Changes in the Delivery of Ecosystem Services in Galveston Bay, TX, under a Sea-Level Rise Scenario

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Abstract

In this document, we quantify the coastal protection, fisheries and carbon storage and sequestration services delivered by wetlands in Galveston Bay, currently and under a 1 m sea-level rise through 2100. During coastal storms, wetlands have the capacity to store millions of cubic meters of storm water and attenuate total water levels. They also can avoid millions of dollars in property damage and protect hundreds of people against the impact of storms. Those services can continue to be delivered as sea-level rises, as long as wetlands are allowed to migrate. In addition, wetlands currently avoid millions of dollars of environmental damages by sequestering carbon. However, if sea level rises 1 m by 2100, the amount of CO_2 released in the atmosphere by defunct wetlands might cause millions of dollars of environmental damages. Finally, it appears that most fisheries species might benefit, in the form of increased populations, from more available wetland habitats as sea level rises.

1   Introduction

Galveston Bay, TX (the Bay) and its surroundings are home to more than half a million people, excluding the greater Houston area (USCB 2013). The low-lying areas surrounding the Bay include more than 1,000 km^2 of terrestrial, freshwater and coastal wetlands (WPC 2011), which provide a suite of diverse and valuable services, or benefits, to communities in the Gulf of Mexico. Wetlands have the potential to reduce storm surges, wave height, and the height of structures required to protect people and property from coastal hazards (Wamsley et al. 2010; Shepard et al. 2011; The Bay Institute 2013). Wetlands can also maintain and improve the health of key fisheries (Zimmerman et al. 2002; Minello et al. 2008; Bohannon 2013), provide habitat for valued bird species (Lester & Gonzalez 2011; U.S. Fish and Wildlife Service 1997), and provide recreational opportunities (Lester & Gonzalez 2011; Yoskowitz et al. 2012). Additionally, they can store and sequester important quantities of carbon (Siikamäki et al. 2013; Feagin et al. 2010).

Global climate change impacts, in the form of increases in air and water temperatures, and ocean acidity (IPCC 2013) have the potential to negatively affect coastal wetlands and the ecosystem services they deliver (Burkett & Kusler 2000; Erwin 2008; Millennium Ecosystem Assessment 2005). However, one of the most pressing and visible threats to the bay and its residents is the rise in sea level that it experiences (Rice 2013; Yoskowitz et al. 2009). Galveston Bay experienced substantial land loss due to coastal development, shoreline erosion, land subsidence and sea-level rise (SLR) (Yoskowitz et al. 2009). In turn, these shoreline changes make the region even more vulnerable to sea-level rise, which might lead to higher storm-surge-levels and inland inundation levels (Leatherman 1984; Warner & Tissot 2012). Impacts from these stressors could result in the displacement of more than 100,000 households and create more

than $12 billion in infrastructure losses (Yoskowitz et al. 2009). Consequently, it is likely that shoreline protection measures will be implemented in order to limit the impacts of coastal hazards on communities.

Typical coastal protective measures have been the construction of seawalls and other structural/engineered solutions. For instance, the 17 feet high, 10 miles long Galveston Seawall, built in response to the deadly Hurricane of 1900, was once one of the longest in the world. While such structures can provide protection against storm inundation they can be costly and require constant maintenance and upgrade (USACE 2002), especially in the face of a changing climate (Sekimoto et al. 2013). Shore-parallel structures also have adverse impacts on nearshore environments. In Galveston, they increase the erosion of adjacent shoreline (Leatherman 1984; Wallace et al. 2009) and fix the shoreline in place, which can yield deeper water depths nearshore and the disappearance of intertidal habitats (Dugan & Hubbard 2009; USACE 2002; Miles et al. 2001). In addition, by creating a barrier between marsh areas and their surrounding environment, shoreline structures can lead to the modification and/or disappearance of certain types of coastal wetlands that depend on regular or irregular exposure to salt or brackish waters (Bozek & Burdick 2005; Mattheus et al. 2010). These losses can negatively impact the amount and quality of ecosystem services provided by these important habitats.

Ecosystem services, and their continuous delivery, are essential for the health and long-term sustainability of communities. However, as mentioned above, communities sometimes put in place protective measures that alter the delivery of services to coastal communities. Consequently, it is important to identify and quantify the different services delivered by natural habitats in order to properly account for their loss when those habitats are removed or degraded directly by humans or indirectly by climate change impacts.

In this study, we quantify the changes in the delivery of three ecosystem services – coastal protection, carbon storage and sequestration, and fisheries – in Galveston Bay, TX, under a 1 m by 2100 sea-level rise scenario. We also estimate the amount of avoided damages and the number of people protected by wetlands against the impacts of storms under current and future environmental conditions. We compute the amount and value of carbon stored, sequestered and re-emitted in the Bay. Finally, we explore changes in the amount of habitat for a number of important fisheries in the Bay and model expected changes in brown shrimp landings.

The report is organized as follows. In Section 2, we present the physical setting of Galveston Bay, discuss climate stressors, and introduce some of the ecosystem services the Bay's wetlands deliver to people. In Section 3 we present the scenarios we chose to explore and the methods we used to quantify changes in the delivery of coastal benefits under those scenarios. In Section 4 we present the results of our analyses. In Section 5 we introduce the protection and restoration of oyster reefs as a mechanism for minimizing the adverse impacts of sea level rise on wetlands. We present conclusions and recommendations in Section 6.

2 Physical Setting

2.1 Geophysical setting

Galveston Bay (the Bay) is sheltered from the Gulf of Mexico by Galveston Island and the Bolivar Peninsula. It is the 7th largest estuary in the U.S., and home to the Port of Houston, the 2nd busiest port in the U.S. in terms of tonnage (Port of Houston Authority 2012). This 1,600 km^2 wide, shallow, micro-tidal estuary has an approximate tidal range of 0.62 m [MHHW is 0.28 m above MSL (NOAA-CSC, 2013)] and an average depth of 2.4 m. The Bay can be separated into 4 distinct regions: West Bay, East Bay, Trinity Bay and Upper Galveston Bay (Figure 1).

Twenty percent of the Bay's shores are protected by man-made structures, such as seawalls or riprap revetments (NOAA, 2012). The other major shoreline types are gravel and sandy beaches (10%), muddy shore banks (15%) and wetlands (52%) (Figure 2). Wetlands, oyster reefs, and seagrasses are the primary living coastal habitats in the Bay. Seagrasses currently cover only 0.1% of the Bay's bed (1.7 out of 1,550 km^2). Although the amount of seagrass in the Bay has increased slightly in the recent past, it is unlikely that gains will be sufficient enough to match its historical coverage of more than 10 km^2 (Dahl 2011; Lester & Gonzalez 2011). Thus, we focus our analyses on oyster reefs and wetlands.
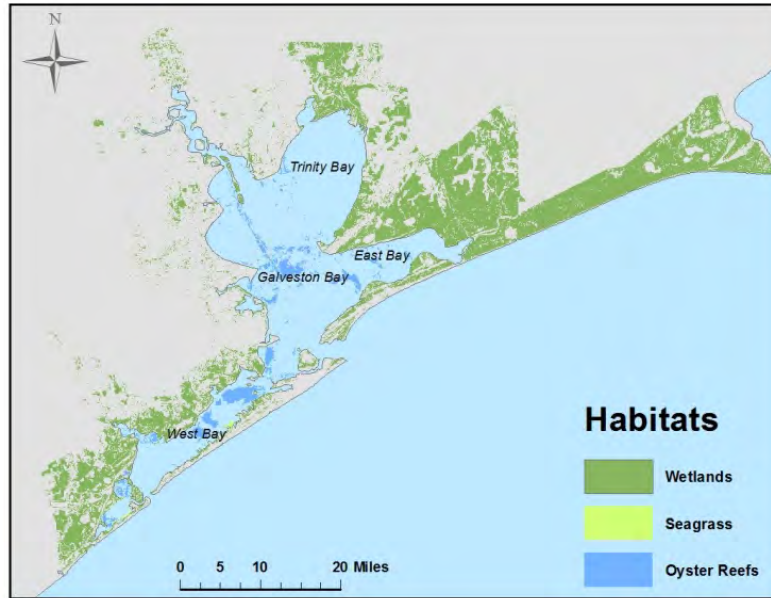
Figure 1: Living Habitats in Galveston Bay

Most of the Bay's oyster reefs are subtidal and located in the middle of the Bay, more than 1 km from the shore. Reefs occupy approximately 6.5% of the Bay's bed (TPWD 2010), and this number is slowly increasing thanks to recent restoration efforts by the Texas Parks and Wildlife Department and others. Oyster reefs are typically severely impacted by hurricanes: over 60% of the oyster reefs in Galveston Bay were destroyed during Hurricane Ike, and despite extensive ongoing restoration efforts, reefs have still not recovered (Haby et al. 2009). We explore the role of oyster reefs in protecting marshes in Section 5.

Estuarine wetlands cover approximately 570 km^2 of the land in and around Galveston Bay (Figure 1), roughly half of the Bay's water surface area. These wetlands are primarily composed of salt marsh and brackish marsh communities (Appendix A), which are common habitats for many fisheries species in the Bay. Although the current acreage of marsh has declined as compared to coverage in the 1950's, groundwater pumping regulations and habitat

conservation and restoration efforts have slowed, if not halted, the rate of salt marsh loss (Lester & Gonzalez 2011). Given that development is forecasted to continue to occur within or directly adjacent to existing development (H-GAC 2013), most of the future marsh losses are likely to be fragmented marshes near current population centers (Appendix B).
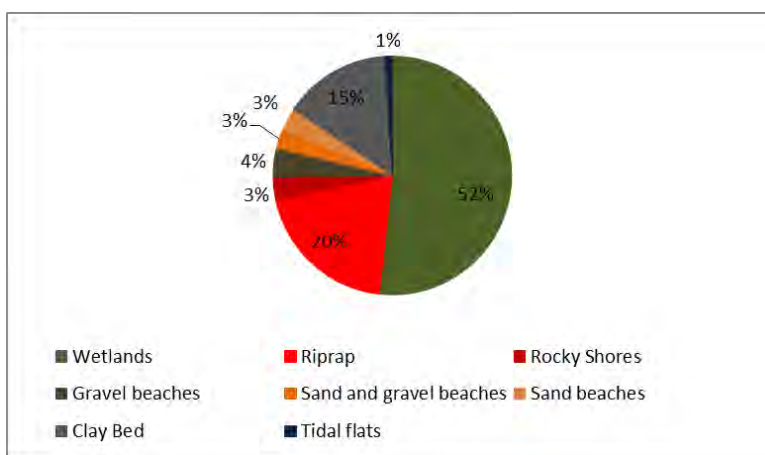

Figure 2: Shoreline classification in Galveston Bay [data from NOAA (2012)].

2.2   Climate Stressors

Positioned along the Gulf Coast, the greater Galveston Bay region is regularly impacted by tropical storms and hurricanes. Since the 1860's, 32 hurricanes or tropical storms have made landfall or passed within a 30 mile radius of Galveston Bay (NOAA 2014a). Each of these storms generates high waves and surge in the bay. The 25 year return surge height (surge that has a 4% chance of being exceeded in any one year) is estimated to be approximately 3 m, and the 100 year return surge height is estimated to be above 5 m (Keim et al. 2007; Needham & Keim 2011; Figure 3). In particular, Hurricane Ike made landfall on the north end of Galveston Island

in 2008 as a Category 2 hurricane and became the most expensive hurricane in Texas history,

causing an estimated 29.5 billion dollars in property damage (Blake et al. 2011).
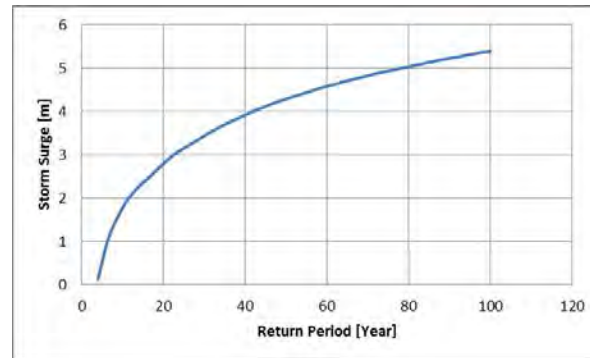

Figure 3: Storm surge return period in Galveston Bay

Increases in the frequency and magnitude of storms, air and sea-surface temperature, sea

water salinity and sea-level are all likely to impact the Bay (Yoskowitz et al. 2009; Elsner et al.

2008; Leatherman 1984; Texas Seagrant 2013) and to intensify the impacts of storms (Warner &

Tissot 2012). In this work, we focus on the impacts of sea-level rise, one component of climatic

change that has already been observed to occur in Galveston Bay and that is likely to have wide-

ranging impacts (Yoskowitz et al. 2009; Hallegatte et al. 2013; Texas Seagrant 2013;

Leatherman 1984). (Given the concerns about vulnerability to coastal hazards and the possibility

that increases in storm intensity and frequency could exacerbate the effects of increases in sea

level, we analyzed 33 years of hindcast wind and wave data just offshore of the Bay (WIS

dataset (USACE 2010) and NOAA buoy 42035 (NDBC 2014). We did not measure an increase

in observed wave height or wind speed in the region over that time period.) Sea-level is

estimated to be rising in the Bay at approximately 6.84 mm/yr (NOAA 2014b). This rise will

cause wetlands to migrate landwards and cause major shifts in the amount and distributions of

natural habitats in and around the Bay. This rate of sea-level rise also includes local subsidence, which is higher than many other areas in the Gulf.

Warren Pinnacle Consulting (2011) simulated the changes in land class types around the Bay under different sea-level rise scenarios using a model named Sea-Level Affecting Marshes Model (SLAMM, WPC 2009; Park et al. 1986). Results for the 1 m sea-level rise by 2100 scenario, -- a reasonable estimate since sea level is expected to increase by approximately 70 cm over 100 years (NOAA 2014b) -- predict a net loss of marine and freshwater wetlands and a net gain of open water areas (Figure 4, Figure 5).
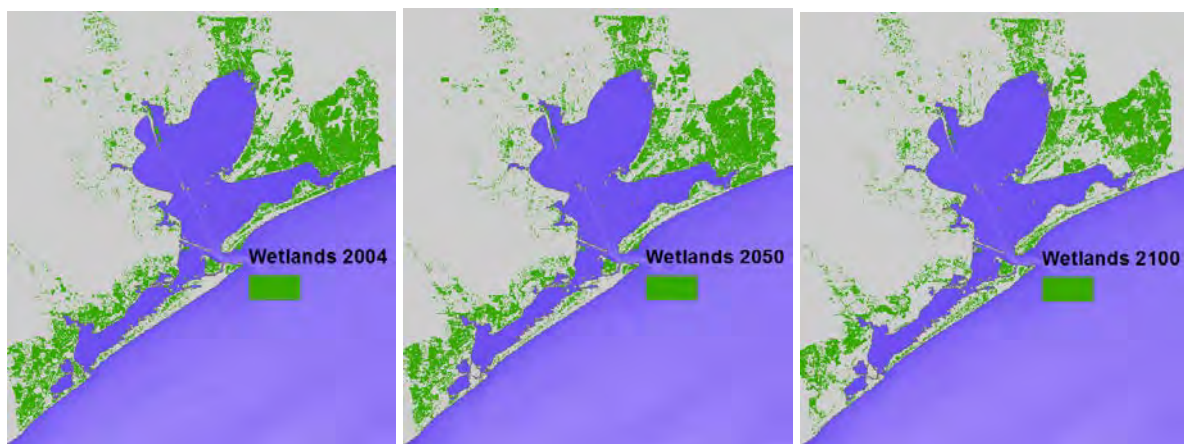

Figure 4: Marine and fresh marsh coverage around the Bay in 2004, 2050 and 2100 under a 1 m SLR scenario.
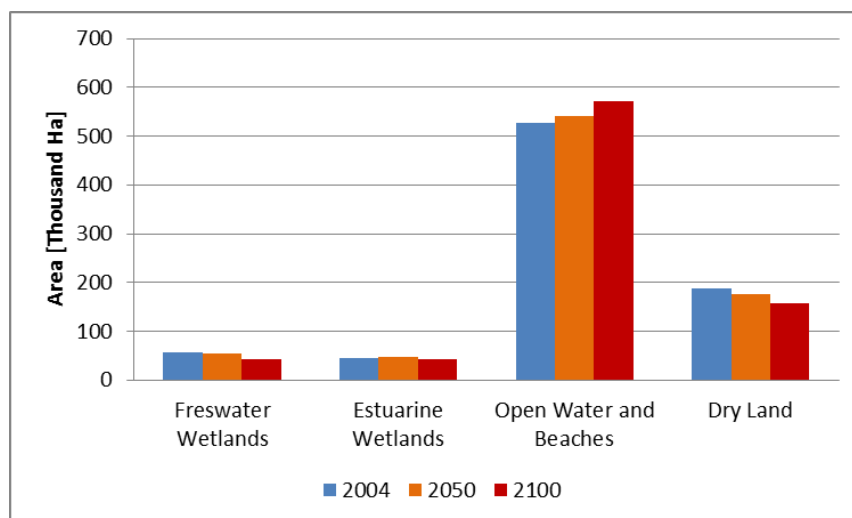
Figure 5: Change in area of different habitat classes in 2050 and 2100, as predicted by SLAMM, for the 1 m SLR scenario.

