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Invited review

Modeling wetland transitions and loss in coastal Louisiana under scenarios of future relative sea-level rise

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A B S T R A C T

This study uses modeling results for coastal Louisiana to examine spatial and temporal variation in future wetland loss, and how this variation is influenced by different causes of land loss represented in the modeled processes. Fifty-year model predictions illustrate specific vulnerabilities of the wetlands and the conditions under which they occur, e.g., long-term changes vs. specific events. Environmental scenarios were used to examine model sensitivity to changes in future patterns of precipitation, evapotranspiration, subsidence, and eustatic sea level rise. Based on the model results, the magnitude of wetland loss increases more than three-fold from low to high scenario. The model allows vegetation types to change over time as environmental conditions change. Each type is sensitive to different land-loss causing factors. Across all scenarios, the largest contributor to wetland loss is inundation loss of saline marsh ->40% of loss. Inundation loss of brackish marsh increases from low to high scenarios. Salinity induced loss of fresh wetlands increases from low to high scenario and coastwide contributes <10% of the total wetland loss. Marsh edge erosion is relatively consistent in magnitude across scenarios but its relative contribution decreases from low to high. Model outputs show two contrasting responses to environmental change over a 50-year simulation: a relatively linear response of land area over time, and a non-linear response where a large collapse event is triggered in a single year. Land loss varied dramatically over time within the 50-year simulations with little loss in the first two decades and high rates of loss 25–40 years into the future. Across most of the coast, and for all scenarios, the majority of land loss is caused by excessive inundation. Understanding the threshold conditions for inundation for different species and species mixtures is crucial to predictions of vegetation change, and subsequent wetland loss.

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1. Introduction

The ecological importance of coastal wetlands as habitat for fish and wildlife and modulators of water quality has long been recognized (Barbier et al., 2011; Mitsch et al., 2015). They provide food security and economic support for many coastal communities through habitat provision for commercial species as well as ecotourism opportunities (Gramling and Hagelman, 2005; Rahman et al., 2018). In some areas extensive wetland systems mitigate flooding by storing floodwaters or attenuating surge and waves from coastal storms (Glass et al., 2018; Stark et al., 2015; Wamsley et al., 2010). Most of these functions rely on the specific biogeomorphic character of the wetlands and their coastal context (Reed et al., 2018). Whether these functions continue in future decades, and to what degree, depends on how the wetlands themselves respond to changing coastal conditions.

Climate change and coastal and watershed development presents numerous challenges to coastal systems (Saintilan et al., 2018; Sanger et al., 2015; Spencer et al., 2016). Changes in sediment delivery from watersheds (Syvitski et al., 2005) and exacerbated flood or drought events (Dettinger, 2011; Keim et al., 2011) compound coastal changes such as subsidence (especially in deltaic areas (Syvitski et al., 2009)), sea-level rise (Sweet et al., 2017), salinity penetration (White and Kaplan, 2017) and altered hydrology due to human activities (Lee et al., 2006; Swenson and Turner, 1987). How these factors interact to alter the extent and character of coastal wetlands needs to be understood in order to predict whether they will still provide the functions and services on which many rely, and what interventions can be made to maintain functionality or manage transitions.

The future of coastal wetlands in the face of relative sea level rise has been a subject of extensive study for decades. Field studies have used measurements of marsh accretion and sedimentation over several years (Cahoon and Reed, 1995), decades (Williams, 2003) and longer (White et al., 2002) to compare to local rates of sea-level change, often based on tide gauge records (Cahoon, 2015). While many of these studies take a future perspective about whether wetlands will survive in the future often expressed in terms of elevation deficit or capital (Cahoon et al., 2018), they largely assume that recent or historical conditions, and the resulting accretion rates, will persist for decades to come. Combinations of measurements have allowed consideration of compaction and other within soil processes (Cahoon et al., 1995; Chambers et al., 2019; Jankowski et al., 2017), pointing to a more complex set of biogeophysical interactions that control surface elevation change. Numerical modeling studies have enabled evaluation of wetlands under various scenarios of future sea-level rise and sediment supply (Best et al., 2018; Fagherazzi et al., 2012; Spencer et al., 2016). However, the focus of many studies is on the survival of the existing wetlands, although several recent analyses have shown how the extent of wetlands is dependent on whether they survive in place and on the opportunity to migrate upslope (Schieder et al., 2018; Schuerch et al., 2018; Thorne et al., 2018).

Lateral erosion of coastal wetlands is common (Bendoni et al., 2016; Marani et al., 2011; Wilson and Allison, 2008), although in some systems, at least at the sub-decadal scale it has been seen as part of a cycle of alternate erosion and progradation (Koppel et al., 2005). The character of the flat-wetland interface is governed by a complex set of biogeomorphic interactions (Reed et al., 2018). Several authors have documented the relative importance of lateral erosion relative to interior loss due to submergence (Fagherazzi et al., 2013), while others have linked these processes as part of sediment budget analysis (Hopkinson et al., 2018; Li et al., 2018). In most coastal wetlands, lateral erosion depends on a different set of environmental factors, e.g., wind forcing, water depth, fetch, soil strength (Bendoni et al., 2016) than prolonged inundation due to an elevation deficit, e.g., sediment deposition, soil organic accumulation, vegetative growth/survival. Thus, these loss mechanisms, termed here marsh edge erosion (MEE) and inundation, respond differently to climate change and development pressures.

Moreover, many studies of coastal wetland geomorphic change have examined areas where vegetation is salt tolerant to some degree. In several systems, comparisons have been made between more saline and brackish or fresh zones of estuaries, e.g., the Western Scheldt (Temmerman et al., 2003), and Elbe (Butzeck et al., 2015). For most estuaries, there is an expectation that sea-level rise will lead to greater saline penetration and, depending on estuarine circulation, an increase in salinity in areas which are currently fresh (Hong and Shen, 2012; Yang et al., 2015). The consequences of potential shifts in salinity for coastal wetlands and how any changes in wetland character, e.g., species distribution, could modulate their ability to tolerate inundation, for example, has rarely been studied. Mesocosm and greenhouse experiments have been used to simulate the effects of salinity changes on coastal wetland plants. Some studies examine short-lived salinity pulses to simulate effects of hurricanes, e.g., (Howard and Mendelssohn, 2000). Li and Pennings (2018) found the response of fresh and brackish species to salinity pulses varied by species and with the duration of the pulse, with most species recovering from low salinity, short duration pulses (Li and Pennings, 2019). Wilson et al. (2018b) examined the response of marsh vegetation and soil in the field to experimental increases in salinity and found an interaction between salinity and water level in effects on ecosystem productivity. The interaction between increased flooding due to sea-level rise and increased salinity was explored in a greenhouse experimental study by Baustian and Mendelssohn (2018) who found that sulfide was significantly higher at 18 ppt as inundation increased, while at 36 ppt sulfide was equally high the two increased inundation level examined.

While field measurements and experiments provide insight into the response of vegetation to salinity or inundation stress (e.g., Snedden et al., 2015), there are few opportunities to observe wetland vegetation and morphological change as sea-levels rise and salinity changes. One exception is the Virginia Coast Reserve where Brinson et al. (1995) identified five 'ecosystem-level states' in a transition from upland forest to deep benthic habitats with transitions between states dependent upon both episodic disturbances and progressive abiotic stress. Another is coastal Louisiana, where extensive coastal wetlands across the estuarine gradient have been subject to extensive alterations in hydrology in the 20th century and ongoing subsidence and sea-level rise (Reed, 2002; Swenson and Turner, 1987; Yuill et al., 2009). The result of these changes has been massive loss of coastal wetlands to open water. Couvillion et al. (2017) estimated approximately 4833 km^2 land loss, mostly coastal wetlands, occurred between 1932 and 2016. However, extensive coastal wetlands remain (Table 1).

