100 (m)	0.45, 0.55, 0.75, 1.40	0.45, 0.55, 0.75, 1.40	(Lovelock et al. 2015; Trần et al. 2016; Oppenheimer et al. 2019)

scenarios, respectively, for the MD. The RCP2.6, RCP4.5, RCP8.5 scenarios used are the regional scenarios, which were developed by the Ministry of Natural Resources and Environment (MONRE), Vietnam, used projections identical to the IPCC Special Report on Ocean and Cryosphere (Chapter 4) (Oppenheimer et al. 2019). According to the report, RCP2.6 is a low emission scenario in which global warming relative to 1850–1900 is projected to be likely below 2 °C and more likely than not to exceed 1.5 °C by the year 2100. For RCP4.5, a medium emission scenario, warming is likely below 3 °C and more likely than not to exceed 2 °C. For RCP8.5, a high emission scenario, warming is likely to exceed 2 °C and more likely than not to exceed 4 °C at the end of this century (Oppenheimer et al. 2019). Especially, RCP8.5 is considered to be unrealistically pessimistic, an extreme “no action” scenario (Hausfather and Peters 2020). Meanwhile, a higher value of 1.40 m of SLR was considered in this study to evaluate the more extreme scenario for Indo-Pacific region including the MD based on Horton et al. (2014) and Lovelock et al. (2015), which is considered plausible but unlikely (Jevrejeva et al. 2014). However, it is worth noting that the latest IPCC report (AR6) projects a global high-end scenario of 1.70 m (Arias et al. 2021).

Land subsidence in the MD due to natural processes of soil compaction and subsurface dissolution (i.e., formation of caves and sinkholes in karst landscapes), as well as the effects of anthropogenic activities (viz., ground-water extraction), occurs at annual rates of several centimeters. This rate exceeds current absolute SLR by an order of magnitude (Erban et al. 2014; Minderhoud et al. 2017). The mean annual subsidence rate for the MD was identified at 6.05 mm/year for the period of 1987–2006 based on surface water level trend analysis (Fujihara et al. 2016); similarly, using the InSAR-derived approach, rates of 6–7 mm/year from 1988 to 2009 were determined for undeveloped land-use categories, such as marshland and wetland forest in the MD (Minderhoud et al. 2018). In the present study, the subsidence, stable, and accretion scenarios refer to the net surface elevation change as a result of processes of subsidence and vertical accumulation of sediments. We used the average net surface elevation change of −6.50 mm/year, hereafter referred to as the subsidence scenario, in the MD for the SLAMM simulations, and assumed the rate to be constant until 2100. To evaluate the sensitivity of the simulation results in response to uncertainties on surface elevation change, simulations were also performed employing net surface elevation change of 0 mm/year and +6.50 mm/year, hereafter referred to as the stable and accretion scenarios, respectively.

Site parameters The study area was divided into 2 sites using the defined polygon function in SLAMM software due to the difference in terms of site-specific characteristics. Sites 1 and 2 represent the West and East Sea areas, respectively (Fig. 1). It is critical to note that tidal range and erosion rates are much higher for site 2 in comparison to site 1. The specific parameters for each sub-site are shown in Table 2.

In SLAMM, wetland types are assumed to remain from mean lower low water (MLLW, mean of the lower low water height each day) as the lower elevation boundary for this category up to an elevation equivalent to the mean high higher water (MHHW, mean of the higher high-water height each day). The Great Diurnal Tide Range (GT), which represents the difference between MLLW and MHHW, is hence an important parameter in SLAMM to evaluate any changes in wetland coverage. The present study used the GT values of 0.18 and 1.80 m for site 1 and site 2, respectively (Hak et al.

2016). The values were observed in Song Doc station (West Sea) and Ganh Hao station (East Sea), which were the closest stations to site 1 and site 2, respectively. It is assumed that these values remain constant over time and under different SLR scenarios. It is noted that the lowest and highest elevations for mangroves were assumed as mean tide level and salt boundary (or MHHW in the present study), respectively, in SLAMM based on Clough (2010) and Clough et al. (2012).

The historic trend of SLR in the study area is based on two sources. First, long-term rates of sea level change in the Ca Mau River for the period of 1979–2001 were determined from tide gauges and satellite altimetry; and second, an analysis of surface water level trends for the period from 2011 to 2014 (Lovelock et al. 2015). These SLR estimations were 5.74 mm/year and 5.00 mm/year, respectively (Lovelock et al. 2015); hence, we averaged the values for the two periods and set the historic trend of SLR to 5.37 mm/year for the region. Vertical accretion rate estimates for fringe mangroves range from 1.6 to 8.6 mm/year (average rate of 5.10 mm/year) (Krauss et al. 2014). For the MD, estimates vary from 3.00 to 5.00 mm/year according to Saintilan et al. (2020); however, the authors also found that mangroves were capable of accreting at 6.50 mm/year in the early Holocene, which are comparable to median values of surface elevation gain in Lovelock et al. (2015). Therefore, a mean accretion rate of 4.00 mm/year was used as a conservative estimate for the study area. Tidal flat erosion was set to 0 and 12.79 mm/year for site 1 and site 2, respectively, based on Liu et al. (2017). An overwash value of 25 years, as defined in the model, was used because this value also accords approximately to storm frequency in the MD (Danh 2015).