2.3 Ecosystem Services in the Bay and Impacts of Climate Change

Healthy natural habitats provide valuable and essential services to humans (Millennium Ecosystem Assessment 2005; Tallis et al. 2008). In Galveston Bay, seagrasses, oyster reefs and wetlands protect coastlines from coastal hazards, support fisheries, and provide carbon storage and sequestration services, among other benefits. Table 1 describes some of those services (Yoskowitz et al. 2012; Lester & Gonzalez 2011); italicized text represents the services that are modeled herein. Among these three habitats, coastal wetlands represent the land cover type that will be predominantly affected by sea-level rise stressors since they are located at the water's edge.

Wetlands have the potential to protect people and property against the impact of coastal hazards. They act as a buffer against the impacts of storms and chronic erosion because their very existence can prevent people from building and living directly along the shoreline. They also attenuate storm surge and waves, thus reducing damages during storm events (Anderson et

al. 2011; Jadhav et al. 2013; Wamsley et al. 2010). Finally, they can reduce the size of engineered protective structures, including levees, or dikes (The Bay Institute 2013). Additionally, wetlands are natural stores of carbon, which accumulates in the plant biomass and the soil beneath plants. Lastly, wetlands support the growth of the major commercial and recreational fisheries in the Bay (Bohannon 2013; Orth & van Montfrans 1990; Turner 1977; Zimmerman et al. 2002), which include white and brown shrimp, blue crab, southern flounder and red drum. The delivery of these services can be altered by sea-level rise if marshes are not allowed to naturally migrate landward, which can lead to a reduction of their footprint and the services they provide.

Table 1: Habitats in Galveston Bay and the Some of the Key Services they Provide

Service 　 Habitat	Wetlands	Seagrass meadows	Oyster Reefs
Coastal Protection	*Wave and surge attenuation; storm water retention*; sediments stabilization	Wave attenuation; sediments stabilization	*Wave attenuation*
Fisheries	*Habitats for certain species*	Habitats for certain species	Habitat and source of food for certain species
Blue Carbon	*Storage and sequestration*	Storage and sequestration	Storage and sequestration

3 Ecosystem Service Quantification Methods

3.1 Modeling Scenarios

We quantify the coastal protection, blue carbon and fisheries services delivered by wetlands in Galveston Bay currently, and under a future scenario where sea level increases 1 m by 2100, from its 1992 baseline. This scenario, which also predicts nearly 0.7 m of sea-level rise by 2050 (WPC 2011), captures the current trend of sea-level rise in the Bay (NOAA 2014b).

The primary inputs used in our analysis are land cover class distribution maps around the Bay for 2004 (current), as well as SLAMM modeled projections for 2050 and 2100. The initial land cover maps were generated primarily by converting the National Wetlands Inventory (USFWS 2009) maps to categories used in SLAMM. Future land cover projections come from previous efforts using SLAMM in Galveston Bay (WPC 2011) (Figure 4). SLAMM converts different land cover classes, and wetland types in particular, into other classes based on the rate of sea-level rise, nearshore topography, local land subsidence, shoreline erosion and wetland accretion, among other parameters. In particular, it converts land to new wetland habitat as sea-level rises, unless barriers to wetland migration exist. This assumption in the model can lead to unrealistic results since local communities might not allow natural marsh migration to occur in certain regions that currently do not contain any physical barriers.

In order to reflect the extent of that model's limitations, we also consider, in the quantification of coastal protection and blue carbon services, a scenario where no new wetlands areas are created. We did not include such a scenario in the fisheries model because it didn't yield any significant difference in outputs. To create land use-land cover maps for this scenario, we replaced by estuarine water all regularly and irregularly flooded salt marsh wetlands predicted to migrate in 2050 and 2100 onto regions that do not contain wetlands under the initial

conditions. These maps might over-estimate the amount of estuarine habitats present in the future as local communities might put into place different mitigation measures to prevent land from being inundated.

The potential future land cover maps described above are used to quantify the changes in the delivery of ecosystem services delivered by wetlands as sea level rises. The models and scenario parameters for each service are described in further details in the remainder of this document.

3.2 Coastal Protection

The coastal protection services delivered by wetlands during storms under current and future conditions (Section 3.2.6) are quantified using four different metrics: 1) the storm water retention capacity of the wetlands, 2) the amount of avoided increase in total water level around the Bay due to the presence of wetlands, which is translated into 3) the amount of avoided damages, and 4) the reduction in required levee height needed for protection due to the presence of the marsh, in the West Bay region. We perform the last two analyses in the West Bay region because it is where we had access to parcel data and the U.S. Federal Emergency Management Agency (FEMA) FIRM database (see Section 3.2.4).

The total water level during storms is defined as the sum of storm surge, wave height and wave-induced setup. It is computed by modeling storm surge and wave evolution over a series of transects drawn perpendicular to the coast, starting a few meters offshore of the 2004 shore boundary (Figure 6). While this approach neglects 2-D hydrodynamic processes, it is conceptually equivalent to the process and standards of the FEMA approach (FEMA & FIS 1988).
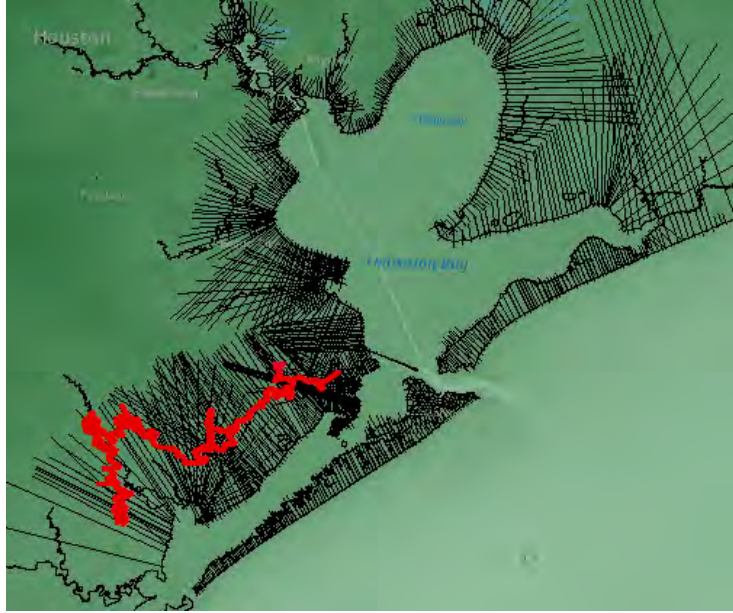
Figure 6: Transect layout around Galveston Bay. Black lines represent modeling transects. Red line indicates the hypothetical levee location.

3.2.1 Storm Water Retention

Wetlands are highly complex three dimensional features that can retain water because of the permeability of their soil as well as their topography. We estimate the storm water retention capacity of wetlands by computing the total volume of water that they can retain if they were to become fully submerged.

3.2.2 Storm Surge

We model storm surge overland in the presence of wetlands by taking a modified bathtub approach, where coastal land is progressively inundated until the local topography exceeds the surge elevation. For each 1-D transect used in the analysis (Figure 6), we project the surge landward from the offshore boundary of that transect. When surge waters encounter wetlands, we reduce the water elevation using a fixed attenuation rate (Table 2). This approach captures the

uncertainty in attenuation rate values associated with storm characteristics and path, topography

and configuration of the wetland captured in Figure 7 (see Appendix C). It also illustrates that

the potential for wetlands to reduce storm surge is inversely proportional to the storm strength

(Appendix C).

Table 2: Default surge attenuation rates by wetlands

Surge at Coast [m]	0.5	1	1.5	2	2.5	3	3.5	4
Attenuation Rate [cm/km]	20	17.4	14.8	12.2	9.6	7	4.5	2
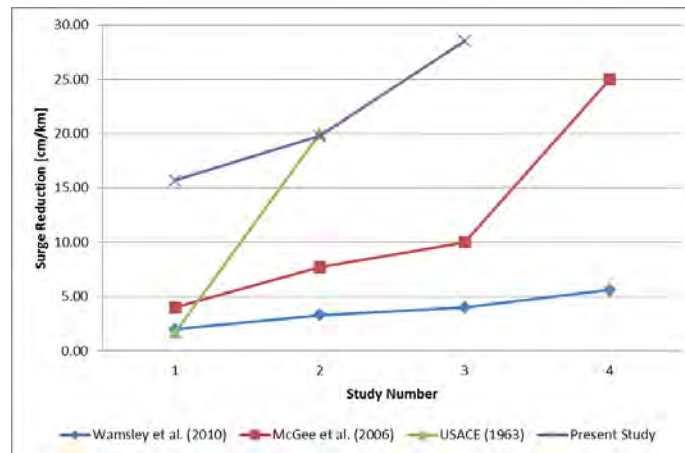
Figure 7: Rates of surge reduction as measured by different studies. In addition to rates measured by other researchers, we also quantified the potential for marshes in the Bay to reduce the surge generated by 3 idealized storms in West Bay (see Appendix C).

3.2.3 Nearshore Waves

In addition to reducing storm surge, wetlands can also reduce wave height and wave-

induced mean water level during storms. From values of significant wave height and peak period

at the seaward boundary of a particular transect, the model computes wave height and wave setup at each point along that transect following Equation (1):

$$\frac{\partial E_w C_g}{\partial x} = S_{Wind} - D_b - D_f - D_v \tag{1}$$

where E_w is the wave energy density, C_g is the wave group velocity. In Equation (1), waves are re-generated by winds via the source term S_{Wind} (FEMA & FIS, 1988), which is a function of the sustained wind speed of the storm, wave height and the local water depth h. The water depth is equal to the sum of the still-water depth – zero for locations on land – and the surge height.

Waves also dissipate their energy via breaking, D_b in Equation (1) and bottom friction, D_f. Those terms are expressed following the standard formulation of Thornton & Guza (1983). The expression of D_f is a function of the bed friction coefficient C_f, which is fixed at $C_f = 0.01$ for sandy beds, among other terms. C_f for agricultural and bare lands is computed from their corresponding Manning coefficients M_n, following the classic conversion expression (Augustin et al. 2009):

$$C_f = \frac{8g}{h^{1/3}} M_n \tag{2}$$

Finally, wave energy is dissipated because of the presence of vegetation via the term D_v,

following the expression in Mendez & Losada (2004). It is a function of wetlands density, N_v,

stem diameter, d_v, and a drag coefficient, C_d, which is approximately equal to 0.1 (FEMA

2007b).

In the absence of detailed local field measurements, we created a database of average

plants' physical characteristics in Galveston Bay (

Table 3; see also Appendix A). Based on observations by Feagin et al. (2011), this

database was modified to create "healthy" and "degraded" physical parameter values for each

habitat. For the healthy habitat conditions, average plant densities presented in

Table 3 were increased by 50%, and average plant diameter and height by 30%. For degraded habitat conditions, average plant densities were reduced by 50%, and average plant diameter and height by 30%. These parameter values are not intended to accurately reflect the status of, and expected changes to, each habitat. Rather they are used to evaluate the model's sensitivity to changes in these input parameters. Outputs provide an uncertainty estimate of their protective capacity given changes in their status. (Guannel et al. (2014) performed an uncertainty analysis of Equation (1) and showed the importance of the choice of drag coefficient in estimating the role of habitats in reducing the impacts of wave height.)

Table 3: Physical Characteristics of Vegetation Associated With Different Land Classes

Land Cover Class	Height [m]	Diameter [m]	Density [units/m^2]	Source
Swamps and Forested Wetlands	15	1.5	0.0172	Arborday.org
Scrub/Shrub	2.5	0.1016	0.0861	University of Florida, Alabama Forestry Department, and Davesgarden.com
Forests	10	0.2	0.0861	FEMA
Low Salt Marsh	0.8	6.4x10^{-3}	172	FEMA guidance for the Chenier Plain Region
High Salt Marsh/Brackish Marsh	0.57	5.8x10^{-4}	3,585	FEMA guidance for the Chenier Plain Region
Fresh Marsh	1.22	8.14x10^{-3}	107	FEMA guidance for the Chenier Plain Region

Equation (1) applies to vegetated and non-vegetated bare beds. When waves encounter structures or obstacles such as roads or buildings, isolated or in a residential community, we compute the wave height, H_t, landward of those wave-blocks following (FEMA & FIS, 1988):

$$H_t = \max \begin{cases} \gamma h_t \\ K_t H_i \end{cases} \tag{3}$$

where γ is a breaking coefficient used in the breaking dissipation expression D_b and h_t is the water depth landward of the blocks. The variable H_i is the wave height seaward of a structure block, which has a transmission coefficient K_t. We compute K_t as (FEMA & FIS 1988):

$$K_t = L_S^{NR/2} \tag{4}$$

where L_S is the length of a structure block and NR is the number of structures in that block.

From computed profiles of wave height, we estimate wave-induced mean water level following (Guannel et al. 2014):

$$\rho g\left(h+\bar{\eta}\right)\frac{\partial \bar{\eta}}{\partial x}+\frac{\partial S_{xx}}{\partial x}-\frac{3}{16\sqrt{\pi}}\rho C_d d_v N_v \frac{gk}{\tanh kh}H^3=0 \tag{5}$$

where $\bar{\eta}$ is the mean water level, h is the water depth, S_{xx} is the wave-induced radiation stress, H the wave height, and k the wave number.

3.2.4 Valuation

The value of wetlands is defined herein as the dollar amount of avoided damages to buildings (residential, commercial and industrial) and the number of people protected by those habitats during a storm. The amount of avoided damages is estimated by taking the difference between modeled property damages caused by storms in the presence and absence of wetlands (where in the case of "absence" wetlands are replaced by bare land). The number of people protected by wetlands is computed by taking the difference between the number of people affected by the storm in the presence and absence of wetlands. Because of data limitations, estimates of damages and number of people affected by storms are computed in the West Bay region only (Figure 9).

The total dollar amount of damages D caused by a storm is estimated as

$$D = (s + z)V \qquad\qquad (6)$$

where V is the value of a building (commercial or residential), and s and z are the structural and content damage ratios ($0 \leq \{s, z\} \leq 1$). We obtained a spatial inventory of the properties in that region from the Galveston County appraisal district (GCAD 2014). Damage ratios values are then computed as a function of the flood depth in a building following depth-damage relationships used in FEMA's HAZUS model (Figure 9) (Scawthorn et al. 2006; FEMA 2011; Cannon et al. 1995; USACE 2003). Flood depth is defined as the total water level (Sections 3.2.2 and 3.2.3) in a building, relative to the first floor elevation of that building.

Because of a lack of detailed information about the location and physical characteristics of buildings in West-Bay, various simplifications were made. First, we approximated the location of structures at the centroid of the parcel where they were listed. Also, instead of relying on a replacement cost approach used by HAZUS, we used parcel tax assessment data as a proxy for a structure's value (Cannon et al. 1995). Next, since we did not have parcel-level information on the number of floors for residential buildings, we created a composite damage function for single story and multi-story buildings without a basement, based on the damage function specific to those building types published in USACE (2003). We based the weighting of this function on observed empirical estimates of building heights in the Southern United States (FEMA 2011), where 72% of residences in Texas are single-story, and 28% are 2 stories or higher. Similarly, we made the content damage function into a composite damage function based on the same process as structural damages. For commercial and industrial buildings, we used the depth-damage functions published by FEMA (2011). Finally, we estimated first floor elevation values for the various buildings in the region by creating composite substructure height for the different

Flood Insurance Rate Map (FIRM) zones in West Bay. We identified the FIRM zone of each building in the region by overlaying FEMA's preliminary Digital FIRM database information (FEMA 2012; Figure 9) over the parcel data in the region (GCAD 2014). Then, we computed the first floor elevation for buildings in each zone using information about the distribution of different types of substructures in the area as well as pre/post FIRM heights from (FEMA 2011; Chap. 3). We obtained a height of 1.52 m for A zone structures, 1.76 m for V zone structures, and 1.17 m for all other structures.