The future of these coastal wetlands is of concern due to their value as fish and wildlife habitat (Zimmerman et al., 2000), their ecological importance (Gosselink and Pendleton, 1984) and their potential contribution to storm surge attenuation (Barbier et al., 2013; Wamsley et al., 2010). Local management and restoration efforts began over 50 years ago. Federal recognition of the problem led to passage of the Coastal Wetlands Planning, Protection and Restoration Act (CWPPRA) in 1990 (Public Law 101–646, Title III). Following Hurricanes Katrina and Rita in 2005, the State of Louisiana established a coastal master plan process with revisions required every 5 years (recently changed to a 6-year cycle). The first plan was produced in 2007. The 2012 and 2017 Coastal Master Plans both utilized numerical models (see Methods) to predict change in the coastal landscape 50 years into the future. These analyses predict changes in hydrology, e.g., salinity and water level, transitions in vegetation type and whether wetlands are maintained or lost to open water, under different scenarios of sea-level rise, subsidence, precipitation and evapotranspiration. In the master planning process, potential projects are evaluated on their ability to sustain or rebuild wetlands over the 50-year simulation period. This study focuses on the future without additional restoration projects in place (Future Without Action – FWOA) and describes the processes and patterns of loss which restoration projects need to address if they are to provide benefit. As well as the coastal wetland restoration analyses which are the focus of this paper, the master planning process also includes storm-surge modeling and examines risk reduction projects (Cobell et al., 2013; Johnson, 2019).

This study uses modeling results from the 2017 master plan analysis for the coastal area of Louisiana and summarizes the information by coastal basin (Fig. 1). The goal is to examine spatial and temporal variation in future wetland loss, and how this variation is influenced by different causes of land loss that are represented in the modeled processes. As causes of land loss vary according to vegetation type, transitions in coastal wetland type as environmental conditions change over the 50 -year predictions are examined to illustrate specific vulnerabilities of the coastal system and the conditions under which they occur, e.g., long-term changes vs. specific events. Overall results at the coastwide scale for three future scenarios are presented, with specific examples from different coastal basins provided to illustrate how the influence of different causes of land loss changes as a result of different assumptions about future conditions and coastal landscape context.

2. Methods

2.1. Modeling approach

The Integrated Compartment Model (ICM) (White et al., 2018; White et al., 2019a, 2019b) is a planning-level model that is used for coastal zone planning and research within the state of Louisiana

Table 1
Area of coastal wetlands by type in Louisiana (Sasser et al., 2014).

Vegetation/Land type	Area (km^2)	%
Fresh Forested Wetlands	1881.00	12%
Fresh Marsh	3871.29	25%
Intermediate Marsh	4036.48	26%
Brackish Marsh	2953.97	19%
Saline Marsh	2953.97	19%

(Peyronnin et al., 2013). The ICM framework contains four subroutines that interact on an annual time step: a hydrologic and hydraulic model (ICM-Hydro) (Meselhe et al., 2013), a vegetation dynamics model (ICM-LAVegMod) (Visser and Duke-Sylvester, 2017), a barrier island morphology model (ICM-BIMODE) (Poff et al., 2017) and a wetland morphology/elevation change model (ICM-Morph) (Couvillion et al., 2013).

2.1.1. ICM overview

The key interactions underlying the analysis in this paper are summarized here with more information available in White et al. (2018, 2019a, 2019b). The ICM-Hydro subroutine is a link-node mass balance model that is capable of simulating: water level (stage), flow rate, salinity, water temperature, suspended sediment concentration, sediment deposition and resuspension within open water areas, sediment deposition on the marsh surface, and a variety of water quality/nutrient constituents (McCorquodale et al., 2017; Meselhe et al., 2013; White et al., 2017). The hydro component of the ICM was calibrated using four years (2010—2013) of daily water level, daily salinity, monthly suspended sediment and monthly water quality/nutrient data. The same data for another four year period (2006–2013) was used for independent model validation (see Supplementary Material Section 1 for Model Performance Tables for ICM-Hydro). The statistical tools and visual inspection between the model output and the field observations confirm the ability of the ICM model to capture spatial and temporal patterns. The model replicated the water level and salinity variations quite well; and to lesser extent the sediment and water quality parameters. The scarcity of sediment and water quality data limited our ability to improve the model performance. However, for a large scale planning-level model, the ICM captured the seasonal patterns sufficiently to discern patterns of long-term change.

ICM-Hydro provides inputs to ICM-LAVegMod vegetation subroutine (Visser and Duke-Sylvester, 2017) which determines the relatively likelihood that a wetland plant species currently on the modeled landscape would experience growth or mortality leading to a transition to bare ground, available for vegetation establishment as conditions change. It also allows more suitable species to establish based on hydrologic conditions if they already occur within a defined distance (reflecting species dispersal). Individual species modeled are then grouped into vegetated habitat type (fresh forest, fresh marsh, intermediate marsh, brackish marsh and salt marsh) (Supplementary Material Section 2) and the predominant type is assigned to the landscape. Outputs from ICM Hydro are also used in ICM-Morph which simulates annual elevation change and wetland collapse (transition from vegetated to open water) over time in response to hydrology, inorganic sediment deposition, and organic accumulation which is assigned by predominant vegetation type. ICM-Morph also allows land to be gained, e.g., in areas of deltaic development. However, without additional restoration projects, this is restricted to small areas of the coast and is not described here.

This analysis focuses on the loss of vegetated wetlands to open water (i.e., lack of emergent vegetation). Table 2 shows the threshold values used in the ICM to determine the occurrence of wetland loss during simulations.

The inundation collapse thresholds shown in Table 2 were chosen based on the coast wide mean inundation depth that coincides with vegetated biomass that is two standard deviations below the mean biomass (as represented by the normalized vegetation index) (Couvillion and Beck, 2013). Salinity thresholds were based on data from the Coastwide Reference Monitoring System (CRMS - https://lacoast.gov/crms/), a network of ~390 sites across coastal Louisiana) which was examined to identify salinities above which fresh marshes and fresh forested wetlands do not occur in coastal Louisiana.

The salinity and inundation collapse thresholds that define wetland loss (and conversely, persistence) in ICM-Morph mean that vertical land movement is an important model process in the ICM framework. Lowering of the land surface increases inundation, and increased tidal