Model accuracy assessment

To assess the accuracy of SLAMM, the simulated mangrove areas obtained from the model were compared to observation data. Of particular relevance is the study by Dang et al. (2021b), which analyzed Landsat 8 OLI images for the MD over the period 2013–2020. The distribution of mangroves in 2020 was simulated using SLAMM approach, as per the above description, based on initiated wetland map 2013. For this period, we tested the three scenarios of elevation changes, namely subsidence, stable, and accretion. The results showed that mangrove losses were nearly same for the three scenarios, and mainly due to erosion, but not inundation. This is understandable because within this 7-year period, mangroves were not affected by inundation due to SLR. Therefore, we used the modelled results of the stable/no net subsidence scenario to compare with the observed data. The spatial and quantitative changes of mangroves obtained from SLAMM for the period 2013–2020, assuming no net subsidence, were validated against the observation data from Dang et al. (2021b).

Change analysis

Spatial analysis and potential change of mangroves

The impact of SLR on mangrove ecosystems in the study area was analyzed using spatial analysis functions in ArcGIS 10.4.1 software. Wetland maps under different SLR scenarios by 2100 were overlaid with the initial wetland map of 2013 to identify any changes in area and conversions between wetland categories. The analysis also included the identification of impacted zones of mangrove areas under various SLR scenarios.

Variable importance of mangrove loss

The contribution of erosion and inundation processes to mangrove losses was identified using spatial analysis techniques in ArcGIS 10.4.1. The analysis separated the proportion of mangrove losses by the year 2100 due to inundation and erosion. A straightforward analysis combined cell conversions with ground elevation, which is relative to mean sea level (Payo et al. 2016). Inundated cells were identified as those converted cells with elevation relative to mean sea level of below zero, whereas eroded cells were converted cells, which have elevation above this threshold.

Results

Model accuracy assessment

Figure 3 shows the loss or conversion of mangroves in the study area for the period 2013/2020 according to observation data from Dang et al. (2021b) and our simulation results using SLAMM. Dang et al. (2021b) found a total net loss of mangroves through conversion to ocean open water along the East coast of the MD of 11.23 km^2, equating to an annual loss rate of 1.60 km^2. This rate compares well to simulated mangrove loss for the period 2013–2020 obtained from SLAMM, assuming no net subsidence, of 10.3 km^2 (annual loss rate of 1.47 km^2). In the SLAMM, this loss was mostly to ocean open water, except very small area of less than 1 km^2 which was converted to tidal flat. However, the progradation of mangroves observed in the West Sea site (the Gulf of Thailand) over this period was not simulated by SLAMM.

Future projections of mangrove extent

Simulated maps of spatial distribution and projected area of mangroves under different relative SLR scenarios for 2100, produced by the SLAMM approach, are shown in Figs. 4 and 5. For the stable scenario of surface elevation

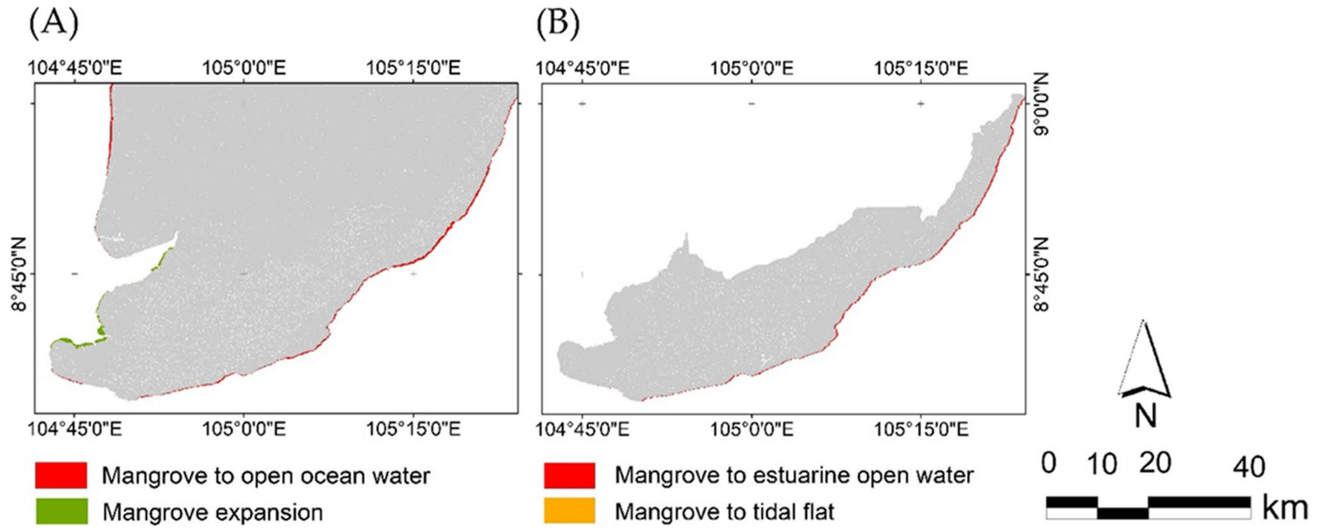

Fig. 3 Mangrove loss/conversion during period 2013–2020 according to (**A**) observations using remotely sensed data from Dang et al. (2021a, b) and (**B**) SLAMM simulation in the present study