In addition to damages, we also compute the size of the affected population by a storm by multiplying the average population per residence by the number of damaged residential properties. We estimated an average population of 2.34 people per residence in West Bay from census data (USCB 2013).
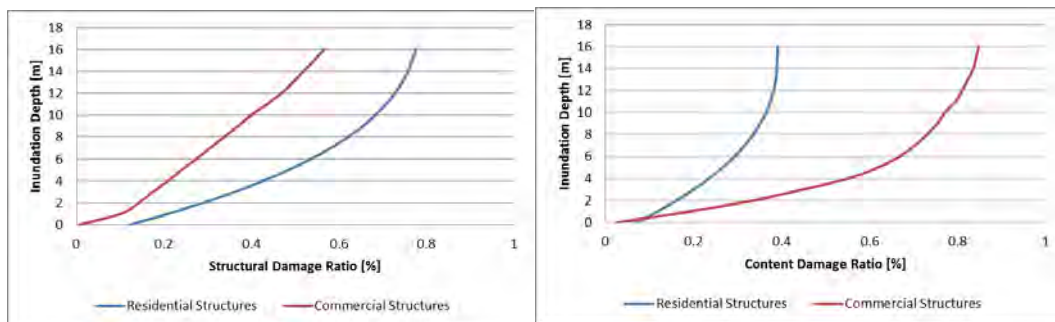

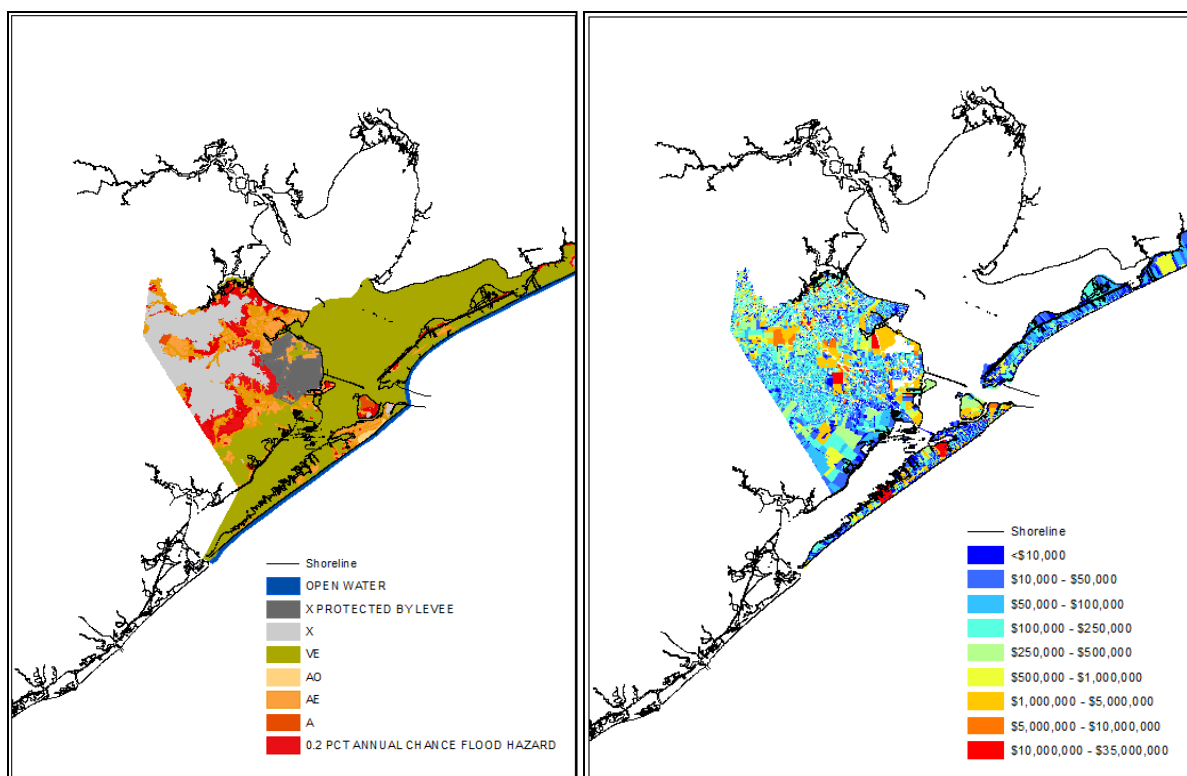Figure 8: Damage ratio curves use in the analysis.

Figure 9: Preliminary FEMA FIRM database (left) and parcel data value (right) used for the valuation of marshes in West Bay.

The approach adopted herein is likely to underestimate total property damages since it does not capture losses to market driven commodities (e.g., agriculture or land on which infrastructure is built), and property values may not have been recently calibrated to the local market (Cannon et al. 1995). Finally, because the approach does not address people's willingness to pay for coastal storm protection, the avoided damages estimates obtained should not be interpreted as the economic value of the wetlands. Instead, they represent one component of the value of the wetlands for storm protection.

3.2.5   Structures

Although wetland habitats have the capability to reduce total storm water levels, it is sometimes preferable to build levees to prevent inundation in a region of interest. However, wetlands can still help reduce the size, and thus the construction cost, of such structures (The Bay Institute 2013). We measure the effectiveness of wetlands at reducing the size of levees by computing the height of a hypothetical levee, in the presence and absence of wetlands, located at the inland limit of the marsh footprint as computed by the SLAMM model under the 1m SLR scenario; such a levee would allow for marshes to migrate to the toe of the structure. Where this inland limit intersected roads or developed areas, we placed this hypothetical levee on their seaward edge (Figure 6).

For each transect, we compute the height of the hypothetical levee with 3H:1V side slopes following the method presented in the Overtop Manual (Pullen et al. 2007, chap.6), which relates levee height to levee slope, wave height and surge depth at the levee's toe and the overtopping rate q over the structure. Assuming that the levee would be stable and only suffer minimum damage during a particular storm, we fixed the overtopping rate at $q = 1$ l/s per m during a storm (USACE 2002, chap.VI–5).

3.2.6   Forcing Conditions

To model the protection services of wetlands, we generated a series of potential hurricane forcing conditions. Based on an analysis of storm surge return period values in and around Galveston Bay (Figure 3), we created a range of realistic values of open coast surge elevations and corresponding sustained wind speed values (Table 4), using the Saffir-Simpson scale (NOAA 2013). We defined the deep water wave characteristics created by each of these

synthetic hurricanes based on the range of wave height and period that those storms could

potentially generate (USACE 2002).

Table 4: Surge and offshore wave characteristics used to quantify the protective services of wetlands in Galveston Bay

Storm Name	C1	C2	C3	C4	C5
Surge at Coast [m]	2.0	2.5	3.0	3.5	4.0
Sustained Wind Speed [m/s]	35.0	40.0	45.3	50.3	55.6
Offshore Wave Height [m]	7	8	8	10	11
Offshore Wave Peak Period [s]	10	11	11	12	13

We compute the total water level during each storm by linearly adding storm surge, wave

height and wave setup along each of the transects. If different transects cross paths, the total

water level at the crossing points is equal to the maximum water level of the two. Total water

level over the entire surface is then computed by linearly interpolating and smoothing maximum

total water level over the area.

As mentioned earlier, we compute the total water level generated by hurricanes by

computing the evolution of surge and waves over a series of transects drawn perpendicular to the

Bay's shoreline, with an offshore boundary located a few meters inside the shallow Bay. To

obtain values of storm surge at that boundary, we assume that the storm surge in the Bay is equal

to the surge at the coast (Table 4). We also assign wave characteristics at that boundary using

outputs of the wave propagation model, Simulating Waves Nearshore (SWAN) (Booij et al.

1999).

We modeled the evolution of deep water hurricane waves inside the Bay using the

SWAN model on a 50 m resolution mesh, with an offshore boundary located approximately 5

km offshore and lateral boundaries located approximately 10 km away from the Bay's

boundaries. For each storm in Table 4, we first deepen the whole computational domain

uniformly by the input storm surge value. Then we compute the wave characteristics in the Bay

using the offshore wave characteristics associated with each storm, assuming that the associated

wind speed blows constantly from any of the 16 cardinal and half cardinal compass directions

(Stockdon et al. 2012). Thus, for each storm, we run the SWAN model 16 times. We compute

wave forcing conditions along the Bay's shoreline by taking the maximum wave height and

associated wave period generated by the 16 runs. Example outputs of the SWAN model for the

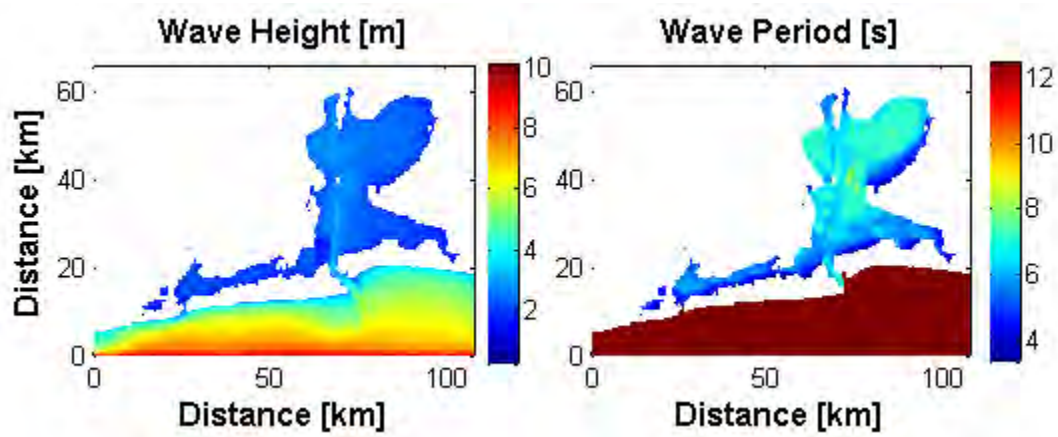storm C4 with southerly winds is presented in Figure 10.

Figure 10: Example SWAN outputs for Storm C4, with constant winds blowing from south to north.

For a particular storm, we quantify the protection services delivered by wetlands by

computing the total water level over inundated areas when wetlands are present and when they

are replaced by bare land. Under current conditions, we compute total water level for each of the

forcing conditions presented in Table 4, assuming that the habitat is healthy or degraded (see

Section 3.2.3). For storm C4, we also compute total water level assuming average health

conditions using the physical parameters presented in

Table 3. This is then used to compute total water level for each time step in the 1 m SLR scenario considered herein.

3.3 Blue Carbon

3.3.1 Carbon Storage and Sequestration

Wetlands are some of the most productive carbon storage and sequestering habitat types in the world. Carbon in wetland vegetation can accumulate in four "pools": (1) the above ground biomass, (2) the below ground biomass, (3) the soil, and (4) the surface detrital layer. Carbon accumulates when wetlands persist over time and 'sequester' it, or when non-wetlands land cover classes convert to wetlands. On the other hand, carbon decays and is lost, or re-emitted, when a wetland coverts to a non- wetland land cover class such as estuarine water, agricultural land or a different type of vegetation. Hence, the amount of carbon present in wetlands at a given time is a function of the amount and type of habitats currently present, as well as the type of habitats that were present in the past.

We estimate changes in the amount of carbon in marshes, or blue carbon, in the first three pools (changes in carbon content of the surface detrital layer are not captured) as sea level rises using the InVEST Terrestrial and Blue Carbon models (Sharp et al. 2014), along with land cover class information from SLAMM outputs. We provide a brief description of those models and their inputs in Appendix D. Elemental carbon values in all carbon pools and accumulation and decay rates were assigned based on values used in previous blue carbon modeling efforts in Freeport, Texas (Walsh et al. 2013), and values found in the literature (e.g., Chmura et al. (2003), Loomis & Craft (2010), Choi et al. (2001), or Yu et al. (2012), see also Appendix D).

Since the blue carbon model is currently designed for salt marshes, we only model carbon accumulation and decay over time for the following three marine wetland categories: regularly

flooded, irregularly flooded and transitional salt marshes. Accumulation and decay that also occur in the fresh marsh and swamp land cover classes are not incorporated into the analysis, since the model only accounts for changes in salt marshes coverage. Furthermore, we assume that accumulation and decay only occur in the soil pool, where the vast majority of carbon is stored and sequestered in wetland habitats. This approach is reasonable because the amount of carbon stored in the plant biomass remains relatively constant over time and does not accumulate or get lost like carbon in the soil pool.

In order to take into account the wide variability in model inputs, we evaluate the relative importance of those inputs on the amount of carbon stored and sequestered by wetlands via an uncertainty analysis called tornado analysis (Howard 1988; Celona & McNamee 2001). To conduct the uncertainty analysis, we first define a range of possible values (minimum, typical value, maximum) for soil carbon content, along with carbon accumulation and decay rates. We obtained those ranges by taking the maximum and minimum values reported in the literature that are applicable to Gulf wetlands (see Appendix D). Next, we run the terrestrial and blue carbon models for each of the three possible values of a single variable (e.g., soil carbon content), while keeping all the other variables (e.g., accumulation and decay rates) constant at their typical values. This step produces three model outputs for each variable. The variable for which outputs vary the most is the one that drives most of the model uncertainty.

3.3.2 Valuation

In addition to computing the amount of carbon stored and sequestered by wetlands, we also compute the economic value of sequestration (not storage) as a function of the amount of carbon sequestered, the monetary value of each unit of carbon, and the change in the value of carbon sequestration over time. Because local carbon emissions affect the atmosphere at a global

scale, we compute the economic value of sequestration using the Social Cost of Carbon (SCC) following USIWGSCC (2010). This approach estimates the additional costs induced by climate change caused by increased emissions of CO_2, via impact on agricultural productivity, human health, and storm frequency (U.S. EPA 2013). Since the damages incurred from carbon emissions occur beyond the date of their initial release into the atmosphere, the damages from emissions in any one period are the sum of future damages, discounted back to that point. Thus, to calculate the SCC for emissions in 2050 or 2100, we computed the present value of the sum of future damages (until 2050 or 2100), using a 3% discount rate (and appropriate SCC schedule in USIWGSCC (2010) guidance).

3.4 Fisheries

Patterns of use of marshes change among fish species seasonally and with age classes (Stunz et al. 2010). Fisheries in Galveston Bay strongly prefer saltmarshes, and especially marsh edges (~1m shoreward from open water; (Minello & Rozas 2002; Stunz et al. 2010), rather than shallow open water habitats, in their juvenile stage (Minello et al. 2008; Thomas et al. 1990).

We estimate the impacts of sea-level rise and marsh migration on fisheries by using two different types of models. First we estimate changes in available marsh edge habitats for key fisheries using the InVEST Fisheries Habitat model. Next, we estimate changes in landing of shrimp for different sea-level rise scenarios using the InVEST Shrimp model. Note that neither of these models has been released as part of the InVEST package.

3.4.1 Fishery Habitat Change

We assess changes in the amount of marsh edge habitats of various fisheries as a function of changes in the location and quantity of land and marine habitats for four different

commercially and recreationally important fish species in Galveston Bay: (1) Blue Crab (*Callinectes sapidus*), (2) Brown Shrimp (*Farfantepenaeus aztecus*), (3) Southern Flounder (*Paralichthys lethostigma*), and (4) Red Drum (*Sciaenops ocellatus*). Since these species are present in the Bay's wetlands primarily during their larval and juvenile stages, we only consider these life stages in our analysis (Appendix E).

Near wetlands areas, the most important and common habitat and environmental parameters that define where these larval and juvenile fish can live are sediment type, water depth, temperature, salinity and turbidity. Accordingly, we collected information about the environmental and habitat preferences for each species of interest (Appendix E). However, because of the low spatial resolution of publicly available data, and because we focus on marsh edge regions, we only consider water depth in this analysis.