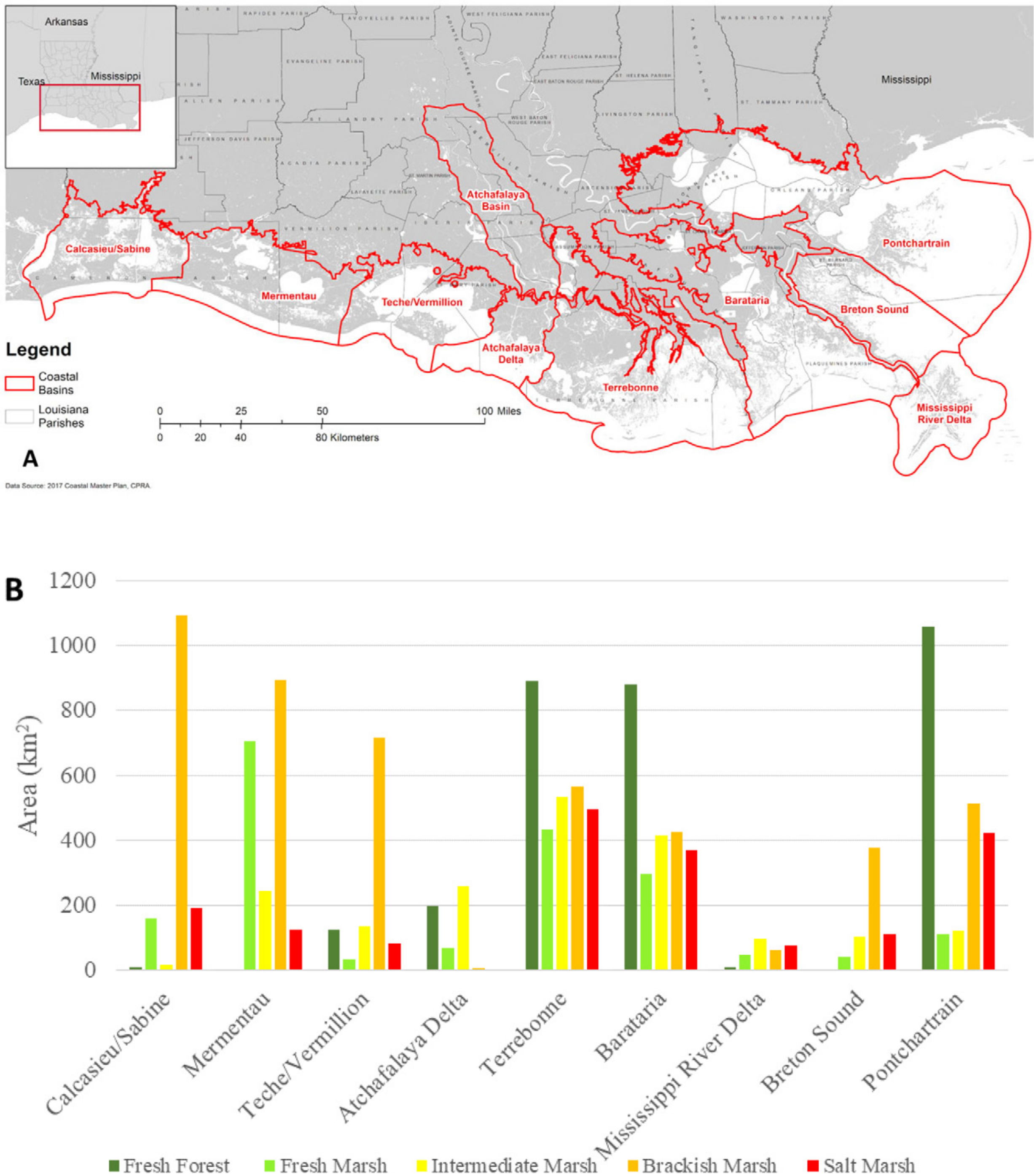

Fig. 1. Coastal area of Louisiana and the designated coastal basins (A) and the initial distribution of wetland types by basin (B).

Table 2
Thresholds for transition from wetland to open water used in the ICM.

Land type	Collapse threshold
Fresh Forested Wetlands	The maximum two-week mean salinity during the year is >7 ppt.
Fresh Marsh	The maximum two-week mean salinity during the year is >5.5 ppt.
Intermediate Marsh	Annual mean water depth over the marsh for two consecutive years is >0.36 m.
Brackish Marsh	Annual mean water depth over the marsh for two consecutive years is >0.26 m.
Saline Marsh	Annual mean water depth over the marsh for two consecutive years is >0.24 m.

prism leads to greater salinity penetration and potential for loss of fresh wetlands (White et al., 2019b). Negative vertical land movement (e.g., loss of elevation) is handled via a scenarios-based approach with varying rates of assumed subsidence values; these scenarios are described in more detail below. Positive vertical land movement (e.g. increases in elevation) of wetland soils are calculated via modeled accretion processes.

2.1.2. Sediment deposition and vertical accretion

Wetland accretion rates are modeled within the ICM by treating inorganic (mineral) sediment deposition and organic matter accretion as separate model processes. Mineral sediment deposition is calculated within the ICM-Hydro subroutine whereas organic accretion is calculated between the ICM-LAVegMod and ICM-Morph subroutines. For every model time step, sediment deposition and re-suspension rates are calculated for each model compartment; shear stress at the bed/water interface are calculated from flow and wind/wave forcings, and compared to critical shear stress thresholds for three particle size classes: sand, silt, and clay derived from previous studies as described in McCorquodale et al. (2017). If the shear stress at the bed is greater than the critical threshold, sediment re-suspension occurs and particles from an assumed erodible bed are suspended and assumed uniformly mixed in the water column. Sediment resuspension can only occur in open water; the marsh surface is not assumed to be erodible and any sediment deposited on the marsh will remain indefinitely.

Once suspended, the sediment particles are transported throughout the model domain via convection and dispersion mass balance routing. If the calculated shear stress at the bed is less than a particle size's critical threshold, suspended particles will deposit onto the bed surface (at which point they are added to the compartment's erodible bed depth available for resuspension at later model time steps). The mass of sediment deposited during each model time step is calculated as a function of the simulated water depth and the settling velocity of the mineral sediment particle size class. Sediment deposition on the marsh surface is calculated in an analogous manner: at every time step the inundation depth over the marsh surface is calculated and a deposited mass is determined from these depths and the settling velocities of the particles. Sediment particles that did not drop out of the water column during the time step are assumed to remain uniformly mixed within a compartment. During the next time step, the suspended sediment is transported back to the open water portion of the model compartment if the marsh is draining. Otherwise, it will remain in suspension over the marsh surface and will be available for deposition once again. Larger sediment particles (>20 μm) are assumed to be deposited within 30 m of the marsh edge (Christiansen et al., 2000), whereas finer sediments are assumed to be transported into the marsh interior.

Marsh edge, however, is assumed erodible perpendicular to the marsh shoreline. Historic marsh edge erosion rates were determined for 2004–2012 at over 1300 sites across the coast from aerial imagery. These rates were then applied throughout the 50-year simulations as an annual average rate (Allison et al., 2017). The linear retreat rates were converted into a volumetric rate using an assumed marsh scarp profile. The annual volume of edge erosion was assigned to masses of sand, silt, and clay based on the characteristics of the eroding marsh (bulk density, organic content, and mineral content – see discussion of CRMS soil data below). This sediment source was split between open water and marsh as a function of the marsh inundation frequency on an annual basis. Refer to McCorquodale et al. (2017) for a full discussion of how mineral sediment deposition, resuspension, and distribution throughout the model domain are handled within the ICM.

The mineral sediment deposition is modeled dynamically with ICM-Hydro, whereas the organic accretion rates are calculated via static lookup tables based on data-derived values of organic matter accumulation and soil bulk density for each of the five vegetation habitat types modeled by the ICM (fresh forested wetlands, fresh marsh, intermediate marsh, brackish marsh and saline marsh). CRMS dry bulk density, soil organic matter and marker horizon accretion data from the top 20 cm were calibrated with longer term data from Cs^{137} dated cores to represent the characteristics of soil at equilibrium (Couvillion et al., 2013). The resulting data were parsed by the five wetland types and by the nine coastal basins to ensure observed regional variations were accounted for, e.g., fresh marshes in the Birdsfoot Delta will have organic and bulk density values that differ from fresh marshes in the Terrebonne basin. As the hydrologic conditions change within the model and ICM-LAVegMod simulates changing vegetation communities, the assumed organic accretion will change resulting in vegetation-dependent vertical accretion from organic processes within the ICM.

2.2. Future conditions

Simulations were conducted for 50 years into the future. The initial topography is based on available LIDAR imagery with a 30 m resolution and RMSE of 7 cm in the vertical for the wetlands (Couvillion, 2017). A number of inputs to ICM-Hydro where historical data provided a reasonable bound for future conditions were represented by time series of boundary conditions. However, several environmental drivers were identified for which it was challenging to determine a set of values to drive the modeling effort. Some of these environmental drivers are influenced by climate change, e.g., eustatic sea level rise, and some are based on processes that are not fully understood, e.g., land subsidence. These were represented through the use of environmental scenarios.

2.2.1. Boundary conditions

The model runs for all scenarios analyzed here used identical time series for tributary freshwater inflows to the upstream model boundary. The 50-year observed water hydrograph for the Mississippi River at Tarbert Landing from 1964 through 2013 was used for the future 50-year simulations. River flow for the same time period for 36 tributaries to the coastal zone was developed from a combination of observed records and rating curves (Brown, 2017). Suspended inorganic sediment concentrations were derived for the Mississippi River based upon a separate sediment rating curves for sand and fines based developed from field sampling conducted in the Mississippi River at Belle Chasse (Allison et al., 2012). Gridded wind velocity and direction time series were compiled from the North American Regional Reanalysis climate dataset, time series of salinity concentrations, water and air temperature were developed from observed data samples (Brown, 2017). Wind, temperature and salinity time series were developed for an eight-year period used for model calibration/validation (2006–2013) and were repeated 6.25 times to compile a 50-year time series. The boundary condition time series for the 50 year simulations do not change by scenario but variation in conditions over time, e.g., the occurrence of flood and droughts in the record, provide insight on how such conditions influence change in the wetland landscape.

2.2.2. Scenarios

Environmental scenarios were used to represent potential changes in future patterns of precipitation, evapotranspiration, subsidence, and eustatic sea level rise (ESLR). The process used to develop the scenarios is described in detail in Meselhe et al. (2017).