change, mangrove area was projected to range from 732.44 to 354.24 km^2 for the RCP2.6 and more extreme SLR scenarios, respectively, whereas, for the accretion scenario, the figure slightly increased to 770.88–632.02 km^2 for the corresponding SLR scenarios. In contrast, the figures considerably decreased to 588.04–106.27 km^2 under RCP2.6 and more extreme scenarios with the subsidence scenario.

Changes in mangrove extent

Overall, mangrove extents are consistently reduced under the RCP2.6, RCP4.5, RCP8.5, and more extreme SLR scenarios for the year 2100 in comparison to the year 2013. Unsurprisingly, mangrove loss was higher for the more extreme SLR and subsidence scenarios (Figs. 6 and 7). For the stable scenarios, projected annual loss rates ranged from 0.10% (0.79 km^2) to 0.64% (5.07 km^2) under the RCP2.6 and more extreme scenarios, respectively, whereas for the accretion scenarios, the figure slightly decreased to 0.05% (0.37 km^2) and 0.24% (1.91 km^2) for the same scenarios. For the subsidence scenarios, the projected annual loss rates were much greater, with rates of 0.31% (2.42 km^2) and 0.93% (7.91 km^2) for the RCP2.6 and more extreme scenarios, respectively.

Figure 6 shows the potential conversions of mangroves and other wetland categories under the 12 relative SLR scenarios by the year 2100. Mangrove loss was mostly due to its conversion to estuarine open water due to SLR. The conversion was taking place mostly in the West Sea for the lower SLR scenarios (RCP2.6 and RCP4.5 with the stable and accretion scenarios) and extended to the East Sea and further inland under higher SLR scenarios. A very small area of mangrove (less than 0.10%) was transferred to tidal flat in the east sea of the study area. Finally, there were conversions of inland open water to estuarine open water in the study area.

Variable importance of mangrove loss

The relative importance of inundation and erosion as drivers of mangrove loss in the model is shown in Table 3. The findings show that inundation was dominant in 9 of the 12 scenarios, and thus can be expected to be the main cause for mangrove loss in the study area by 2100. Erosion was, however, dominant under accreting conditions for most SLR scenarios (i.e., RCP2.6, RCP4.5, and RCP8.5) due to high erosion rates in along the East Sea coast under these scenarios. Inundation remains the dominant driver with the accretion scenario under the more extreme SLR scenario.

The sensitivity of the model to inundation versus erosion was calibrated using different rates of erosion. We increased erosion rates up to 100 m/year and tested the model for the RCP2.6 and RCP8.5 scenarios. The results show that mangrove loss by the year 2100 due to erosion only increased by less than 1% for the three subsidence scenarios. Therefore, the use of a constant value of 12.79 m/year for annual erosion rate had minimal influence on the relative contribution of inundation and erosion to the loss of mangrove.

Discussion

Model projections and performance

Findings from the present study suggest that SLR with substantial inundation will have adverse effects on mangrove