From bathymetric and topographic data, and sediment and living habitat location information, the model first extracts the maximum livable area for a given species' life-stage based on depth preferences. For marsh edge areas, those are waters between -2 meters and Mean High Water (MHW) for all species. Then the model incorporates information about location and quantity of available living habitats for those species by generating suitable habitat maps based on the species preferences (Appendix E). We obtained changes in the areal extent of estuarine and wetlands areas from SLAMM outputs (WPC 2011). Lastly, using information from the different bathymetric and environmental layers, the model computes the amount of suitable habitat for that species. We conducted this process for the different time horizons considered, based on projected land cover maps. Results from the model provide a first estimate of the changes in habitat for each key species under different sea-level rise scenarios.

3.4.2 Shrimp Fishery Model

Shrimps life cycle is modeled using a stage-based single species population model (Figure 11) using a Leslie matrix approach (Jensen 1974). Starting with recruitment, the model captures the survival rate of egg/larvae to juvenile, then from juvenile to adult. These survival rates are a function of salinity and temperature. The survival rate from juvenile to adult is also a function of the area of marsh habitat present. The output of the model is the weight of shrimp harvested commercially or recreationally. Harvest and natural mortality dictate recruitment. The total area of marsh available in the Bay also influences the survival rates of shrimps at all stages.
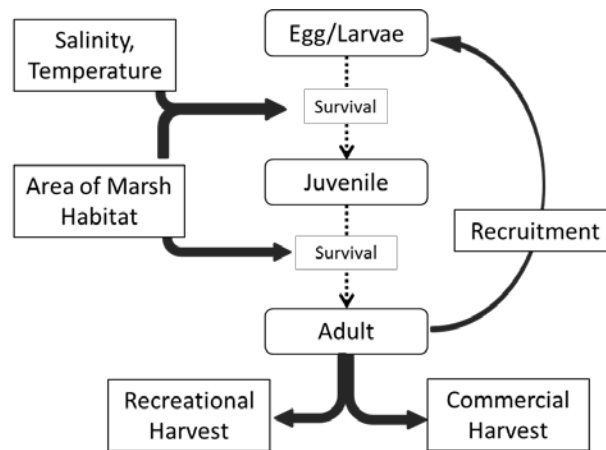
Figure 11: Shrimp growth model.

To evaluate changes in habitat we calculated the percent change in salt marsh habitat using the SLAMM maps for each sea-level rise scenario. We only considered changes in the SLAMM category 'regularly flooded marsh,' as this is the low elevation saltmarsh used primarily by juvenile shrimp.

4   Quantification of Ecosystem Services in the Bay

Coastal wetlands are valued by their capacity to deliver three different services: 1) coastal protection, 2) blue carbon storage and sequestration, and 3) habitat for fisheries and food from shrimp fishing activities. The delivery of these three services is quantified under current and future conditions, as sea-level rises and marshes are (not) allowed to migrate landward.

4.1   Coastal Protection

In this section, the coastal protection services delivered by wetlands are quantified in three different ways. First, we explore the value of wetlands to serve as a physical buffer between people and storm waters. Next, we investigate the dollar value of wetlands due to their ability to reduce water levels during storms and thereby avoid damages to structures and reduce the number of people displaced. Finally, we discuss how the wetlands can reduce the height of levees, if such structures were built to prevent any damages to existing structures.

4.1.1   Wetlands as Storm Buffer and Storm Water Retention Capacity

Existing shoreline structures protect people living at the water's edge. For people living landward of wetlands in Galveston Bay, one primary coastal protection value of this habitat is its ability to serve as a physical buffer between them and storm waters (Costanza et al. 2008). First, wetlands around Galveston Bay (East and West Bay, Trinity Bay) extend, on average, 6 km inland from the shore, but can be as wide as 16 km. Additionally, with an average elevation of approximately 1.3 m at its landward edge, wetlands protect people and property against the impacts of tropical or subtropical storms (NOAA 2014a), which are the majority of storm types that move through the region. Those storms typically generate surges lower than 1 m (USACE 2002), which is lower than the 7 year return surge (surge that has a 14% chance of being exceeded in any one year) height of 1.1 m (Figure 3).

In addition to serving as a buffer, wetlands also deliver valuable coastal protection services by serving as natural storm water retention systems. Currently, wetlands around Galveston Bay can hold more than 300 million m^3 of water during a storm that generates a 1 m surge (Figure 12). As sea level rises and wetlands' footprint migrates landward, their retention volume also increases. However, if they are not allowed to migrate, their retention volume decreases (Figure 12). By 2100, for example, almost 200 million m^3 of water would have to be treated or would runoff if marshes are not allowed to migrate, because of the ensuing decrease in wetlands area.
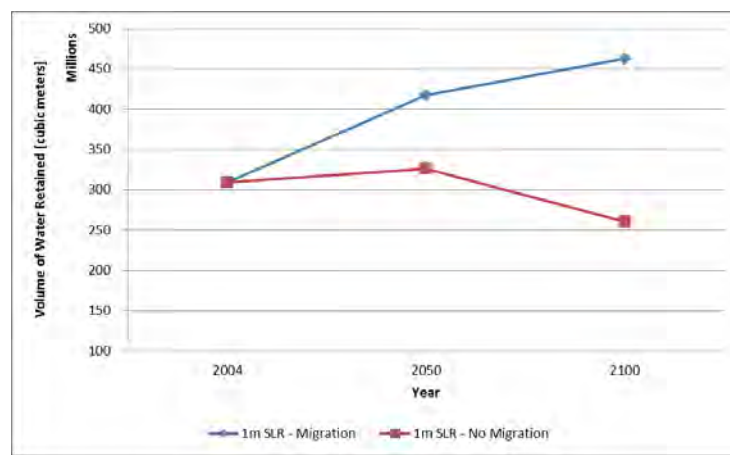

Figure 12: Storm water retention volume by wetlands in Galveston Bay for a 1 m SLR scenario.

4.1.2 Storm Impact Reduction and Avoided Damages due to Wetlands

Another important coastal protection service delivered by wetlands is their capacity to limit the amount of damages to properties and the number of people displaced by large storms. This service is quantified by the wetlands' ability to reduce storm-induced total water level, which is obtained here by modeling storm surge elevation, wave height and wave-induced mean water level along a series of transects drawn perpendicularly to the shoreline, along the Bay

(FEMA & FIS 1988; FEMA 2007a). In this section, we first discuss the ability of wetlands to reduce total water level generated by storm C4 (Table 4) in and around the Bay, under current and future sea-level conditions. Next, we compute the dollar value of avoided damages and the number of people protected by wetlands for all storms considered herein.

4.1.2.1   Current Sea-Level Conditions

Under current sea-level conditions, the coastal protection value of wetlands is computed from total water level values obtained in the presence and absence of healthy and degraded marsh plants (Section 3.1). Water level input conditions along the Bay's shoreline were obtained from fixed storm surge values corresponding to the storms presented in Table 4 and from SWAN outputs (Figure 10).

Comparison of model outputs obtained in the presence and absence of wetlands for Storm C4 shows that marshes have the ability to reduce total water levels during storms in Galveston Bay (Figure 13 and Figure 14). However, as indicated in the discussion of storm surge attenuation (Section 3.2.2 and Appendix C), the protection services delivered by wetlands is a function of the amount of habitat present, the "health" (or physical characteristics) of the plants, and topography. Indeed, we find that healthy marshes can reduce total water level by more than 0.5 m (50%), whereas degraded habitats only reduce wave height by a few centimeters. Additionally, even though healthy habitats are supposedly more efficient at reducing wave energy than degraded habitats, their relative effectiveness is highly spatially variable (Figure 13). Wetlands seem to attenuate more wave energy in the Trinity Bay region, for example, than in West or East Bay. Also, within the West Bay region, wetlands near Chocolate Bay moderate total water levels more than wetlands near Texas City. (Note that the isolated wave height smears in the different maps are the results of the interpolation between different 1-D transects with

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different lengths, which is a function of local topography.) These differences are due to small differences in input wave height conditions at the shoreline (Figure 10), the size of the marsh habitat in those regions, but more importantly to local differences in topography.
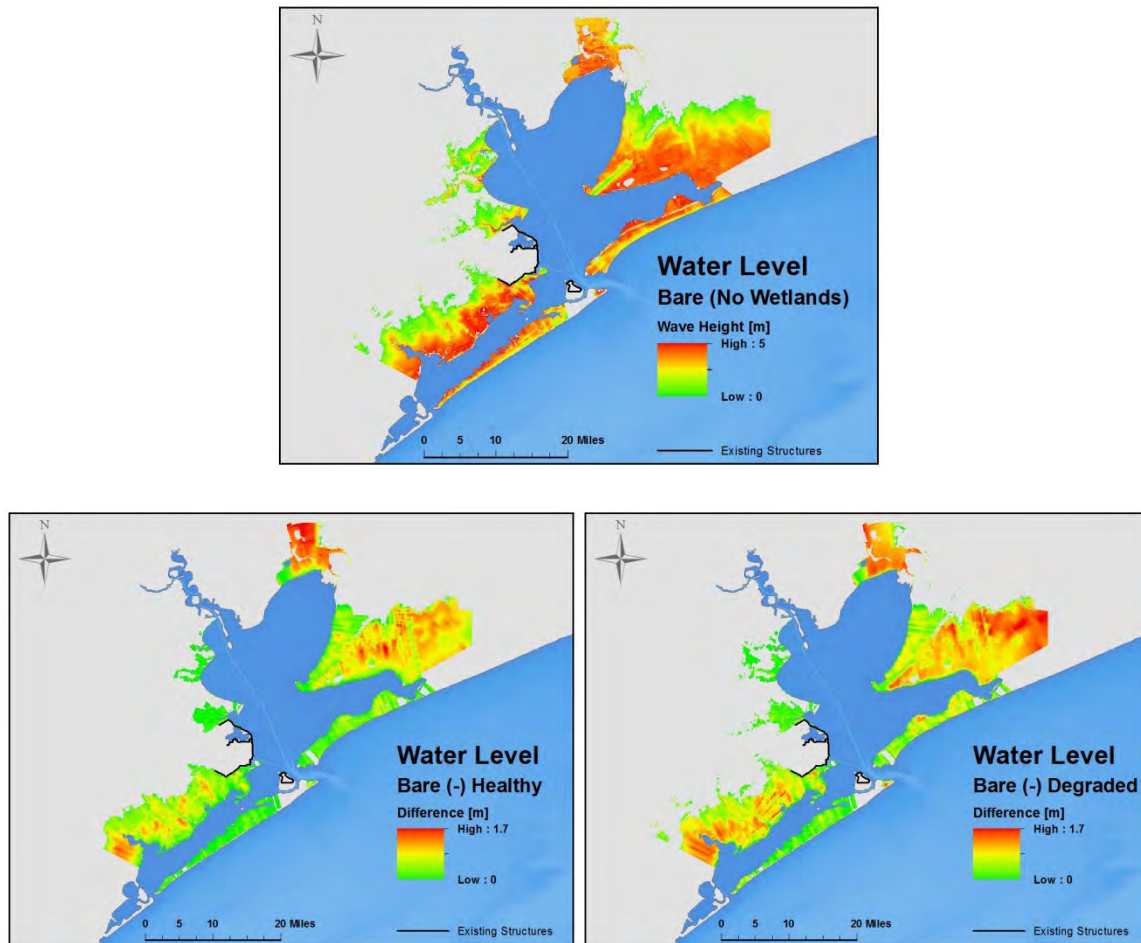

Figure 13: Investigation of role of wetlands at reducing total water levels during storms. Top: Total water level during storm C4, in the absence of wetlands. Bottom Left: Difference in total water level during storm C4 between "No Wetland" and "Healthy Wetland". Bottom Right: Difference in total water level during storm C4 between "No Wetland" and "Degraded Wetland".
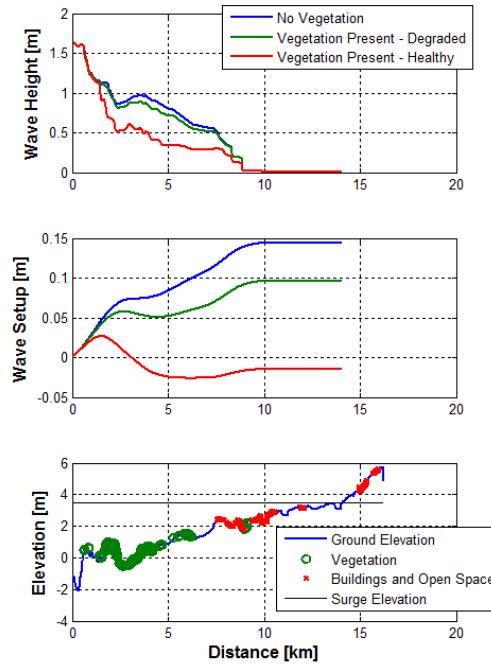
Figure 14: Example Nearshore Wave model output for a transect in Galveston Bay, in the absence and presence of vegetation.

We repeated this exercise for the storms described in Table 4. Although the difference between outputs of the different habitat conditions varied slightly, the overall patterns and conclusions are similar to those presented above. The only difference between the outputs for different storms is that total water level increases with the strength of the forcing.

4.1.2.2   Sea Level Rise Scenario

To quantify the value of wetlands in Galveston Bay under a sea-level rise scenario, we computed total water level in the Bay generated by Storm C4. It appears that as sea level rises, storm waters reach further inland, although the overall pattern of total water distribution is similar from one time period to another (Figure 15). Comparison of total water level values obtained in the presence and absence of wetlands (Figure 16) shows high heterogeneity in the potential of wetlands to reduce the impact of storm, as was observed in the previous section. Not

allowing wetlands to migrate also results in higher water levels, as exemplified by results in
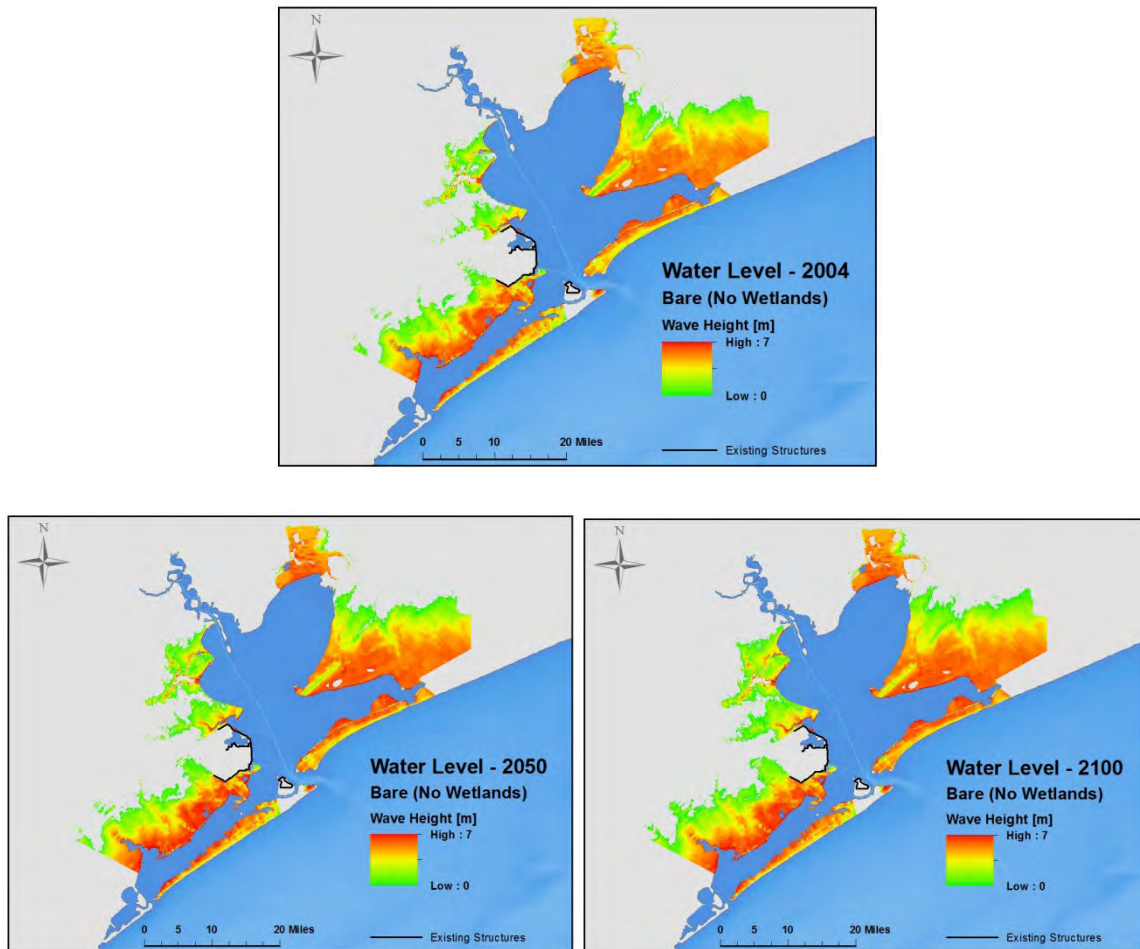
Figure 16.