Regional climate projections (Hostetler et al., 2011), together with historical records were used to determine a range of future precipitation and evapotranspiration conditions across coastal Louisiana (Habib et al., 2017). At one end of the range of conditions represented the scenario represents an approximate 14% increase in 50-year cumulative precipitation compared to historical data, and a 30% decrease in 50-year cumulative evapotranspiration compared to historical (calculated via Penman-Monteith). The scenario at the other end of the range of conditions uses historical mean monthly precipitation and potential evapotranspiration rates (Penman-Monteith). For precipitation and evapotranspiration, a sensitivity analysis based on the effect of values on future land loss was conducted to select scenario values for use in the 2017 Coastal Master Plan (Meselhe et al., 2017).

Three rates of ESLR were assumed for this analysis: 1.0, 1.5, and 2.0 m of ESLR by 2100 compared to 1992 sea level - each of which utilized different ESLR acceleration terms (Pahl, 2017). For the 50-year planning period used in this analysis, these scenarios correspond, respectively, to 0.43 m, 0.63 m, and 0.83 m of ESLR from 2015 through the end of 2064. While there is broad recognition that coastal Louisiana is experiencing relatively high rates of subsidence due to the geologic framework, ongoing natural processes and anthropogenic influence (Yuill et al., 2009) there is little agreement on relative process contributions and their spatial and temporal scales of influence. For previous planning efforts subsidence measurements were compiled and plausible ranges developed for regional subsidence within zones designated based on geological conditions (CPRA, 2012; Reed and Yuill, 2017). These ranges were used in this study to develop spatially variable subsidence rates across the model domain by selecting the 20th and 50th percentile values from across each of the plausible ranges for each zone. For example, the Birdsfoot Delta subsidence zone had a range of observed subsidence rates between 15 and 35 mm/year; therefore, the 20th percentile value was equal to 19 mm/yr (15 + 0.2*(35–15) = 19 mm/yr).

Table 3 shows how the values were combined across three scenarios used in the development of the Louisiana Coastal Master Plan.

3. Results

3.1. Coastwide analysis

3.1.1. Loss rates and mechanisms

Based on the model results, the magnitude of wetland loss to open water increases more than three-fold from the low scenario to the high scenario (Table 4). If the high scenario conditions occur, it would result in a wetland loss of over 10,000 km^2 in 50 years. Across all scenarios, the largest contributing mechanism to wetland loss is inundation loss of saline marsh, resulting in >40% of the loss. Inundation loss of brackish marsh increases markedly both in magnitude and relative contribution from low to high scenarios (Table 4), while inundation loss of intermediate marsh remains minimal across all scenarios. Salinity induced loss of fresh wetlands increases from the low to the high scenario and, at the coastwide scale, contributes <10% of the total wetland loss. MEE is relatively consistent in magnitude across the three scenarios but its relative contribution decreases from low to high.

3.1.2. Temporal variability: change over simulation period

Fig. 2 shows how the rate of land loss changes throughout the 50-year simulations for each scenario and how the different mechanisms vary in their relative contributions over time. For the low scenario (Fig. 2A) MEE contributes most of the loss through year 20 when inundation loss increases with rising sea-level. While there is interannual variability, inundation loss, mostly of saline marsh, increases over time with MEE decreasing. Loss of fresh wetlands due to salinity stress is periodic, making substantial contributions in years 24, 33, 34 and 43. Overall there is an increase in the rate of land loss over time for the low scenario.

The same general pattern holds under the medium scenario (Fig. 2B) although the magnitude of loss increases. MEE still dominates in the first two decades but the increased contribution of inundation loss occurs earlier (year 17). Inundation loss for all years is dominated by loss of saline marshes. Loss of fresh wetlands to salinity stress again contributes only in specific years, and is marked in years 24, 41, 43, and 44. In year 44, salinity-induced loss accounts for almost 64% of total coast wide wetland loss.

Fig. 2C shows results for the high scenario. The relative contribution of MEE decreases as the contributions of other mechanisms increase, after year 12 in this case, and is minimal in the last decade of the simulation. Similar to the low and medium scenarios, rates of loss vary among years but decrease in the last decade. The high scenario shows a greater relative contribution of loss of brackish marsh, which increases over time. Loss of fresh wetlands due to salinity stress occurs in more years than other scenarios and is greatest in year 24 and year 42.

Table 3
Values used in the low, medium and high scenarios.

Scenario	Precipitation	Evapotranspiration	ESLR (m/50 yr)	Subsidence
Low	>Historical	<Historical	0.43	20% of range
Medium	>Historical	Historical	0.63	20% of range
High	Historical	Historical	0.83	50% of range

Table 4
Cumulative Land Loss by Mechanism for the Low, Medium and High Scenarios.

Cumulative Loss over 50 years	Low scenario		Medium scenario		High scenario	
	km^2	%	km^2	%	km^2	%
Salinity Collapse	214.5	7.1	552.0	10.0	726.5	7.3
Inundation Collapse - Intermediate Marsh	41.7	1.4	64.5	1.2	121.1	1.2
Inundation Collapse - Brackish Marsh	599.6	20.0	1287.0	23.3	4001.2	40.1
Inundation Collapse - Saline Marsh	1289.6	42.9	2754.8	50.0	4269.5	42.8
Marsh Edge Erosion	859.1	28.6	856.6	15.5	867.9	8.7
Total	3004.5		5514.8		9986.2	

3.1.3. Spatial variability: change across the coast

The loss mechanisms shown in Fig. 2 and Table 4 are not evenly distributed across the coast. Fig. 3 shows the cumulative loss over 50 years for each coastal basin by loss mechanism for the high scenario (see Supplementary Material Section 3 for low and medium scenarios). Some basins show minimal loss of fresh wetlands due to salinity, e.g., Calcasieu/Sabine and the Atchafalaya Delta under the high scenario. However, those basins that do have substantial loss of wetlands due to salinity stress are not consistent across scenarios: Terrebonne has high salinity loss under medium and high scenarios, Barataria under the medium scenario, and Mermentau only under the low scenario. Fig. 3 shows that MEE contributes to wetland loss across the coast under the high scenario. For the low scenario MEE is the greatest source of loss in Mississippi River Delta, Breton and Pontchartrain Basin (Supplementary Material Section 3). Inundation of saline marshes is the greatest cause of loss in Calcasieu/Sabine, Mermentau, Teche/Vermilion, and Terrebonne under all scenarios. However, under the high scenario (Fig. 3) inundation loss of brackish marshes is the greatest contributor in Barataria, Breton and Pontchartrain Basins.

3.2. Mechanisms of loss

The spatial patterns in cumulative loss described above reflect complex interactions between local processes and changing environmental conditions over the 50-year simulation and among scenarios. These interactions are illustrated with selected examples from different parts of the coast.

3.2.1. Marsh edge erosion – Breton basin

The role of MEE can be illustrated using the Breton basin where it is an important cause of land loss across all three scenarios (Table 5). The amount of wetland loss attributed to edge erosion is remarkably consistent across the scenarios while inundation loss increases from low to high. However, examination of the contribution of MEE over time in Breton (see Supplementary Material Section 4) shows that while the annual rate of MEE remains fairly consistent over time for the low scenario, it decreases over time in the medium and high scenarios. Fig. 4 shows the pattern of loss across the Breton basin showing the fragmented nature of the landscape with many exposed shorelines subject to MEE. In the low scenario (Fig. 4A), when inundation loss is less significant, edge erosion continues over the length of the simulation as the islands and shorelines retreat. For the medium and high scenarios, the rate of MEE diminishes over time as the islands and other marshes are lost to inundation.