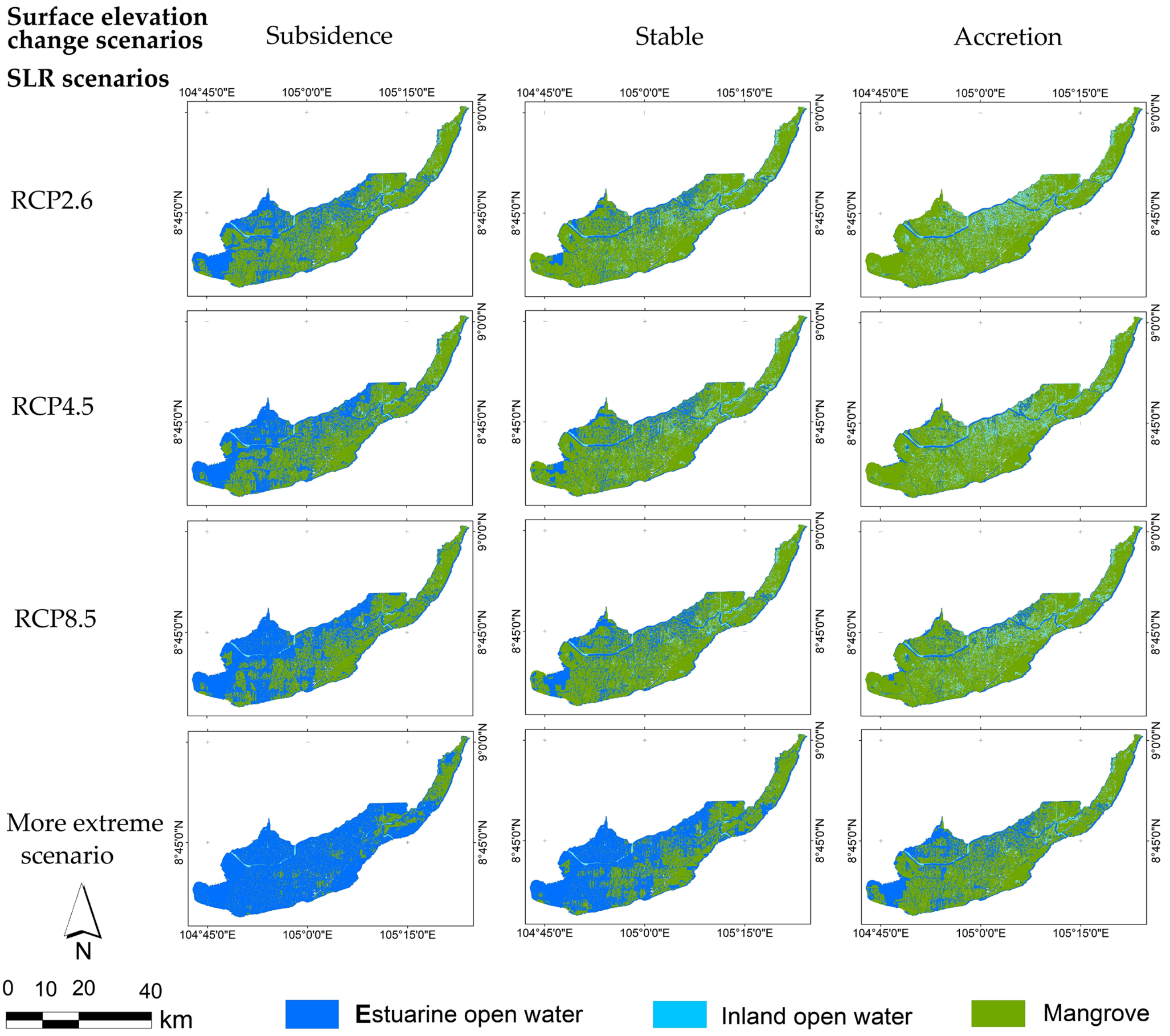

Fig. 4 Simulated maps of spatial distribution of mangroves in the MD under different relative SLR scenarios by the year 2100

ecosystems in the study area by the end of this century. For the worst case (more extreme SLR and subsidence scenarios), mangrove loss was projected to be 80.60% (annual loss rate of 0.93% or 7.91 km^2), whereas the figure for more extreme SLR and no subsidence scenarios was 55.68% (annual loss rate of 0.64% or 5.07 km^2). The more extreme SLR scenarios are plausible but unlikely (Jevrejeva et al. 2014), and so the RCP8.5 SLR scenario is likely to be more reasonable in representing the risk for mangrove ecosystems. Under the RCP8.5 SLR scenario, mangroves are projected to decrease by 45.84% (annual loss rate of 0.53% or 4.17 km^2) in combination with the subsidence scenario, in comparison to a decrease of 15.74% (annual loss rate of 0.18% or 1.43 km^2) in combination with the stable scenario.

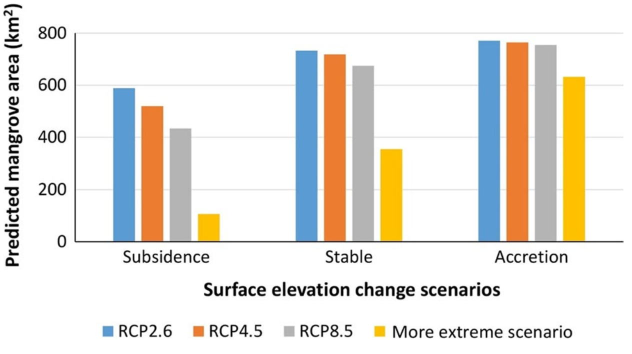

Fig. 5 Simulated mangrove area under different relative SLR scenarios by the year 2100