Figure 15: Total water level surface in Galveston Bay currently (2004, top), in 2050 (bottom left) and 2100 (bottom right), using forcing conditions corresponding to storm C4.
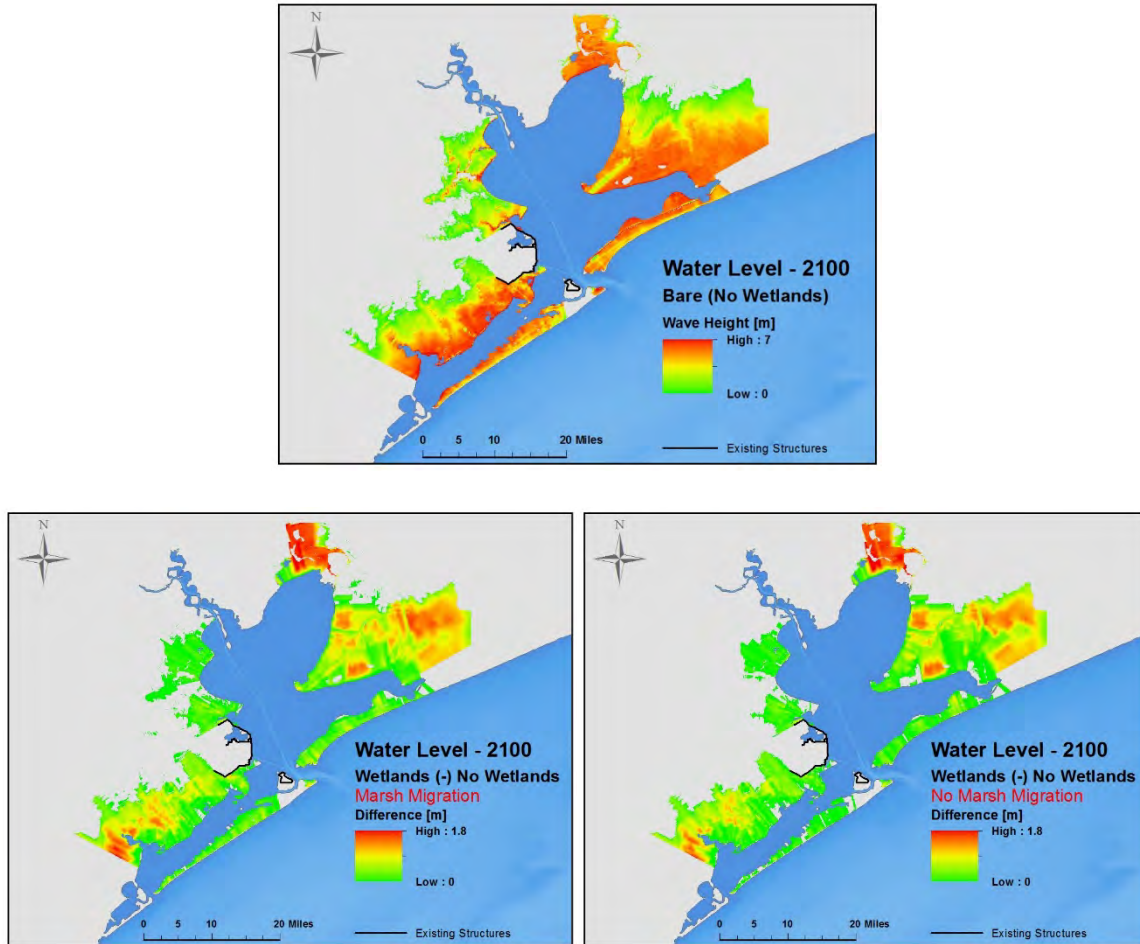
Figure 16: Estimates of the wetlands role in total water level attenuation, for the year 2100. Top: Total water level. Bottom Left: Difference in total water level computed in the absence and presence of wetlands, assuming that wetlands can migrate. Bottom Right: Difference between total water levels computed in the absence and presence of wetlands, assuming that wetlands do not migrate.

4.1.2.3 Valuation

For current and future wetlands coverage conditions, the protective service of wetlands are given in dollars of avoided damages and change in the number of affected people. Results indicate that, for current conditions, the total amount of damages and number of people affected during storms increases with the storm intensity (Figure 17). As expected from the biophysical results presented above, wetlands reduce the amount of damages and protect communities from

storm impacts. On average, healthy wetlands reduce damages by $15 million and protect 450 people whereas degraded wetlands reduce damages by $3 million and protect 105 people. For storm C4 in particular, healthy wetlands reduce damages by $17 million and protect 348 people. Degraded wetlands reduce damages by $4 million and protect 63 people. If we use average physical parameters values under this storm scenario (

Table 3), the wetlands reduce damages by $8 million and protect 167 people, for storm C4. Note that, although the amount of avoided damages increases with storm intensity, the number of people protected by wetlands does not necessarily increase with storm intensity since we assumed a fixed number of people per household, regardless of property value.
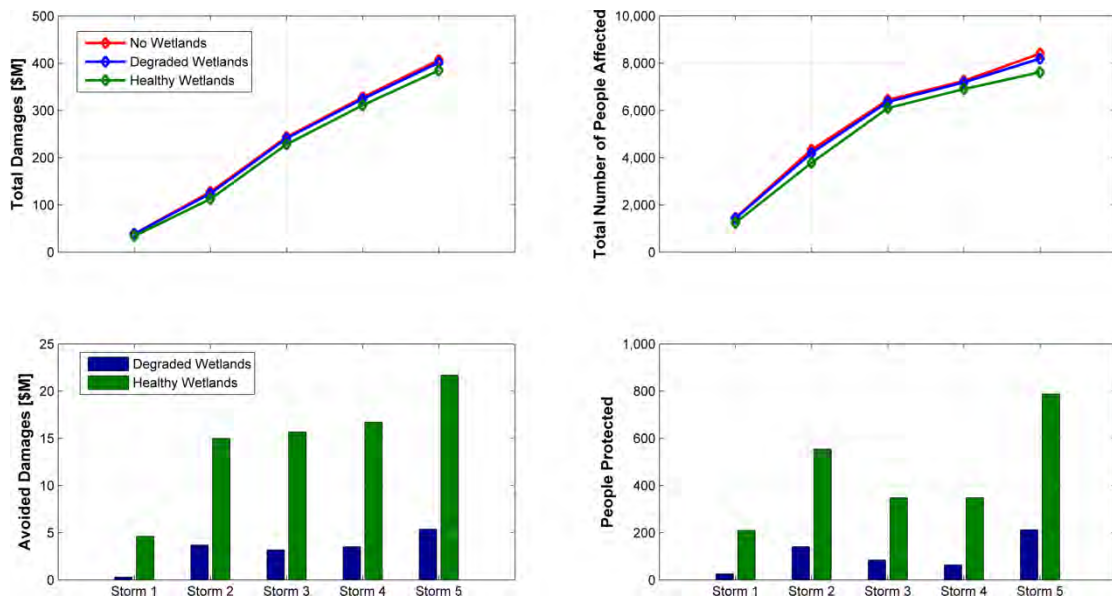

Figure 17: Top: Total amount of damages (left) and total number of people affected (right) by storms in Galveston Bay for different storms under current conditions and for different habitat conditions. Bottom: Amount of avoided damages (left) and people protected (right) by wetlands.

As sea level rises, the total amount of damages increases since storm waters extend further inland (Figure 18). However, wetlands reduce the total amount of damages by an average of $12 million, and protect an average of 395 people if they are allowed to migrate. If marshes are not allowed to migrate, the amount of avoided damages decreases to $9 million and the number of people protected reduces to 310 people. Interestingly, wetlands are able to prevent more damages as sea level rises, if they are allowed to migrate, since there is slightly more wetland area in 2050

than in 2100. However, if wetlands are not allowed to migrate, their total coverage declines from 2050 to 2100, yielding a lower amount of avoided damages for storms occurring in 2100 compared to 2050.
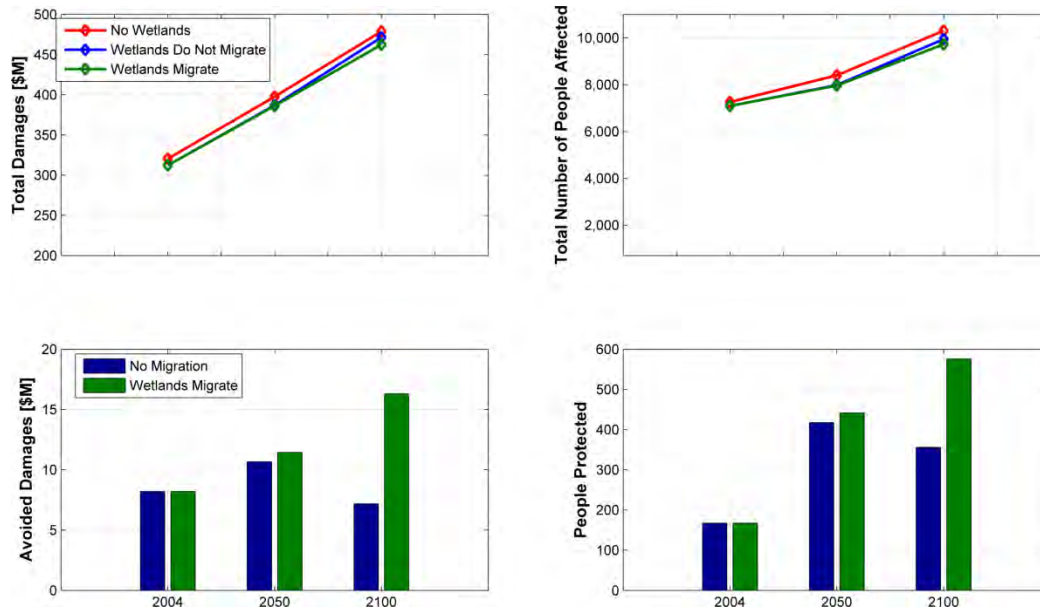

Figure 18: Same as Figure 17, for a 1 m SLR scenario.

4.1.3 Structures

Wetlands can reduce total water levels during storms, but they do not prevent damages from occurring, as shown in the above section. Man-made protective structures can protect people and properties against the impact of storms. But structures are also costly, especially if they are designed to prevent damages during the strongest storms (USACE 2002).

We investigate the potential for wetlands to reduce the height of levees in West Bay by assuming that a levee is built at the landward edge of the 2100 marsh footprint. Such a levee is

designed to reduce overtopping and inland inundation to a minimum. We find that, in the

absence of marsh, levee height requirements are quite spatially variable (Figure 19), as wave

height and total water levels varied in this region of the Bay (see Section 4.1.2.2). The presence

of wetlands reduces the height requirements of levees by as much as 4 to 5 m, but, again, the

potential for wetlands to reduce the height of the structure varies along West Bay. We conducted

this analysis for all the scenarios and found similar trends. In addition, we also find that levee

heights increase as sea level rises, and that wetlands are more efficient at reducing levee height if

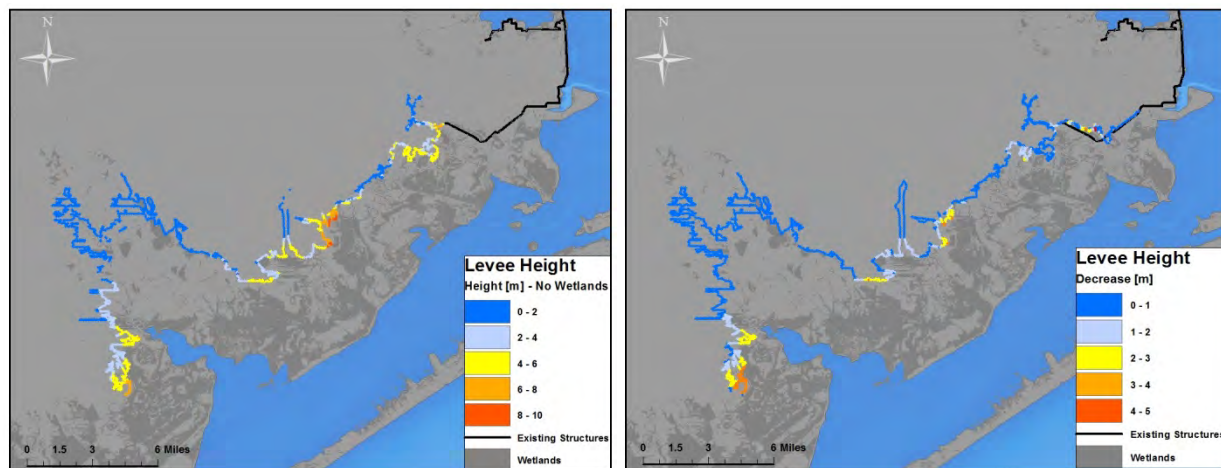they are allowed to migrate inland (Figure 20).

Figure 19: Effect of wetlands on levee height in West Bay, for storm C4. Left: Levee height required to allow 1 l/s per m of overtopping, in the absence of vegetation. Right: Levee height decrease due to the presence of vegetation.
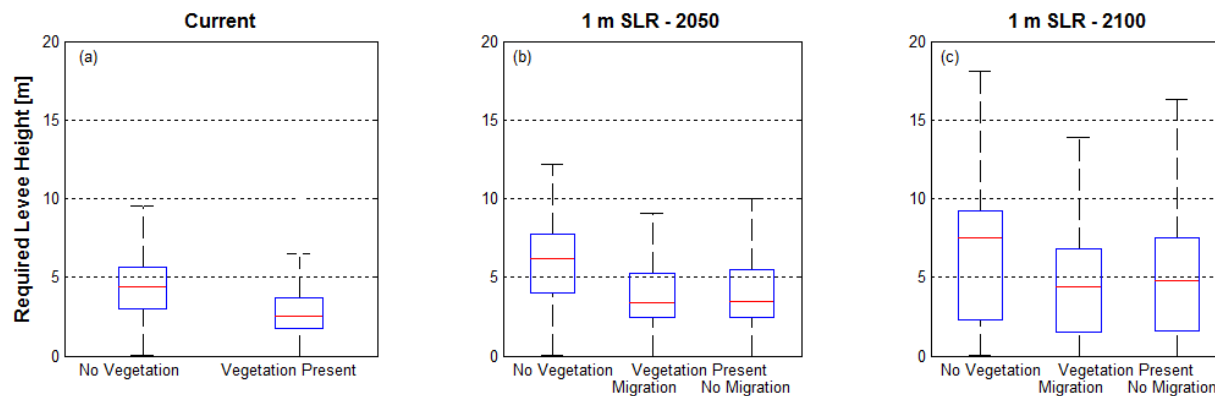
Figure 20: Effect of wetlands on levee height in West Bay, for all storms and SLR scenarios. : Boxplot of levee heights in the presence and absence of vegetation for (a) current conditions, (b) in 2050 and (c) 2100 assuming that wetlands are (not) allowed to migrate.

4.2   Blue Carbon

We computed the total amount of elemental carbon (C) stored by salt marshes – regularly flooded marsh, irregularly flooded marsh and transitional salt marsh – around Galveston Bay for 2004, 2050 and 2100 (Figure 21) by accounting for all the carbon stored in three different pools 1) the above ground biomass, 2) below ground biomass and 3) soil. In 2004, salt marshes stored 29.21 million metric Tons C/ha – 1 metric Ton, or Ton, is equal to 1,000 kg – of which 38% (11 million Tons C/ha) was stored within the three salt marsh habitats that make up 8% of the habitat area. By 2050 the total amount of carbon stored increases by 1.9% to 29.76 million Tons of carbon as marshes migrate landward. Forty-four percent (44%) of this carbon is contained in salt marsh. However, as the rate of SLR increases in the second half of the century the amount of carbon stored decreases by 6.2% from the initial values, a loss of over 1.8 million Tons of carbon storage (Figure 21).