3.2.2. Salinity - Mermentau

Under the low scenario the Mermentau basin shows several years with pronounced loss of fresh wetlands due to salinity (Fig. 5). Years

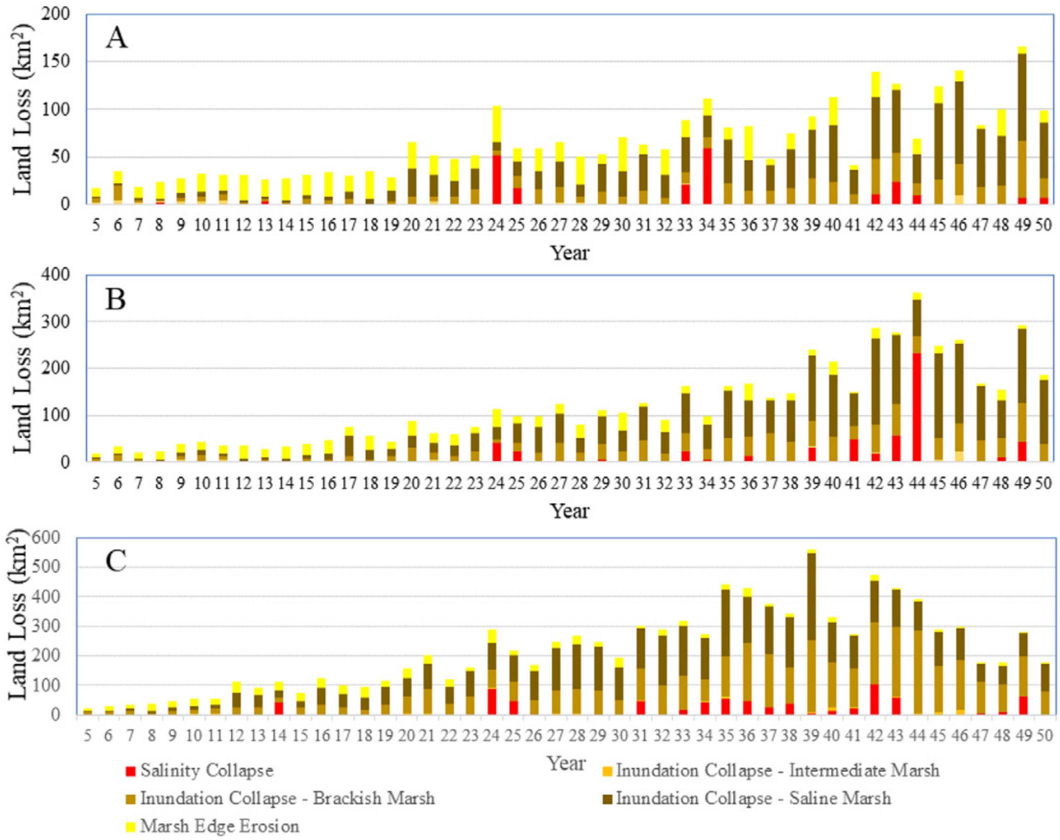

Fig. 2. Causes of land loss by year coastwide for (A) Low, (B) Medium and (C) High scenarios. Note change in axis scale.

24, 33, 34, 42 and 43 all experience elevated salinity concentrations that occur during abnormally dry summertime (June, July, and August) conditions in the Mermentau basin in the low scenario. Fig. 6 provides a snapshot of the summer time dryness conditions for a location in the SW Mermentau basin. As indicated by the vertical dashed lines, the five years with substantial salinity induced loss (24, 33, 34, 42, and 43) all correspond to conditions in which the net precipitation during the summer was very low or negative and the summer flow rates within the Mermentau River were below average. Note that the salinity values plotted in Fig. 6 are the maximum 14-day mean salinity values experienced during summer months; the actual salinity induced collapse threshold within the model uses this same 14-day mean salinity but is

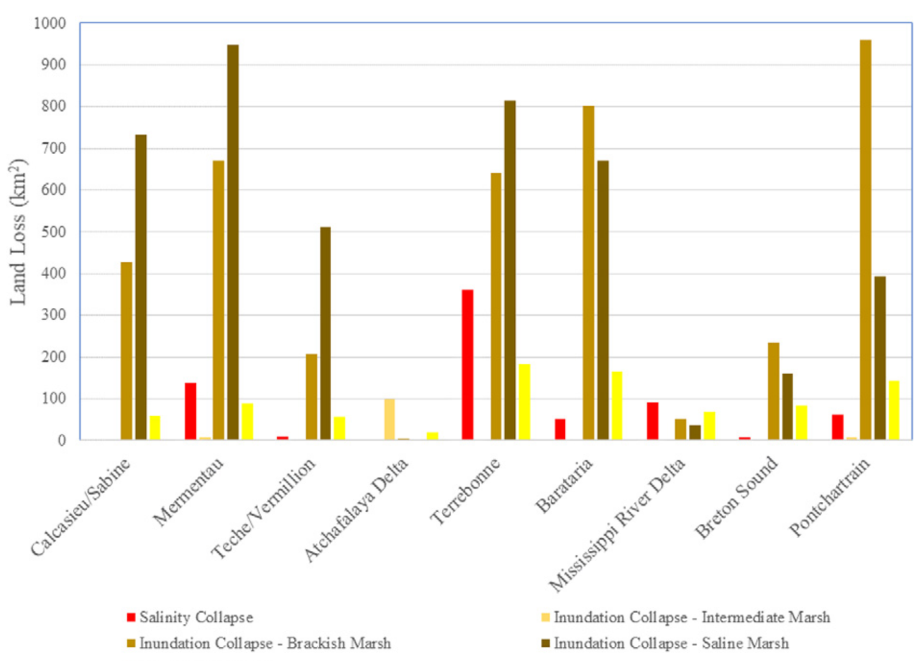

Fig. 3. Cumulative causes of land loss over 50 years by coastal basin - High Scenario.

Table 5
Cumulative land loss (km^2) over 50 years by causal mechanism for the three scenarios - Breton Sound Basin.

	Low scenario	Medium scenario	High scenario
Salinity Collapse	1.3	5.1	7.1
Inundation Collapse – Intermediate Marsh	0.1	0.1	0.1
Inundation Collapse – Brackish Marsh	56.4	140.3	235.7
Inundation Collapse – Saline Marsh	19.6	75.3	159.7
Marsh Edge Erosion	88.9	89.4	83.6
Total	166.3	310.2	486.2

the maximum throughout the entire year, not just during the summer. Additionally, the salinity plotted is for only one location (which experienced widespread collapse during year 24 – see Supplementary Material Section 5), other locations within the Mermentau basin would experience different salinity values in different years resulting in the salinity induced collapse seen in Fig. 5.

3.2.3. Inundation - Terrebonne

Thresholds of water depth that trigger loss of wetlands vary by vegetation type (Table 2) (Couvillion and Beck, 2013). In this analysis, vegetation types lost to inundation are tracked and their patterns of change over time are a result not only of increasing water levels but also vegetation transitions. Fig. 7 shows how total inundation loss changes over time in the Terrebonne basin and also how the relative roles of inundation loss of saline marsh diminishes over time and loss of brackish marshes increases.

Near the start of the simulation (Fig. 8 Year 5) the western Terrebonne basin shows extensive saline, brackish, intermediate marshes with some areas of fresh marsh. Inundation loss in year 5 is minimal (Fig. 7). By year 25 changing environmental conditions under the high scenario have resulted in the marshes in the northwestern part of the basin transitioning to fresh marshes, and the area of saline marsh have moved further inland at the expense of brackish marshes. Wetland loss due to inundation by year 25 is still dominated by inundation of saline marshes (Fig. 7) as the salt marsh front moves inland. By year 37 in the simulation (Fig. 8 lower panels) the brackish marsh zone has extended further north with much or the area that was brackish in year 5 having been lost to open water. In later years of the simulation (Fig. 7) inundation loss of brackish marsh increases over saline marshes as the brackish marshes impinge on the area that was previously fresh, and there are few saline marshes left to be lost.

4. Discussion

There is widespread recognition that globally coastal wetlands are under threat from environmental conditions. Previous studies have considered the importance of their location and extent in providing specific ecosystem services (Barbier et al., 2011) and their relative vulnerability under different rates of sea-level rise and subsidence (Schuerch et al., 2018; Spencer et al., 2016). This study has shown that the future loss of coastal wetlands cannot be predicted by focusing solely on their elevation in relation to mean sea-level or their exposure to wave erosion. This study has highlighted the importance of considering loss mechanisms and how they interact at the landscape scale, the importance of vegetative transitions, and has provided insight into the temporal and spatial scales of vulnerability.

4.1. Loss mechanisms

The three main loss mechanisms considered in this analysis are edge erosion of wetlands adjacent to major water bodies, salinity stress leading to loss of freshwater wetlands, and exceedance of inundation thresholds. In this work, MEE is less sensitive to environmental conditions and progressively retreats shorelines adjacent to water bodies but only as long as those shorelines exist. The example of Breton Sound provided here shows how the interaction of MEE and inundation loss leads to a dramatically different landscape under the high scenario than under the low scenario when inundation loss is minimal. Several authors have pointed to the importance of MEE in releasing sediment which can then be made available for accretion on adjacent wetlands, limiting their vulnerability to inundation loss and evidence to support this ‘cannibalism’ of marsh sediment is available from several localized studies (Duvall et al., 2019; Hopkinson et al., 2018; Reed, 1988). The ICM tracks the release of sediment due to MEE and accounts for it in the sediment budget. This may better enable wetlands in basins with extensive shorelines, such as Breton, to maintain relative elevation and low inundation levels under scenarios with lower rates of sea-level rise and subsidence. However, at the landscape scale, the sediment demand to maintain elevations is unlikely to be supplied by these ‘internal sources.