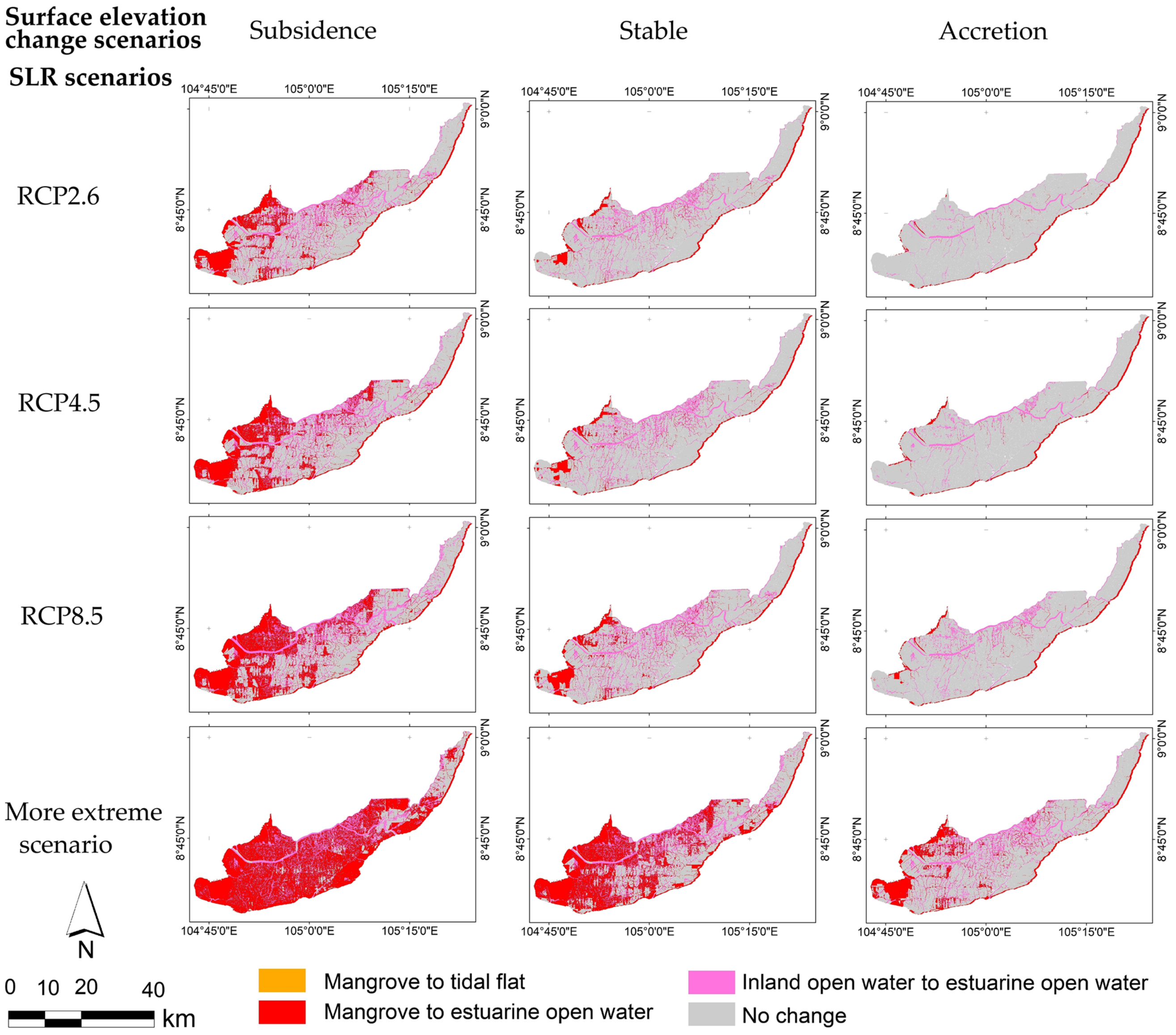

Fig. 6 Simulated mangrove conversions under twelve different relative SLR by the year 2100 compared to baseline scenarios of 2013

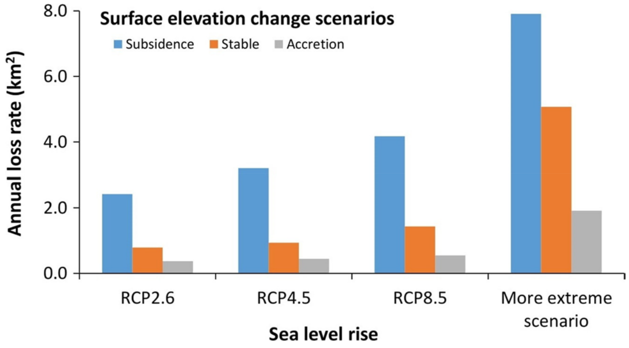

Fig. 7 Annual loss rate of mangrove under different relative SLR scenarios by the year 2100 compared to baseline scenario of 2013

The level of mangrove loss due to SLR projected in the present study is similar to that which has been predicted in previous modelling and observational studies. Among the modelling studies, mangroves were projected to decrease by 0.13–0.20%/year (Gilman et al. 2007, 2008) and 0.10%/year (Payo et al. 2016) for mangroves in the Pacific Islands and the Sundarban (Bangladesh) respectively by 2100 assuming no subsidence. These rates are similar to the average annual loss rate 0.22% for mangroves obtained in our study. For the subsidence scenarios, the rate of 0.54%/year found in the present study is somewhat higher than the loss of 0.37%/year from Payo et al. (2016). This can be attributed to the higher rate of subsidence of – 6.50 mm/year used in the present study in comparison to – 2.50 mm/year subsidence rate used in the Sundarban modelling.