The decrease in the wetlands storage capacity during the second half of the century is a result of salt marsh loss, and resultant carbon emissions, which exceeds the rate of creation of

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new marsh and its ability to sequester carbon. Emission occurs as salt marsh area is inundated

and converted into open water, where the model assumes that carbon currently stored in

sediment is being re-suspended and lost to other systems. This is happening despite the fact that

salt marshes still constitute 8% of the total wetland area and the total amount of carbon stored

within salt marshes increases by almost 14% (1.5 million Tons C) between 2004 and 2100

(Figure 22). Interestingly, the amount of carbon stored in regularly flooded marshes more than

doubles during this period from 3 million Tons/C to 6.5 Million Tons/C, which offsets a loss of

carbon from irregularly flooded marsh, fresh marsh and swamp habitats of more than 3 million
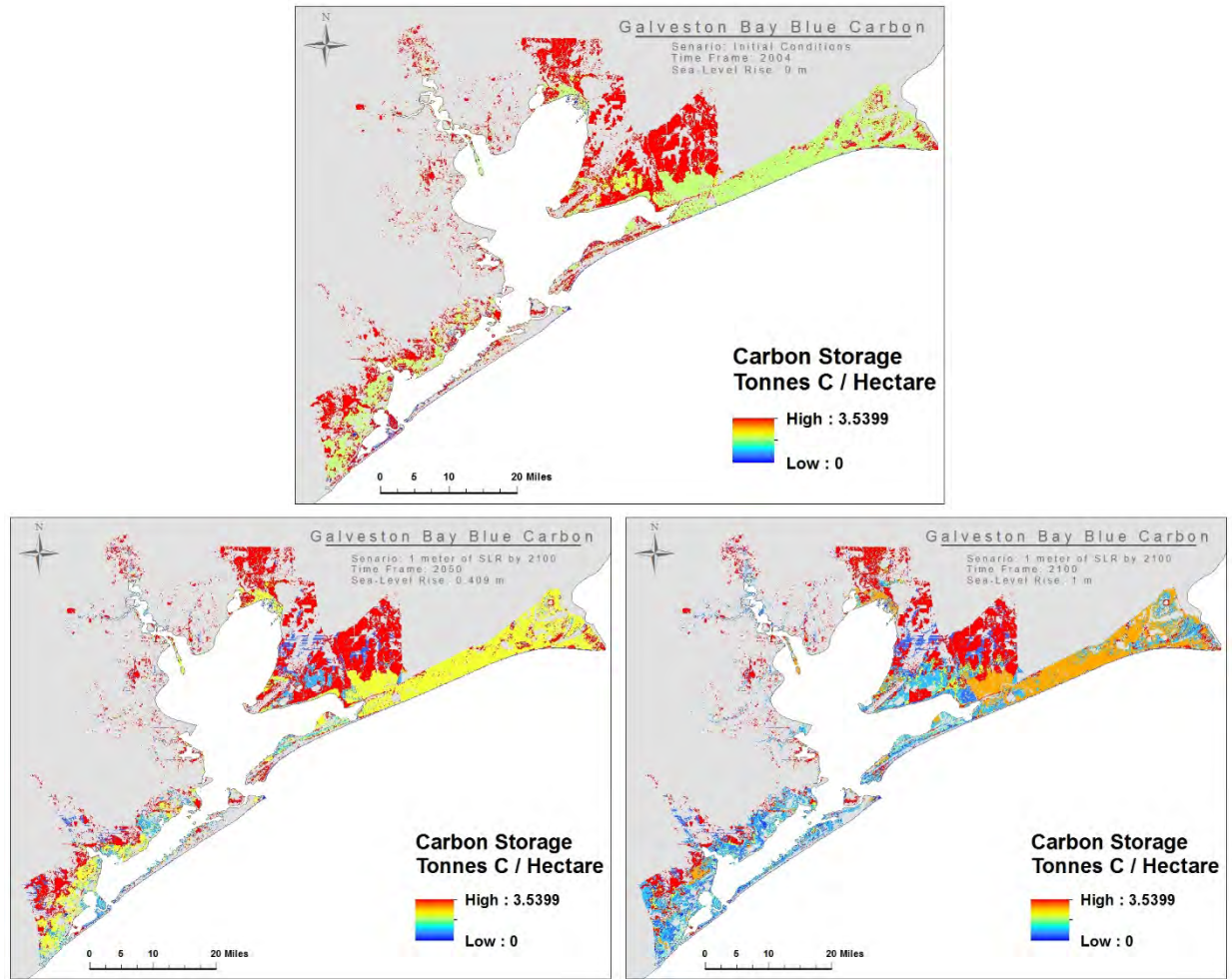
Tons/C.

Figure 21: Carbon stored by wetlands in the Bay in 2004 (top), 2050 (bottom left) and 2100 (bottom right).
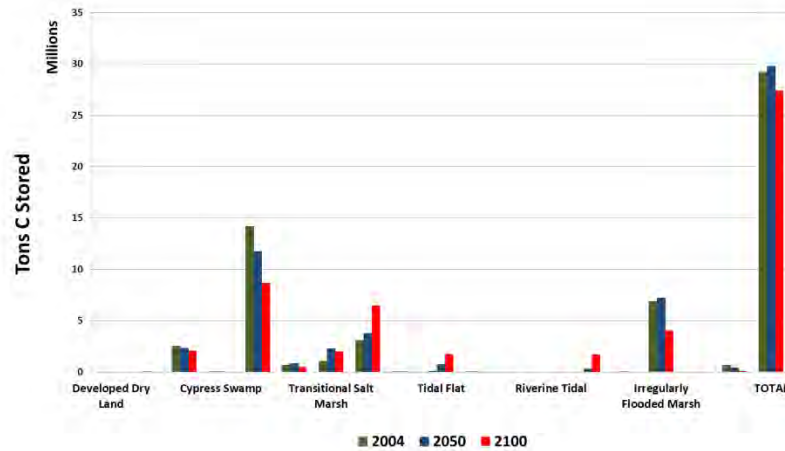
Figure 22: Total carbon stored by different landcover types around Galveston Bay.

We converted the changes in total carbon storage shown above to CO_2 sequestration and emission values for 2050 and 2100 (Figure 23 and Figure 24). We estimate that, between 2050 and 2100, wetlands can store 8.36 million Tons of CO_2, while they can emit 6.35 million Tons. As the rate of SLR increases between 2050 and 2100, an additional 9 million Tons CO_2 are stored, however the loss of existing marsh habitat to inundation causes emissions of CO_2 to increase to 17.6 million Tons. This results in a total predicted loss of 6.6 million Tons of CO_2 from Galveston Bay habitats between 2004 and 2100.

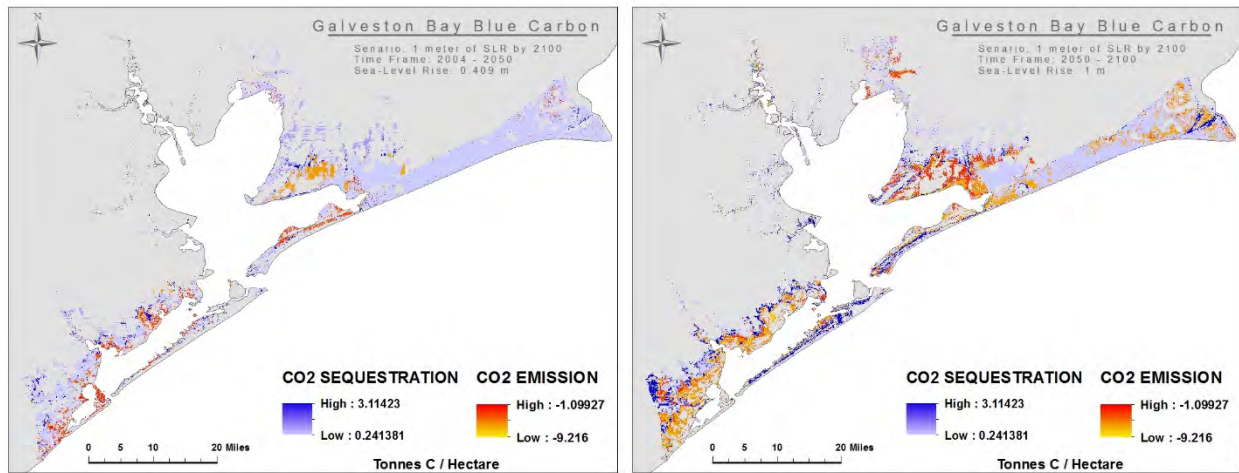Figure 23: Carbon dioxide stored and sequestered by wetlands in Galveston Bay between 2004 and 2050 (left) and 2050 and 2100 (right).
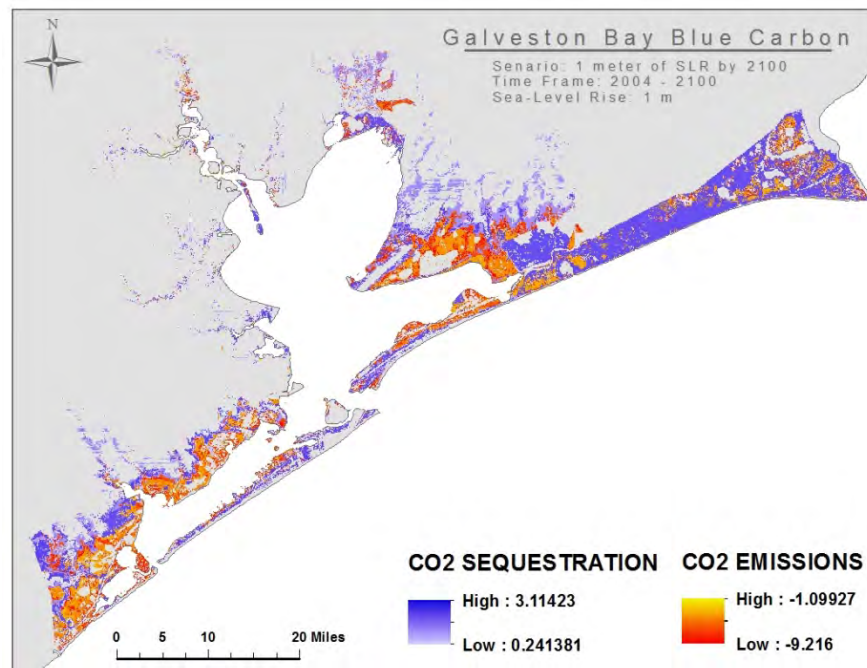

Figure 24: Carbon dioxide stored and sequestered by wetlands in Galveston Bay between 2004 2100.

The greatest losses of carbon (>99% of total emission) occur as salt marsh becomes inundated and converts to tidal flat or estuarine water habitat. As the rate of sea level rise

increases between 2050 and 2100, so does the amount of carbon lost to emissions. On the other hand, salt marshes, including those newly created by migration, account for over 93% of the CO_2 sequestered in Galveston Bay habitats. It is important to remember that, in our model, the loss of carbon from sediments occurs at a much faster rate than it can accumulate in new marsh habitat so there is not a one-for-one exchange based on area. Indeed, we assume that salt marsh accumulates carbon in the soil at a rate of 28.6 Tons C/ha over a 50 year period, while the loss of regularly flooded marsh emits 189 Tons C/ha in the same time period. In summary, we find that, after an initial gain of carbon between 2004 and 2050, the Bay becomes a net emitter of carbon between 2050 and 2100, for a net emission of CO_2 of -6.6 million Tons through the year 2100 (Figure 25). Interestingly, DeLaune & White (2011) found similar results in Louisiana.
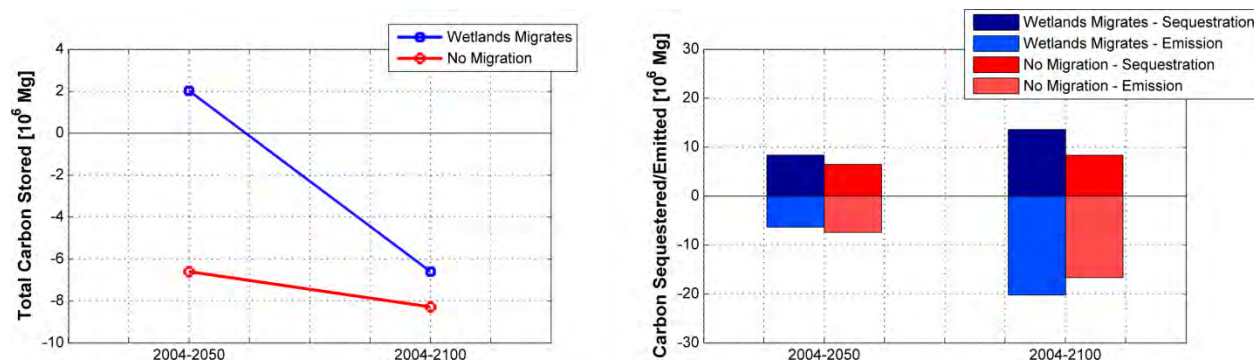
Figure 25: Amount of CO_2 sequestered (positive values) and emitted (negative values) by wetlands for different time horizons. Total amount of carbon stored by wetlands is equal to the sum of carbon sequestered and emitted by wetlands.

We performed the same analysis assuming that marshes were not allowed to migrate. Under this assumption, all new marsh land that was created by SLAMM is removed, but existing salt marshes are allowed to convert to other types of salt marsh habitat within the initial marsh footprint (i.e., from irregularly flooded to regularly flooded). Also, carbon only accumulates where marsh persists, and is lost otherwise. Results indicate that in 2004 the total amount of

carbon stored was approximately 29.21 million Tons, 38% of which was stored within the three salt marsh habitats. By 2050, the total amount of carbon in the project area decreases by approximately 0.9% to 28.96 million Tons of carbon. This decrease is due to the fact that salt marsh area also decreases, as wetlands do not migrate landward. By 2100, where the rate of SLR increases, the amount of carbon stored is projected to be reduced by 7.7% from the initial values, a loss of over 2.2 million Tons C. Note that, since the fresh marsh and swamps habitats are not affected by the "No Migration" restriction, the total reduction in storage capacity is caused by changes in salt marsh habitats, tidal flats, and estuarine waters areas. Without marsh migration salt marsh habitats lose over 10% (1.1 million Tons C) of their initial carbon by 2050, while under the marsh migration scenario they gain 20% (2.2 million Tons C). By 2100, regularly flooded marsh habitat is predicted to gain only 0.5 million Tons C while both irregularly flooded marsh and transitional salt marsh are predicted to lose 3.4 million Tons C and 1.1 million Tons C, respectively. In summary, under the "no marsh migration" scenario, the Bay emits 0.9 million Tons CO_2 between 2004 and 2050. Between 2050 and 2100, an additional 7.4 million Tons of CO_2 are emitted, for a total net loss of 8.3 million Tons of CO_2 (Figure 25).

We translate those estimates of carbon storage and sequestration into net present value of avoided (positive value) or incurred (negative value) climate damages from carbon sequestration/emissions as a result of landcover change from 2004-2100, given in 2010 US dollars. If wetlands are allowed to migrate, they avoid approximately $3 M in climate damage by sequestering carbon between 2004 and 2050 (Figure 26). However, as sea level continues to rise, the bay incurs climate damages of up to $11 M by 2100. Those environmental damages are even greater if marshes are not allowed to migrate, with $23 M in emissions-related damages by 2050 and $31 M of damages by 2100.
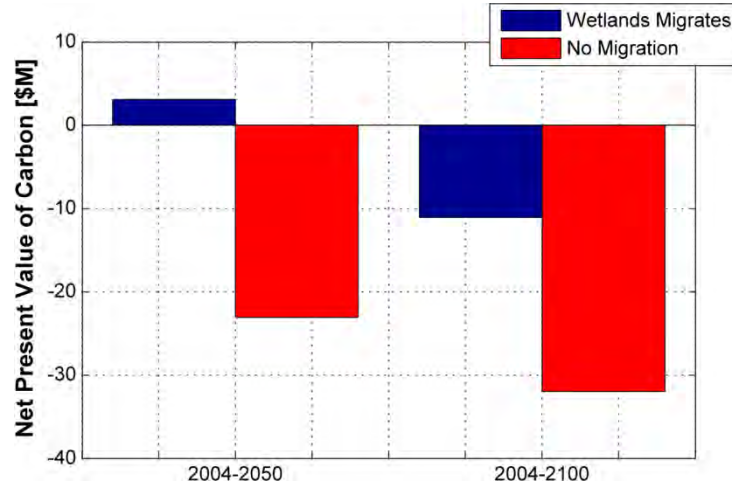
Figure 26: Climate damage from carbon sequestration and emission by marshes in Galveston Bay.