At the landscape scale the three causes of loss cumulatively change the configuration of coastal basins. The example from the Terrebonne basin shows how marsh edge erosion and inundation loss lead to extensive loss of wetland (Fig. 8), open up tidal exchange, and enhance saline incursion from the Gulf. This interaction among loss mechanisms varies with scenario and coastal configuration. The lower Breton basin is already fragmented (Fig. 4) and open to exchange with large coastal embayments and the Gulf (Fig. 1). However, in basins which are currently more restricted in terms of exchange, the effect of wetland loss in the

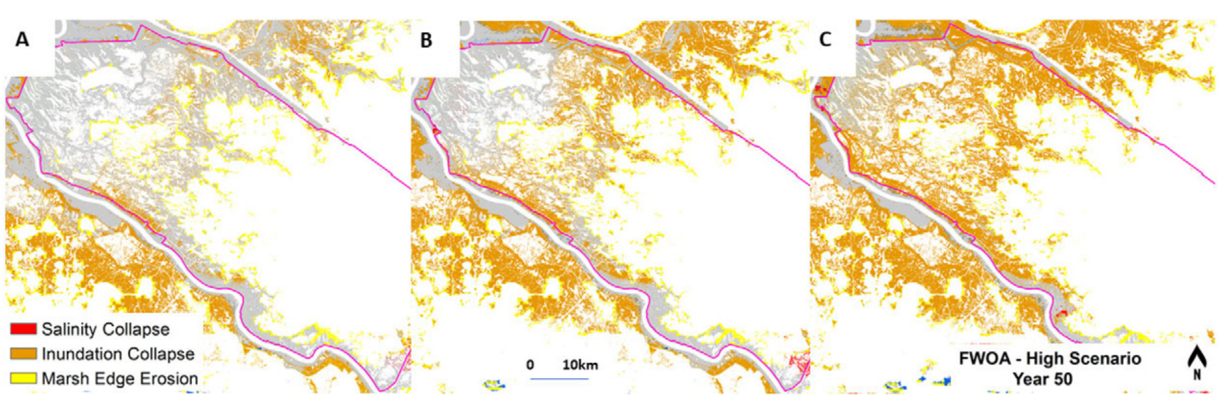

Fig. 4. Cumulative land loss over 50 years for the Breton Sound Basin for (A) Low, (B) Medium and (C) High Scenarios. Pink line shows basin boundary.

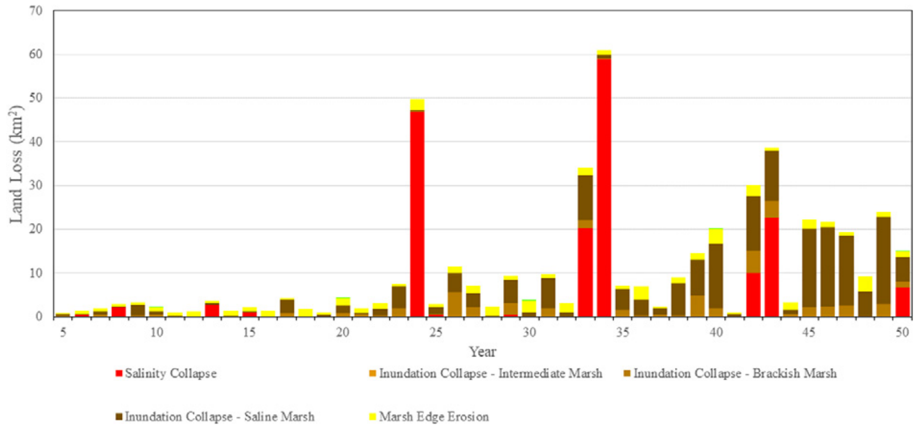

Fig. 5. Causes of land loss by year - Mermentau Basin, Low Scenario.

buffer region currently protecting these basins from the Gulf can increase tidal exchange and salinity penetration, further exacerbating loss due to inundation and salinity. Fig. 9 shows an example for the upper Barataria Basin. In the low scenario salinity shows interannual variation and some increase in the minimum and increasing peak values in the last decade of the simulation. In the medium scenario, patterns are similar through approximately year 30 when salinities begin to trend upward and fluctuate dramatically in the last decade. These spikes in salinity in the medium scenario lead to the increase in salinity loss in Barataria (see Supplementary Material Section 6). In the high scenarios the spikes are less pronounced as salinity is higher and greater wetland loss in the basin leads to extensive open water and penetration of salinity further into the basin.

The loss through one mechanism compounding loss through another mechanism has not previously been demonstrated at such a spatial-temporal scale. Laboratory experiments of coastal wetland plant survival in the face of salinity penetration have also examined how changes in inundation modulate that response. For example, Wilson et al. (2018a) showed differences in carbon flux from experimentally treated Everglades peat soils when treated with salinity, under drawn and inundated conditions. Chambers et al. (2014) found that a certain level of soil inundation reduced organic carbon loss from

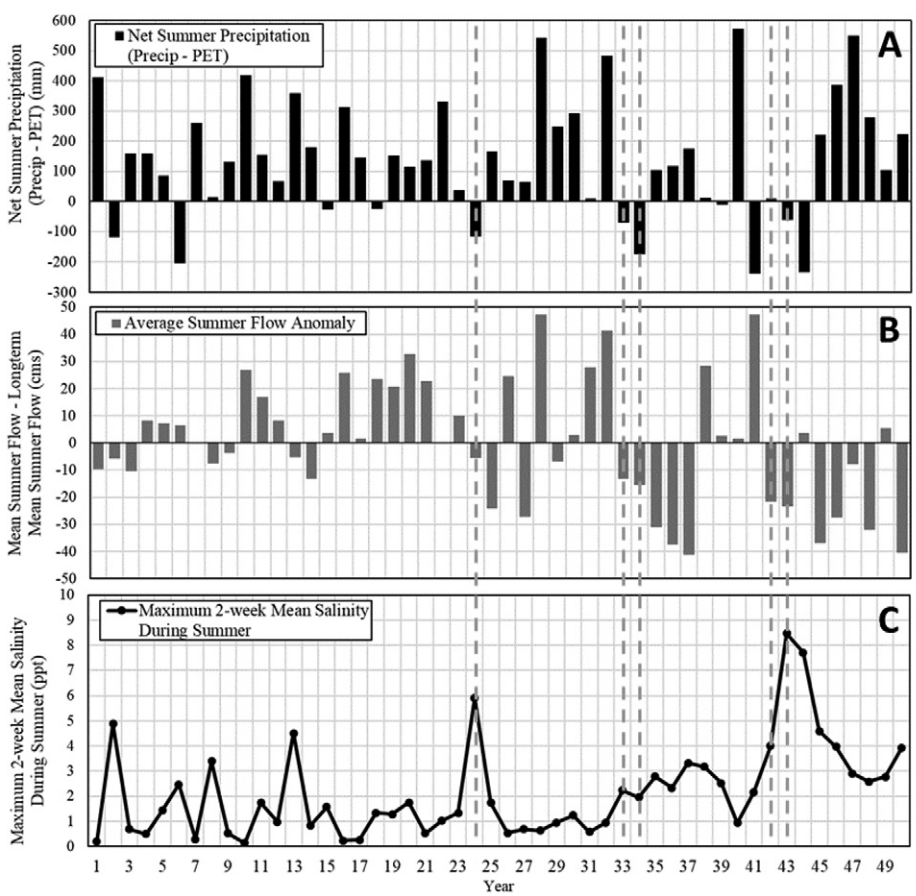

Fig. 6. A. Net precipitation (rainfall minus potential evapotranspiration) for summer months (June, July, August) in SW Mermentau Basin - negative values indicate more evapotranspiration than precipitation (black/top). B. Average summer flowrate anomaly (annual summer mean minus long-term summer mean) for Mermentau River - negative values indicate less summer flow than average (grey/middle). C. Maximum2-week mean salinity during summer for SW Mermentau Basin. Vertical dashed lines indicate years with below average summer flows in the Mermentau, net negative (or near zero) summer precipitation, and substantial areas of salinity induced collapse in the low scenario (Fig. 5).