Our findings also show that the erosion process critically contributes to mangrove loss under all scenarios. The validation shows that the modelling results of mangrove changes with the annual loss rate of 1.47 km^2/year in the East Sea due to erosion correspond well to observation data for the period 2013/2020 (Dang et al. 2021b). This projection is also in agreement with the observations of Liu et al. (2017), which showed that the shoreline in the East Sea of Ca Mau province eroded by 1.71 km^2/year over the period of 2005–2015. These findings, therefore, demonstrate that the simple erosion rule used in SLAMM is capable of reproducing the observed mangrove loss due to erosion from 2013 to 2020 in the study area, thus validating the model's capacity to simulate mangrove changes with reasonable accuracy.

While the SLAMM correctly identified erosion as an important process driving shoreline retreat in the East Sea of Ca Mau province between 2013 and 2020, it did not project the progradation of mangroves observed in the West Sea site (the Gulf of Thailand) over this period. This issue is partly due to the use of the constant net subsidence for the whole study area. In addition, the low wave energy of the West Sea site in combination with high sediment supplied from the erosion of the East Sea shoreline led to land aggradation for mangrove growth in this area (Tran Thi et al. 2014; Marchesiello et al. 2019; Phan 2020). These specific coastal geomorphological processes in the study area are not fully captured in SLAMM, and so these areas of progradation are not reproduced in the simulation model.

In relation to the mechanism of SLR-induced mangrove loss, our findings agree with previous studies. Gilman et al. (2008), in particular, proposed that SLR with considerable inundation is likely to be the key driving force of mangrove loss by 2100. Our simulation results suggest that inundation due to SLR was the dominant driver under 9 out of 12 relative SLR scenarios by 2100 (Table 3). However, subsidence rates do influence the contribution of erosion to mangrove loss, with erosion demonstrated as a dominant cause of mangrove loss under the accretion scenarios. Payo et al. (2016) also suggested that erosion might remain as a main driver of mangrove loss depending on net subsidence rate.

Model uncertainties, study limitations, and contributions

The above validation demonstrates that our approach appears to accurately project mangrove distribution under different SLR scenarios based on comparison with global studies and regional observed data. However, the SLAMM approach is sensitive to several factors, which are likely to influence simulation accuracies. There are also limitations in the current study, which are necessary to highlight directions for future studies. Table 4 summarizes the uncertainties of the SLAMM and the present study as well.

Elevation data is one of the key factors that critically affect the accuracy of simulation output in SLAMM because conversion between wetland habitats is mainly governed by elevation (Wu et al. 2015; Clough et al. 2016; Payo et al. 2016; Fernandez-Nunez et al. 2019). In the present study, a DEM with vertical accuracy of less than 1 m was used to model the impacts of SLR on the mangrove ecosystems. The DEM was constructed using survey points and elevation points derived from photogrammetric data and geodetic survey, which is a common and reliable practice to produce accurate elevation data in Vietnam (Minderhoud et al. 2019). Hence, this topographic data can be currently considered as the best available elevation data for the study area. The DEM was also used for SLR projections for the whole of Vietnam (Trần et al. 2016) and other studies on SLR impacts in the MD region (Dang et al. 2020, 2021a). Moreover, in a similar approach to our study, Payo et al. (2016) utilized a DEM with vertical accuracy of less than 1 m to produce relatively realistic simulation results. Nevertheless, it is possible to improve vertical accuracy in DEMs and therefore improve the accuracy of modelling. For example, LiDAR data can provide vertical accuracy of up to around 10 cm, and thus is recommended for model improvement (Clough et al. 2012; Wu et al. 2015).

Although the simple erosion rule in SLAMM seems to be effective in capturing the linearity of erosion processes in the study area, this function has some limitations, which must be taken into account in further studies. In particular,

Table 3 Importance level of inundation and erosion to mangrove loss in response to different SLR scenarios and net subsidence rates

SLR scenarios	Surface elevation change scenarios					
	Subsidence		Stable		Accretion	
	Inundation (%)	Erosion (%)	Inundation (%)	Erosion (%)	Inundation (%)	Erosion (%)
RCP2.6	83.79	16.21	54.33	45.67	16.72	83.28
RCP4.5	87.32	12.68	60.43	39.57	29.32	70.68
RCP8.5	90.34	9.66	73.09	26.91	39.31	60.69
More extreme scenario	94.46	5.54	91.88	8.12	79.90	20.10