These results indicate that, unless sea level rises slowly, the Bay might become a net emitter of carbon before the end of the century, resulting in significant damages. To gain more insight into the way in which the rate of sea level rise might impact the level of damages incurred by wetlands, we computed the amount of carbon stored and sequestered by the Bay assuming a 0.4 m sea level rise (IPCC mean) and 2 m rise by 2100, which we converted into the amount of damages avoided or incurred (Figure 27). We find that the amount of damages incurred increases dramatically to $86 M if sea level rises by 2 m in 2100. However, if sea level rises moderately to 0.32 m by 2100 (which is lower than current estimates of 0.7 m), wetlands avoid more than $20 M of damages.
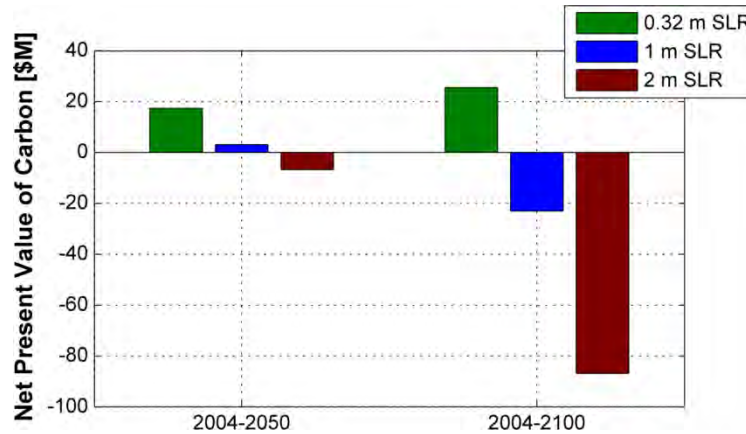
Figure 27: Climate damage from carbon sequestration and emission by marshes in Galveston Bay for different sea level rise scenarios.

As argued in Section 3.2, inputs to the blue carbon model vary widely from region to region. The inputs that we chose are the results of an educated guess that might not reflect the exact (and still unknown) carbon storage and decay rates of habitats in the Bay. In order to determine the relative importance of (1) soil organic carbon content in salt marsh, (2) carbon accumulation rates, and (3) decay rates, we conducted an uncertainty analysis, using a tornado approach, for the 1 m SLR scenario. Table 5 shows the range of values used for each of the input parameters for the tornado analysis.

Table 5: Range of values for the different blue carbon model input parameters used in the uncertainty analysis.

Model Parameter	Minimum	Model Input	Maximum
Soil carbon content [Tons C/ha]	100	222	462
Accumulation [Tons C/ha]	0.18	0.57	8.22
Decay Rate [%]			
Tidal Flat	0.25	0.50	0.75
All other land cover types	0.50	0.75	1.0

We find that when soil carbon content or the decay rates are set at their minimum values, the Bay emits approximately 10 thousand more Tons of carbon compared to when we were using

the default input values. When those two parameters are set at their maximum values, the Bay

stores a few thousand more Tons of carbon. However, if the accumulation rate is set at a

minimum, while the other two remain at their default input value, the Bay stores more than 150

thousand tons of carbon. If this parameter is set at its maximum value, the bay becomes a net

emitter of carbon.

This analysis shows that any estimate of the carbon storage and sequestration potential of

wetlands will contain high uncertainty, as mentioned by Lenart (2009). Also, although the exact

value of the different input parameters is important to correctly estimate the storage potential of

wetlands, the accumulation rate value is the most important input. Changes in its value yield the
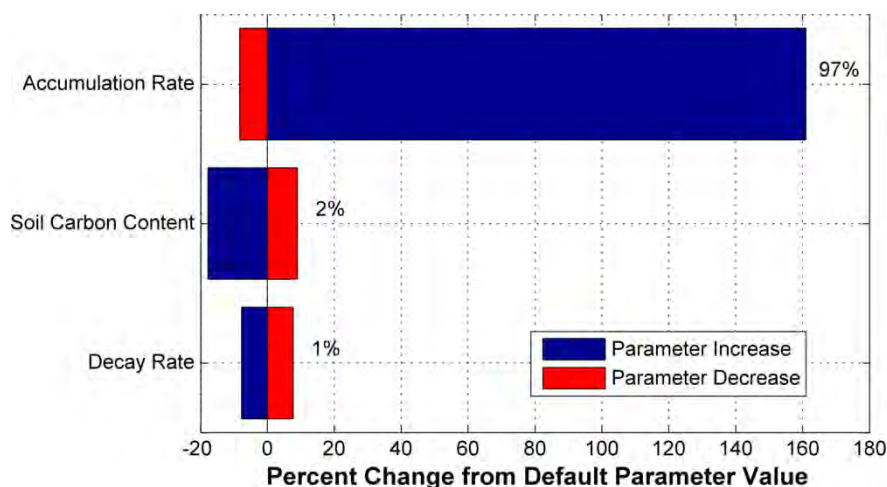
largest swings in carbon storage capacity.

Figure 28: Results of the blue carbon model uncertainty analysis. Results are computed for the
period 2004 to 2100.

4.3 Fisheries

In this section, we quantify the impacts of SLR and changes in marsh habitat coverage on

the selected fisheries in Galveston Bay in two distinct ways. First, we investigate how an

increase in sea level changes the amount of key habitats used by the juvenile fish species

described in Section 3.4.1. Next, we quantify how changes in marsh habitats coverage due to

SLR can translate into changes in landings of shrimp.

4.3.1   Fisheries Habitat Change

Under current conditions, there are nearly 280 km^2 of suitable nearshore habitats

(wetlands, tidal flats and nearshore estuarine waters) for larval and juvenile blue crabs, brown

shrimp, southern flounder and red drum in Galveston Bay (Appendix F). The total footprint of

those habitats will increase as sea-level rises, especially tidal and estuarine water areas.

Additionally, under current conditions, larval and juvenile blue crabs, brown shrimp, southern

flounder and red drum can utilize nearly 1.4 km^2 of marsh edge area in Galveston Bay (Figure 29

and Figure 30). As sea level rises and if wetlands are allowed to migrate, the location of marsh

edge used shifts landward, and the marsh edge area also increases to nearly 2.4 km^2 (a 78%

increase) through 2050 and 4.5 km^2 (a 240% increase) by 2100. This increase is mostly due to

favorable changes in topography, where future edge areas are on flatter surfaces and thus have a

greater surface area than the current edge region.

Figure 29: Marsh edges for the years 2004, 2050 and 2100
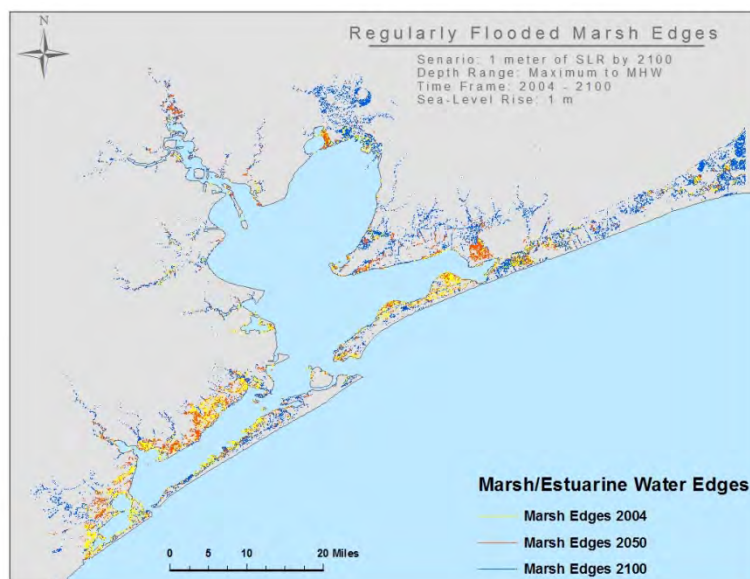

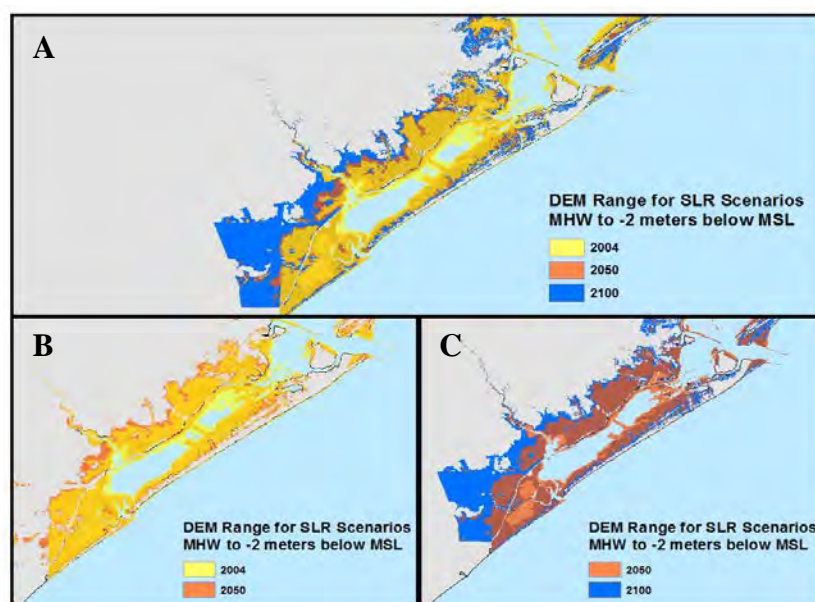Figure 30: Detail of marsh edge spatial extent in West Bay for (A) all three time steps, (B) between 2004 and 2050 and (C) between 2050 and 2100.

4.3.2   Shrimp Population Growth

We investigate how changes in marsh footprint with sea-level rise translate into landing increases of brown shrimp using the shrimp population model described in Section 3.4.2. Model

results indicate that, as anticipated from the habitat modeling results presented in the previous section, the amount of shrimp caught increases as sea level rises, since more marsh habitat is created (Figure 31). Specifically, shrimp catch might increase by 39% by 2050, and by more than 100% by 2100. However, shrimp catch may decrease if marsh area is lost. For example, a 10% loss in wetlands translates into a 7% loss in shrimp catch, and a 20% loss of marsh habitat results in a 17% reduction in the shrimp commercial catches.
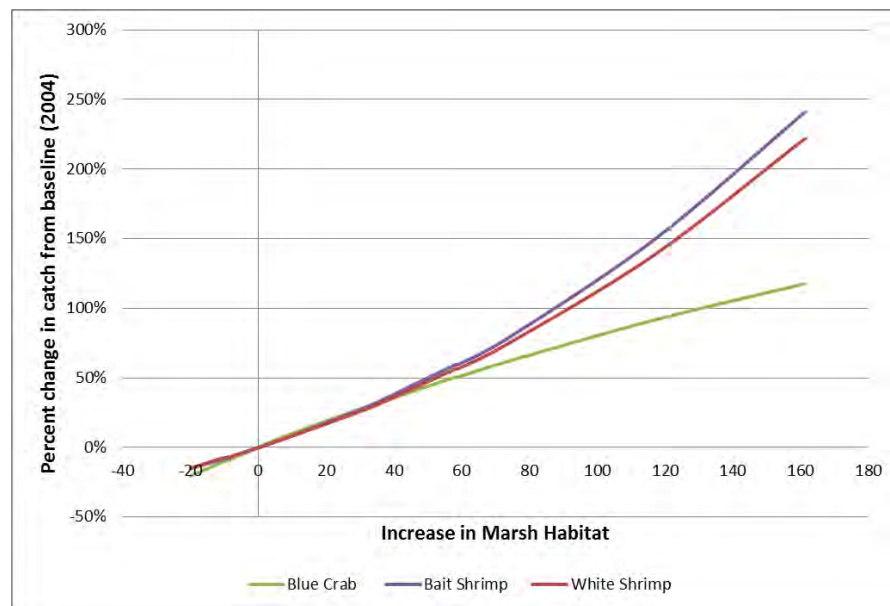
Figure 31: Changes in Shrimp landing as a function of changes in marsh habitats.

5   Protecting Galveston Bay Wetlands

In this document, we have shown that wetlands have the potential to reduce the impacts of storms by reducing damages to properties and by protecting people. Wetlands also store and sequester carbon and provide habitats for fisheries. However, we have also shown that a rise in sea level will create more damages to structures and might cause more damage to the

environment by increasing CO_2 emissions. In this section, we examine how oyster reefs can minimize the impacts of SLR on the Bay's coastline and in the process minimize the loss of wetlands.

Oyster reefs have been shown to moderate nearshore wave energy and protect marshes against wave-induced erosion in the Gulf (Kroeger & Guannel 2014; Scyphers et al. 2011). Similar to artificial submerged breakwaters, oyster reefs have the ability to break nearshore waves and generate a relatively quiet wave climate in nearshore regions. Figure 32 (below) shows a profile of wave height over a 1-D transect in Galveston Bay, in the presence and absence of an oyster reef. The reef reduces the wave height in the nearshore by more than 10 cm. Thus, even though marshes can still be affected by sea-level rise, conveniently situated oyster reefs might help reduce shoreline erosion and the loss of marshes by reducing the number of stressors on wetlands (WPC 2009). This beneficial effect of oyster reefs can last for decades, since oyster reefs can grow vertically as sea-level rises (Rodriguez et al. 2014) and thus maintain the low wave energy climate nearshore.

Roland & Douglass (2005) showed that marsh erosion in Mobile Bay, AL, is partially dictated by the relative height of wind-generated waves in the Bay. Larger waves generate more erosion than smaller waves. To evaluate the potential for reefs to protect marshes in the Bay, we generated a time series of waves in the Bay following the method of Roland & Douglass (2005), using wind speed measured at Pleasure Pier and an average depth of 2.5 m over the profile. Then, we modeled the evolution of those waves on the depth profile shown in Figure 32, in the presence and absence of a reef. In general, reefs can reduce the height of the highest waves in the Bay, and thus reduce their impacts on the wetlands (Figure 33).
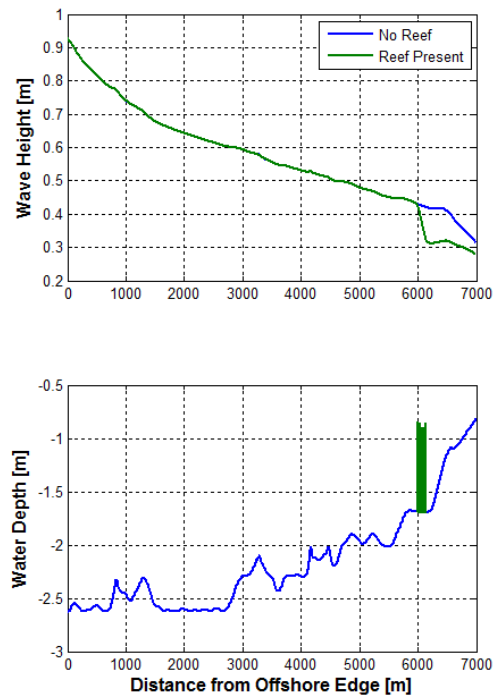
Figure 32: Profile of wave height reduction in the presence of oyster reef on an arbitrary profile in Galveston Bay.
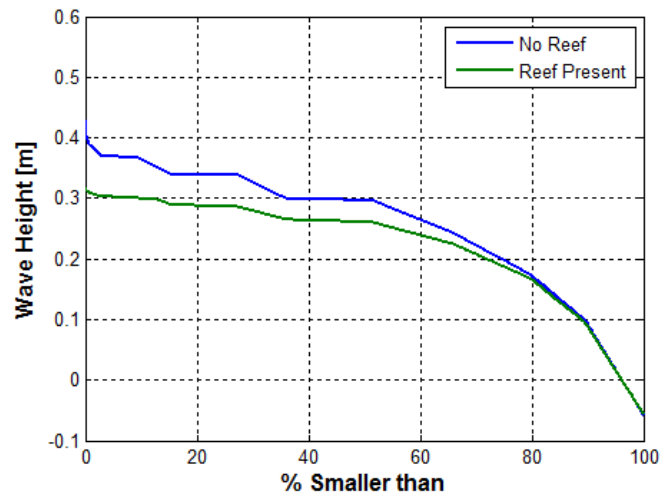
Figure 33: Exceedence profile of wind-wave height in proximity of marshes in the presence and absence of an oyster reef.