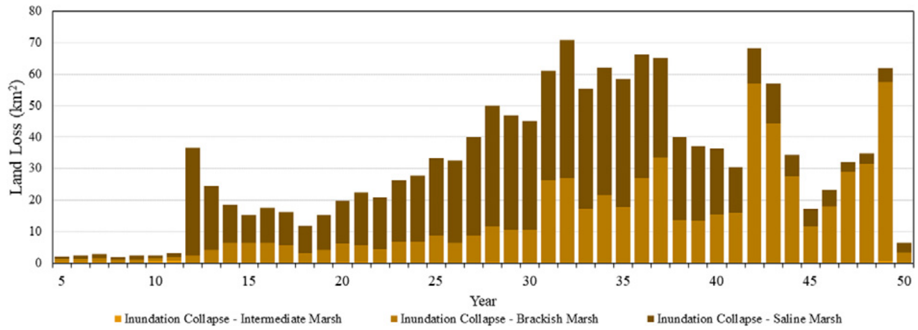

Fig. 7. Variation in inundation loss over time - Terrebonne Basin High Scenario.

mangrove soils. The types of inundation treatments used in laboratory or mesocosm experiments (typically 10-15 cm of flooding) are much lower than the thresholds for loss used in this analysis (Table 1). This illustrates the challenge of linking experimental scale results with landscape dynamics and interannual variability. The analysis presented here shows how decadal scale shifts in landscape configuration can have important feedback to wetland character and survival.

4.2. Importance of vegetation transitions

The ICM-LAVegMod (Visser and Duke-Sylvester, 2017) allows vegetation types to change over time as environmental conditions change. This allows progressive changes in salinity and water level, that do not exceed the thresholds shown in Table 2, to gradually adjust the distribution of vegetation types within the coastal basins. As shown in the example from Terrebonne basin, over decades, extensive areas of marsh can change in vegetation type without converting to open water. Visser and Duke-Sylvester (2017) showed how different projects designed to alter freshwater distribution throughout the Breton basin allowed fresh marshes to increase in extent, compared to conditions without the project, but these marshes are then more vulnerable to increased salinity in later years than those that gradually converted to brackish marsh.

Both the vegetative transitions used in ICM-LAVegMod and the thresholds for loss of wetlands (Table 2) are based on the species commonly found in Louisiana coastal wetlands. In other systems, species may be more or less tolerant to changing environmental conditions. Li and Pennings (2019) used a mix of fresh tidal marsh species from the Altamaha River estuary, Georgia, USA, and showed that with increased salinity levels and increasing duration of the salinity pulses to which the mesocosms were exposed, plant community composition diverged from the control treatment. They also note a switch toward more salt tolerant species with higher salinities and longer pulses. While the salinity treatments affected all their fresh marsh species, the response did vary by species. In a related experiment, Li and Pennings (2018) noted that *Zizaniopsis miliacea* was more tolerant of salinity pulses than *Polygonum hydropiperoides* and *Pontederia cordata*, showing a species-specific response for brackish species from the same estuary.

Understanding these types of vegetative transitions may be less important in areas of salt marsh bordering open coastal areas with no appreciable salinity gradient (e.g., the Essex coast of the UK and back-barrier marshes on the US Mid-Atlantic coast). But in estuarine areas, such as coastal Louisiana, Chesapeake Bay, San Francisco Bay, and the western Scheldt they should not be ignored in decadal scale predictions of wetland survival. The vegetative transitions in ICM-LAVegMod were established using monitoring data from hundreds of sampling stations across coastal Louisiana established in 2007 (https://www.lacoast.gov/crms/Home.aspx) (Visser et al., 2013). With fewer sampling stations, Troxler et al. (2014) used extensive data from the Everglades monitoring system to document decadal scale changes in the distribution on growth of *Cladium jamaicense* and *Eleocharis cellulosa* in response to landscape scale changes in hydrology. There are few wetland systems with such extensive or enduring monitoring programs that can provide information needed to predict landscape scale environmental change. The analysis presented here reinforces the need for additional studies to understand vegetative transitions in order to predict the fate of coastal wetlands.

4.3. Time scales and causes of vulnerability

The model outputs presented here demonstrate two contrasting responses to environmental change over a 50-year time horizon. The first is a relatively linear response of land area over time, and the second is a non-linear (e.g., step-wise) response in which a large collapse event is triggered in a single year. The first response, a gradual and continuous reduction over time is the result of inundation collapse mechanisms in which the increasing rates of relative sea level rise over time progressively increases inundation depth over the coastal wetlands. Inundation collapse within the ICM is only initiated if the wetland surface is inundated to a depth greater than the collapse threshold depth for two consecutive years. This collapse mechanism is, therefore, representative of continuous trends and the resulting land area response is, more or less, a continuous, linear relationship.

The second, step-wise, response is due to singular events impacting large portions of wetland. This is triggered solely by salinity-induced collapse of fresh wetland areas due to a short-term spike in salinity. The effect shown in the model is similar to the rapid decline in plant biomass identified in laboaratory studies of fresh marsh species tolerance of acute salinity exposure (e.g., Howard and Mendelssohn, 2000; Li and Pennings, 2018). Field observations of this type of effects are limited to hurricane impacts (e.g., Neyland, 2007) rather than the combination of sea-level rise and changing basin conditions. This step-wise collapse event can only occur on fresh wetlands, and is therefore more evident in regions of the coast dominated by fresh wetland systems. Areas which remain fresh for some decades due, for example, to their position in the basin or proximity to a riverine source of freshwater are then more vulnerable to salt-spike collapse thresholds during later years as the sea level rises, and the model domain becomes increasingly hydraulically connected. In areas with sustained freshwater inputs, such as the Atchafalaya Delta, salinity loss is minimal even under the high scenario (Fig. 3).

Further, this analysis revealed that conditions other than sea-level rise can induce salinity loss of freshwater wetlands. The example above from the Mermentau basin shows how local drought conditions combined with periodic low river inflows can cause salinity to increase. The salinity loss shown in Fig. 5 is for the low scenario, which represents relatively low rates of sea-level rise. The consideration of climate factors other than sea-level, and their interacting influence on vegetation