Table 4 Summary of the SLAMM uncertainties and study limitation

No	Uncertainties of SLAMM/study	Potential effects on the modelled results
1	Insensitive to erosion rate	SLAMM is unlikely to be effective approach for regions with complex erosion issue
2	Wave energy is not adequately considered in SLAMM	
3	The use of fixed net subsidence rate for the entire study	The failure of SLAMM to simulate locally aggraded mangrove areas in comparison to observed data
4	Locally specific geomorphological processes were not properly captured in SLAMM	
5	Changes in hydrological regimes, storms, erosion rates and sediment supply due to climate change, SLR and dam construction under future SLR scenarios were not included in the model	Impacts on simulated results on mangrove changes under future climate and SLR scenarios
6	The impacts of climate change and local anthropogenic activities on mangrove are absent from the study	Incomplete picture of mangrove ecosystems under future as mangroves could be also impacted by these absent factors

wave energy, which is likely to play a critical role in shoreline erosion, particularly in areas with persistent winds, is not adequately considered in SLAMM (Wu et al. 2017). According to our calibration results, SLAMM also tends to be insensitive to erosion rates, which is also found in previous studies (Chu-Agor et al. 2010; Payo et al. 2016; Wu et al. 2017). For regions where erosion processes are non-linear and more complex, it is likely that the model would not project changes as accurately (Wu et al. 2015). In addition, horizontal accretion within coastal wetlands, which may impact simulations of mangrove changes in the study area, is not considered in SLAMM (Wolters et al. 2005; Clough et al. 2012; Li et al. 2015). These limitations of empirical models call for hybrid modelling approaches, which include empirical algorithms, and mechanistically simulated key wetland processes (viz., accretion and erosion) (Wu et al. 2017).

There are also several processes that critically influence coastal mangrove habitats which are not considered in SLAMM. These processes include positive feedback between mangrove growth and sedimentation (Gilman et al. 2008), changes in hydrological regimes, storms, erosion rates and sediment supply due to climate changes, SLR, and dam construction under future climate and SLR scenarios. The use of fixed net subsidence rate for the entire study area is also a limitation since subsidence or accretion rates spatially vary and are influenced by local coastal geomorphological processes. In particular, lands at Ca Mau Cape in the West Sea site are likely to prograde due to high accretion resulting from the redistribution of eroded sediment from the East Sea. These specific processes, which may modify the general subsidence rate, were not adequately captured in SLAMM, likely explaining the failure of SLAMM to simulate locally aggraded mangrove areas in comparison to the observed period of 2013/2020 (Dang et al. 2021b). It is also likely that subsidence rates will vary over time as boundary conditions change due to SLR and storms.

While the present study only focuses on the impacts of relative SLR on mangrove ecosystems, it is critical to acknowledge that other factors, such as climate change itself and local anthropogenic activities, also affect mangrove ecosystems in the region. Climate change causes increases in temperature, stronger seasonality in rainfall and temperature, and extreme events (i.e., storms, floods, heat waves, and droughts) leading to moisture stress, and hydrological and thermal disturbance (Gopalakrishnan et al. 2019). All these factors affect seedling survival, growth rates, and productivity (Krauss et al. 2008; Veettil et al. 2019b; Dang et al. 2021a; Salimi et al. 2021), and therefore reduce mangrove area. In addition, regional and local anthropogenic activities, such as development of aquaculture and agriculture, over-exploitation of natural resources, and over-fishing are likely to reduce mangrove coverage in the region (VNEPA 2005; Dinh 2016; Hong et al. 2019; Liu et al. 2020).

Despite these limitations, SLAMM is the only landscape model currently available for resource managers that includes essential processes controlling the impacts of SLR on coastal wetlands, such as subsidence and uplift processes, erosion, inundation, accretion, and overwash (Craft et al. 2009; Clough et al. 2012; Wu et al. 2015). Other advantages of this model include its capacity to incorporate uncertainty in spatial inputs (viz., vertical datum and DEM) and specify localized parameters (Clough et al. 2016). Importantly, the model has been shown to simulate wetland changes more accurately than other available methods, such as the growing cluster model (GrC) and the random constraint match model (RCM) (Wu et al. 2015). SLAMM is, therefore, potentially capable of simulating wetland changes by using high vertical accuracy of elevation data, accurate site-specific parameters, and meaningful evaluation of model accuracy (Wu et al. 2015; Mogensen & Rogers 2018). The present study used the best available DEM with reasonable vertical accuracy and precise localized parameters, and our assessment of model accuracy suggests an acceptable agreement between the modelled results and local observed data.

Implications for adaptation measures for conservation of mangrove ecosystems in the Mekong Delta

The potential spatial distribution of mangrove maps under various relative SLR scenarios produced in the present study are a useful resource for conservation and restoration of mangrove ecosystems in the region. Importantly, the maps can aid in identifying suitable areas for conservation and protection. In particular, the remaining mangrove area under high and extreme relative SLR scenarios is recommended as a high priority for protection and conservation. Measures such as the establishment of conservation reserves for these areas with effective buffer zones are valuable to protect core conservation mangroves from the effects of other non-conservation land uses. Autonomous adaptation should also be facilitated by establishing protected natural habitat corridors to support landward migration of mangrove species (Dang et al. 2021a).