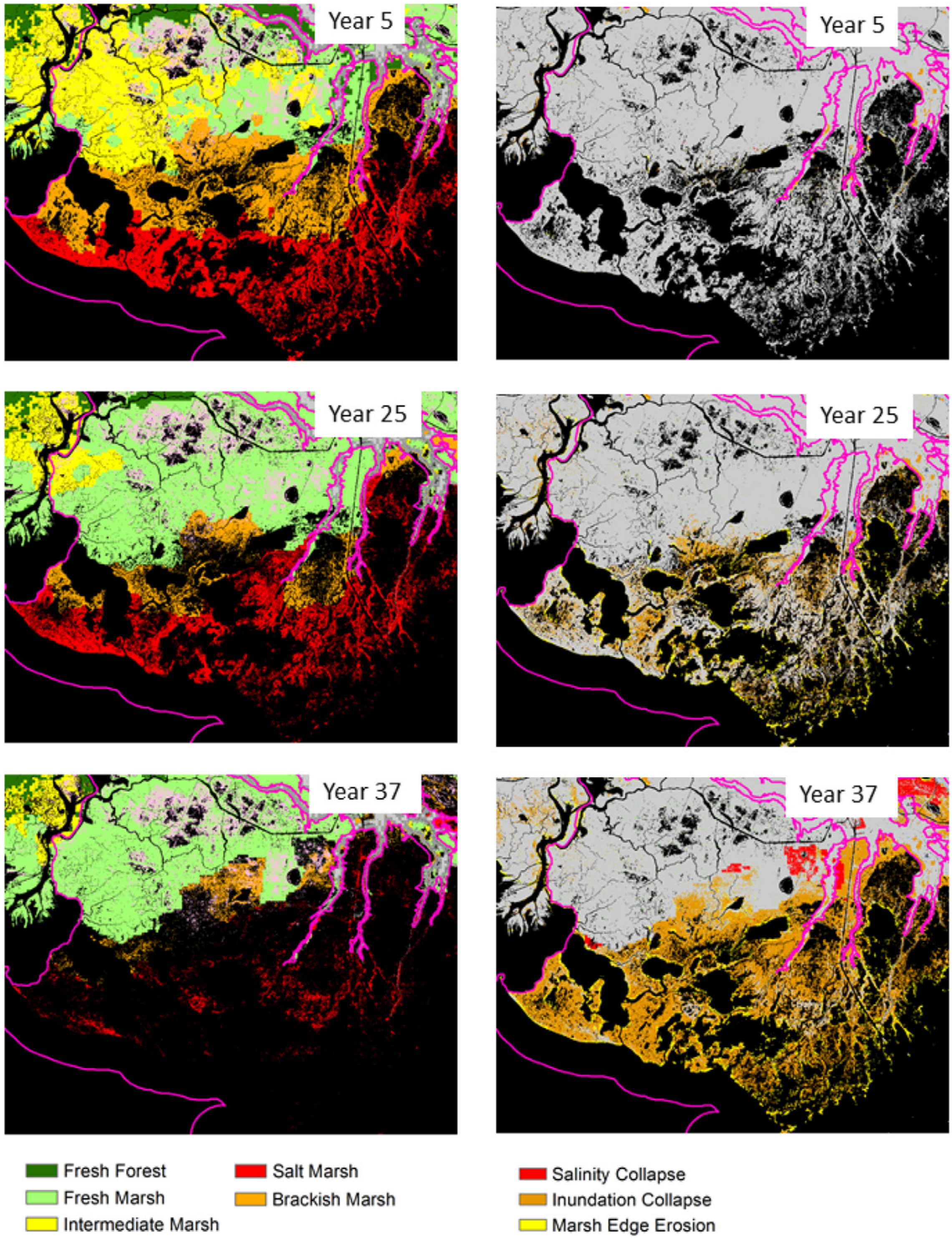

Fig. 8. Vegetation (left) and land loss mechanisms (right) for the Terrebonne basin for selected years for the high scenario. Note all types of inundation loss are combined. Pink line shows basin boundary.

change and wetland survival, is necessary for consideration of future wetland change in estuaries and low-lying coastal systems.

4.4. Addressing future wetland loss

This analysis was conducted using outputs from 'Future Without Action' predictions for coastal Louisiana that assume no new actions are implemented to address changing conditions or coastal land loss. The original motivation for the study was to enable the development of restoration projects or management actions tailored to specific causes of loss across the coast. Details about the effects of specific restoration measures are beyond the scope of this paper, but some general insights can be obtained which can be of use beyond Louisiana.

The change in relative importance of marsh edge erosion over time under conditions of high sea-level rise (Fig. 2) shows that addressing edge erosion through shore protection measures such as bulkheads

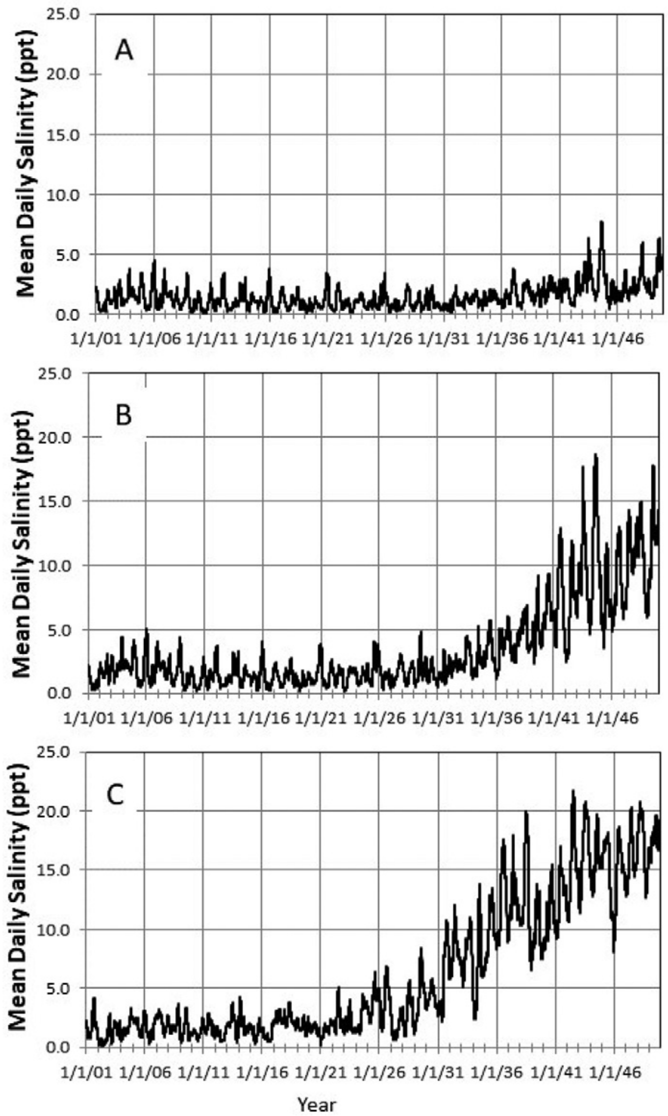

Fig. 9. Mean daily salinity for an ICM Hydro compartment in the upper Barataria Basin for (A) low, (B) medium, and (C) high scenarios.

(Gittman et al., 2015) or living shorelines (Scyphers et al., 2011; Smith et al., 2018) may sustain wetland area in the short-term but is not a long-term solution if the wetlands are vulnerable to inundation loss. Further, protection of shorelines from erosion eliminates or reduces the sediment yield from that erosion, potentially making adjacent wetlands more vulnerable.

The loss of freshwater wetlands due to spikes in salinity can be offset by increased or sustained freshwater inflows. In this study, areas influenced by the Atchafalaya River (mean annual flow 6400 $m^3 s^{-1}$) did not experience this loss even under high rates of sea-level rise. However, maintaining or increasing freshwater flows into estuaries to maintain wetlands needs to be balanced against other competing needs for freshwater, e.g., agricultural, municipal and industrial needs (Lund, 2015). In many deltaic systems, concern about river management and impoundment focuses on the disruption of sediment supply. Syvitski et al. (2009) noted the global scale of modifications to deltas and the balance between human-induced subsidence and reduced sediment load. Giosan et al. (2013) found that management within the Danube delta has redistributed sediments to interior delta areas enabling them to be maintained under sea-level rise while more distal areas of the delta have suffered. Strategic redistribution of river freshwater and sediment is the aim of several planned sediment diversion projects on the Mississippi River (Allison and Meselhe, 2010) with increased freshwater seen as potentially damaging to estuarine species (Peyronnin et al., 2017; White et al., 2018). Concern for freshwater inflows is often focused on maintaining conditions for seafood harvest, e.g., oysters (Fisch and Pine, 2016), or to protect native fishes, e.g., Delta Smelt (Moyle et al., 2018). This analysis has demonstrated that for the Louisiana coast, maintaining estuarine wetlands, and the services they provide, requires both sediment and a reliable source of freshwater.

5. Conclusions

This study has provided several insights concerning the future of coastal wetland landscapes in the 21st century. Marsh edge erosion is locally important, but dominates only in a few areas. In most areas, its effects diminish over time as the land being eroded at the margins is lost to other causes of loss. Loss of fresh wetlands to salinity incursion was found to be localized and often associated with particular combinations of environmental conditions. Land loss varied dramatically over time within the 50-year simulations with little loss in the first two decades and high rates of loss 25–40 years into the future. This response could be attributed to the nonlinearity of the eustatic sea level rise rates; where the rates are lower earlier in the 50-year time period and increases dramatically as time passes. In several basins land loss decreases in the last decade of the simulation, especially under the high scenario, as little land was left to be lost.

In most basins, and for all scenarios, the majority of land loss is caused by excessive inundation. Understanding the threshold conditions for inundation for different species and species mixtures is crucial to predictions of wetland change. This modeling study used thresholds based on mean annual water level, but it is likely that responses to depth are also dependent on the duration of flooding, and potentially duration of soil drainage. Existing studies have shown these responses to be species specific, and that vegetation shifts over time in response to change in environmental conditions. This study has shown the importance of considering lateral transition in wetland types across estuarine landscapes as well as vertical positioning in the tidal frame.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geomorph.2019.106991.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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