Simulation results show that the East Sea shoreline is under persistent erosion risk, potentially leading to mangrove loss, and therefore reduction in the sheltering and wave buffering functions of mangroves. Adaption measures such as wave mitigation, accretion facilitation, and mangrove reforestation/restoration are recommended to not only protect the existing coastal areas from storm surges and erosion but also advance the shoreline to the sea in long-term strategies by taking advantage of river alluvium in the MD. To strengthen shorelines and minimize erosion, mangrove reforestation measures combining accretion facilitation solutions are recommended to be progressively implemented seaward (Tran Thi et al. 2014). Mangrove reforestation is considered the best solution for coastal erosion mitigation in the eastern coast of the MD as mangroves can reduce waves and trap sediment (Phan et al. 2015; Marchesiello et al. 2019); however, the approach does not seem sufficient enough in rapid eroding coasts exposed to high energy waves (Marchesiello et al. 2019). Therefore, wave reduction measures are recommended to be implemented in the short-term (Tran Thi et al. 2014). Infrastructure for wave mitigation, in particular sea dykes, can reduce wave energy; however, large-scale dyke construction potentially lead to the serious squeezing of coastal habitats and mangroves in the MD (Phan et al. 2015; Marchesiello et al. 2019). Hence, while implementing such measures, it is essential to pay critical consideration to mangrove areas and biodiversity conservation of saline and estuarine species (Tran Thi et al. 2014; Dinh 2016).

For the West Sea (the Gulf of Thailand) site, in particular, Ca Mau Cape, mangroves remain stable without erosion effects, and potentially support land aggradation due to accretion of river alluvium and eroded sediments derived from the East Sea within mangrove areas. These mangrove forests are likely to enhance shoreline stability and promote land propagation by trapping sediments, and so critically contribute to reduction of land subsidence and expansion of mangrove ecosystems in the region. In addition, the expanded area in the Ca Mau Cape region where newly formed mangroves are growing naturally can be also useful as laboratory sites for scientific activities on mangrove ecosystems (Tran Thi et al. 2014). These areas are, therefore, optimized zones for conservation and preservation of mangrove ecosystems for the region, which should be strictly protected and conserved while mangroves become established at higher elevations.

By 2100, SLR and subsidence are likely to inundate a substantial portion of the fringe mangroves in the study area. Importantly, our findings demonstrate the importance of processes controlling subsidence to the amount of mangrove loss. Hence, for long-term conservation of the ecosystems, it will be useful to develop measures to mitigate land subsidence to reduce the inundation risks.

Apart from SLR, increasing coastal urbanization leads to overexploitation of ground water extraction, which severely exacerbates subsidence in the MD (Minderhoud et al. 2017; Nicholls et al. 2021). Measures such as restriction of groundwater extraction, flood control, and sedimentation supply could be an effective solutions to minimize subsidence and loss of elevation on deltas, particularly in agricultural areas (Minderhoud et al. 2017; Nicholls et al. 2021). Factors such as maintaining coarse sediment supply and the general health of mangrove communities will likely be important. Coarse sediment provides the raw material for accretion to occur, while the mangroves themselves help to trap these sediments (Phan et al. 2015; Marchesiello et al. 2019) and restoration and maintenance of healthy mangroves via protections and management interventions will likely contribute to their sediment trapping capacity. In addition, land use strategies that provide mangrove ecosystems an avenue to migrate further inland are also necessary to preserve the fringe mangroves in the region. These strategies can provide managers with local “levers” to mitigate SLR impacts in the circumstance that there is little the local managers can do to reduce SLR other than as part of a global effort.

Conclusions

The current study projected SLR impacts on mangrove ecosystems in the south of the Mekong Delta, Vietnam, under different relative SLR scenarios by the year 2100, with the aid of the Sea Level Affects Marshes Model (SLAMM). The model is able to simulate the critical processes involved in the dynamics of coastal mangrove ecosystems. The use of the best available elevation data with high vertical accuracy and precisely localized site-specific parameters improved the simulation results. The accuracy assessment showed that our simulation results were in accord with previous modelling studies for mangroves in other regions and with observed data from the MD itself. The developed approach, therefore, demonstrates its capacity to simulate mangrove changes under future relative SLR scenarios in the study area despite SLAMM's uncertainties.

Our findings suggest considerable effects of SLR on mangrove ecosystems with the full effect being dependent net subsidence rate. Average annual mangrove losses are projected to be 0.54%, 0.22%, and 0.10% for the subsidence, stable, and accretion scenarios, respectively, by the year 2100 compared to the extents recorded in 2013. Inundation was identified as a key driving factor for mangrove loss in the MD by the year 2100, although erosion was more important when land surface elevation was increasing. The simulated mangrove distribution under future SLR scenarios is a useful resource to identify the most valuable areas for protection and conservation of mangroves. Importantly, the current study provides an innovative perspective on the SLR impacts on the critical mangrove ecosystems in the MD, which have not been taken into account in previous studies. Spatial information on mangrove habitats can significantly support resource managers and policy makers in developing appropriate adaptation and mitigation strategies for minimizing the SLR impacts on mangrove ecosystems and their important services, and for long-term mangrove protection and conservation in the region.

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Declarations

Conflict of interest The authors declare no competing interests.

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