Based on this result, we developed an oyster restoration suitability map (Figure 34). This map shows the 80th percentile wave height in different regions of the Bay. According to Roland & Douglass (2005), marshes exposed to H_{80} greater than 0.2 m are more susceptible to erosion than others. Consequently, Figure 34 shows that oyster reef might provide more services in East and Trinity Bay regions than in West Bay.
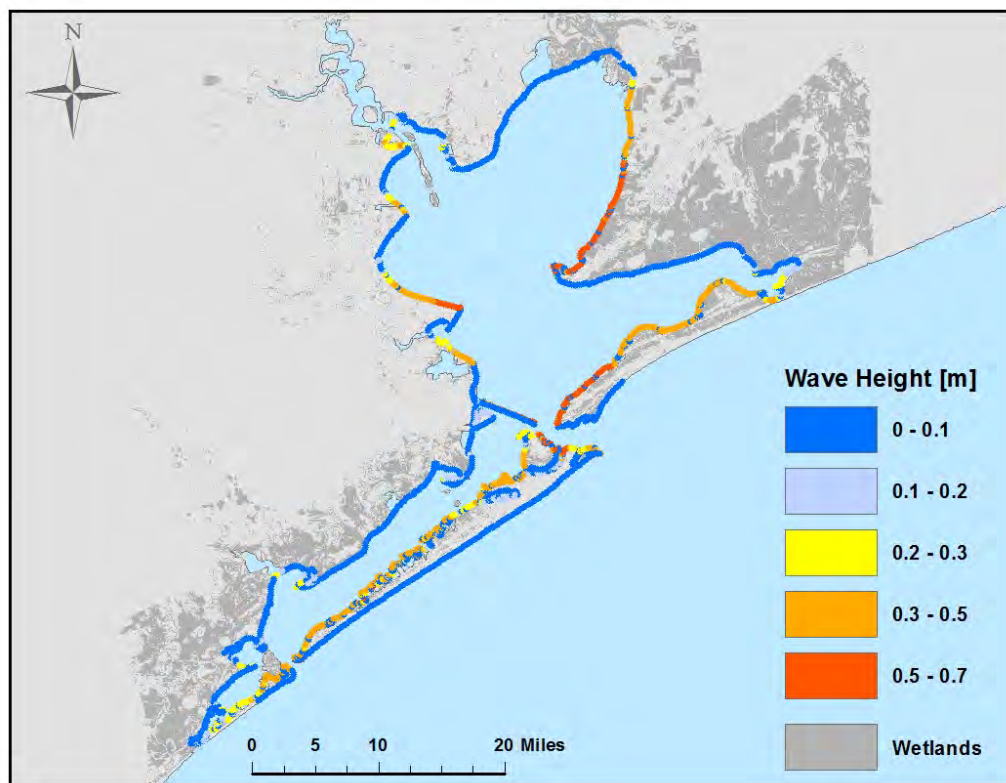
Figure 34: Values of 80th percentile wind-generated wave height inside Galveston Bay.

6   Conclusion

In this document, we quantified the coastal protection, carbon storage and sequestration services, and fisheries services delivered by wetlands in Galveston Bay. We performed this analysis using a marsh footprint representing conditions in 2004 and future marsh footprints for 2050 and 2100, as predicted by the model SLAMM. Those future wetland maps were generated assuming that sea level rises by 1 m through 2100, compared to the 1992 baseline.

We find that wetlands have the capacity to store millions of cubic meters of storm waters, and thus provide a buffer to communities. They also attenuate total water levels during storms in the Bay, and they can avoid millions of dollars in property damage and protect hundreds of people against the impact of storms. However, their ability to moderate the impacts of storms is highly spatially variable, dependent on local topography and on the choice of physical characteristics of the plants. Wetlands can also reduce the construction costs of structures by reducing their height requirements to prevent any overtopping during storms. Those services can continue to be delivered as sea-level rises, as long as wetlands are allowed to migrate and are in relatively healthy conditions. If wetlands are not allowed to migrate, their coastal protection value decreases as sea level rises.

Additionally, we find that currently, wetlands in the Bay avoid millions of dollars of environmental damages by sequestering carbon. The Bay can continue to sequester carbon as sea level rises, as long as that rate of increase is relatively slow. If sea level rises by 1 m in 2100, which is likely given current local estimates, hundreds of millions in environmental damages may result from the release of more CO_2, by submerged carbon-rich sediments, than wetlands can store and sequester.

Finally, we find that most fisheries species might benefit from larger areas of available habitats as sea level rises. Additionally, for brown shrimp, an increase in marsh habitat translates into net gains in landing. Thus, it is likely that all commercial and recreational fisheries activities in the Bay might benefit from a rise in sea level, if marshes are allowed to migrate.

However, as mentioned in the main text, this analysis contains some uncertainty. For example, we used average estimates of habitat conditions and physical characteristics around the Bay to compute the coastal protection services delivered by wetlands. We also assumed that the location, type and value of properties will not change through time. Better habitat data and more accurate predictions of property owners' behaviors are likely to improve the protection service estimates of wetlands. Furthermore, the evaluation of carbon storage and sequestration is highly sensitive to the plants accumulation rate, which is an important unknown characteristic in the Bay. There is also a fair amount of uncertainty in the fate of carbon released by sediments once submerged wetlands disappear. As mentioned earlier, despite the advances made herein, there are still many research gaps to accurately quantify the carbon storage and sequestration capacity of wetlands (Lenart 2009). Finally, despite the fact that we find that fisheries might benefit from more habitats as sea-level rises, fisheries stand to be directly impacted to a much greater degree by changes in temperature and salinity, two variables not included in our analysis. These two environmental factors regulate key life cycle events for many marine species (Copeland 1966; Anger 2003; Ward 2012; Hildebrand & Gunter 1953; Zein-Eldin & Renaud 1986; Buzan et al. 2009; Needham et al. 2012) and may have a greater impact on fisheries than any increase in habitat area caused by sea level rise.

As a consequence, results can be interpreted as an estimate of the storm damage mitigation services provided by wetlands and their capacity to improve fisheries or store carbon.

Nevertheless, our results and approach is one of the first attempts at a comprehensive accounting of different services delivered by wetlands in the face of rising seas. We show that an increase in sea level might increase the impacts of storms and degrade the environment by increasing releases of CO_2 in the atmosphere. Preserving wetlands and allowing them to migrate naturally can reduce the impacts of those stressors and might improve fisheries in the Bay.

7   Future Work

This study has demonstrated that wetlands in Galveston Bay have the potential to moderate the impacts of storms currently, and in the future, as sea-level rises. It also demonstrated that wetlands can sequester large amount of carbon and sustain fisheries. However, this study has also identified a series of research needs to better quantify the services delivered by wetlands.

The quantification of ecosystem services by wetlands requires an accurate accounting of the type and location of different species of marsh plants in and around Galveston Bay under different sea-level rise scenarios. Herein, we used SLAMM outputs for a 1 m SLR scenario, which contain some inaccuracies as SLAMM predicts that non-developed and developed areas will be converted to marsh in 2050 and 2100. It is unlikely that such land conversions will occur based on discussions with local partners. More accurate predictions of marsh footprint at different time horizons will improve the quantification of the changes in the delivery of services by wetlands.

The estimates of total water level in the presence of wetlands might be improved by using a 2D model for wave and storm surge evolution, such as SWAN and/or ADCIRC (Luettich et al. 1992; Booij et al. 1999). However, this analysis has shown the need for better representations of the physical parameters of plants in order to improve the accuracy of the model outputs. In addition, the valuation of the protective role of wetlands might be improved with more information about the location and characteristics of the different structures and the composition of the households around Galveston Bay.

Estimates of the carbon storage and sequestration capacity of the wetlands might be improved with the inclusion of more accurate information about the ability of the soil and marsh plants in and around the Bay to store carbon. The model used in the analysis might also be improved by considering non-linear rates of accumulation or loss.

Finally, the fisheries models might be improved by including the role of more environmental attributes (i.e., sediment type distribution, salinity and temperature data) in both the habitat and population models. In particular, we did not include any information about how temperature or salinity will change in 2050 and 2100 compared to the current baseline. Additionally, any future work should include estimates of population changes for other important species in the Bay, such as blue crabs, red drums, etc.

8 Acknowledgements

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Appendix A Land Class Classification for Storm Surge and Wave Modeling

In order to model storm surge and wave evolution in and around Galveston Bay, we created a land cover map of the Galveston Bay region large enough to include all land classes affected by a storm. We use this map to generate a database of Manning *N* coefficients to model flow resistance and total water level. We also used this map to gather information on the physical characteristics of major vegetation classes present in the region.

We created a land cover map of the Bay region by merging the land cover map provided by WPC (2011) with the Coastal Change Analysis Program (C-CAP) land cover data (NOAA

Coastal Services Center (NOAA-CSC) 2006). We re-classified those data based on the 2009

National Wetland Inventory (NWI) scheme, (U.S. Fish and Wildlife Service (USFWS) 2009) as
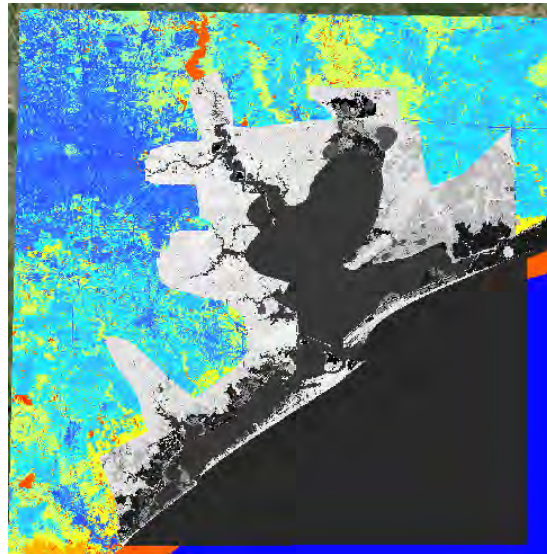
shown in Figure A-1.

Figure A-1: Merged land cover data (WPC (2011) and C-CAP dataset). The grey scale indicates the land cover data used as base conditions in SLAMM modeling and the red to blue color scale indicates the 2006 C-CAP land cover data that was merged with the SLAMM dataset for a more complete footprint.

Next, we associated each land class type with a corresponding Manning N coefficient

(Table A-1). We generated this database from a compilation of various data published in the

literature (Mattocks & Forbes 2008; Bunya et al. 2010; Agnew 2012; Wamsley et al. 2009;

Wamsley et al. 2010). We ensured that estuarine marshes had a lower N value than salt marsh,

assuming that plant species in fresh marshes tend to be taller, more rigid, and more dense and,

therefore, more dissipative. We did not base our choice on model calibration to observed storms

(Wamsley et al. 2010).

Table A-1: Manning Coefficients Associated With Different Land Classes.

SLAMM Code	SLAMM NAME	Manning N	CCAP Code	Land Cover Type	Manning N
1	Developed Dry Land	0.093	02	High Development	0.093
3	Swamp	0.074	03	Med Development	0.093
4	Cypress Swamp	0.074	04	Low Development	0.093
5	Inland Fresh Marsh	0.055	05	Open Development	0.03
6	Tidal Fresh Marsh	0.055	06	Cultivated	0.05
7	Scrub/Shrub	0.074	07	Pasture/Hay	0.05
8	Regularly Flood Marsh	0.045	08	Cropland	0.05
10	Estuarine Beach	0.03	09	Deciduous Forests	0.074
11	Tidal Flat	0.03	10	Evergreen Forests	0.074
12	Ocean Beach	0.03	11	Mixed Forest	0.074
15	Inland Open Water	0.02	12	Scrub/Shrub	0.074
16	Riverine Tidal	0.03	13	Palustrine Forest Wetland	0.074
17	Estuarine Water	0.02	14	Palustrine Scrub/Shrub Wetland	0.074
19	Open Ocean	0.02	15	Palustrine Emergent Wetland	0.055
20	Irregurlarly Flooded Marsh	0.055	16	Estuarine Forested Wetland	0.074
22	Inland Shore	0.03	17	Estuarine Scrub/Shrub Wetland	0.074
23	Tidal Swamp	0.074	18	Estuarine Emergent Wetland	0.045
			19	Unconsolidated Shore	0.03
			20	Bare Land	0.03

			21	Water	0.02

Finally, we used the land cover map to gather information on the physical characteristics of vegetation in and around the Bay. Because of the scarcity of information on physical parameters for all vegetation types present in Galveston, we grouped different land class types as described below and summarized in Table A-2:

- Swamp and forested wetland: cypress swamp, swamp, tidal swamp, palustrine forested wetland, and estuarine forested wetland are grouped together and given the physical characteristics of cypress, which is the dominant species in swamps and forested wetland (Lester & Gonzalez, 2011).

- Scrub/shrub areas: scrub/shrub wetland, estuarine scrub/shrub wetland, palustrine scrub/shrub wetland are characterized by woody vegetation less than 5m in height. While this may contain a variety of species, including young trees, we assigned to all vegetation types in that category the physical characteristics of Buttonbush.

- Forests: deciduous forests, evergreen forests, and mixed forests are grouped together and considered upland forest.

- Low salt marsh: estuarine emergent wetland (CCAP) and regularly flooded marsh (SLAMM) are grouped together and attributed the physical characteristics of *Spartina alterniflora*.

- High salt marsh/brackish marsh: tidal fresh marshes and irregularly flooded marshes were grouped together and represented by marsh hay cordgrass (*Spartina patens)*.

- Fresh Marsh: palustrine emergent wetland (CCAP) and inland fresh marsh (SLAMM) are grouped together and represented as giant cutgrass (*Zizaniopsis miliacea)*, which is the

primary species occurring in fresh marshes (Lester & Gonzalez 2011)). Due to a lack of data, we approximated the physical properties of this species by using data for big cordgrass.

- Agriculture, Bare land, Open Water: land cover of these types are represented in the model with their corresponding Manning coefficient (see Table A-1).

Table A-2: Physical Characteristics of Vegetation Associated with Different Land Classes

Land Cover Class	Height [m]	Diameter [m]	Density [units/m^2]	Source
Swamps and Forested Wetlands	15	1.5	0.0172	Arborday.org
Scrub/Shrub	2.5	0.1016	0.0861	University of Florida, Alabama Forestry Department, and Davesgarden.com
Forests	10	0.2	0.0861	FEMA
Low Salt Marsh	0.8	6.4×10^{-3}	172	FEMA guidance for the Chenier Plain Region
High Salt Marsh/Brackish Marsh	0.57	5.8×10^{-4}	3,585	FEMA guidance for the Chenier Plain Region
Fresh Marsh	1.22	8.14×10^{-3}	107	FEMA guidance for the Chenier Plain Region

Appendix B Marsh Acreage

To investigate the extent to which conversion of marshes to development has reduced marsh coverage, we merged historic (1989 and 1979) NWI marshes with the current NWI marshes (2004) to represent a possible high coverage future marsh footprint. Data from 1956 were not included because they are inadequate to distinguish marshes versus non-marshes. From this merged dataset, we removed areas where the current LULC was classified as "Commercial", "Industrial", "Residential", "Governmental/Medical/Educational", and "Roads" (Developed Areas), according to the regional growth forecast produced by H-GAC (2013). That forecast lacks coverage in Chambers County in GIS format. Therefore, we did not compute any loss of marsh footprint in that area. This approximation is reasonable considering the low population density in that region.

We find that, based on historical data, the marsh might have covered more than 2,162 km^2 before present. This means that approximately 123.89 km^2 of marshes may have been lost due to direct conversion to development between 1979 and today (current marsh area is approximately 2,038 $km^{2)}$. An additional 77 km^2 may be lost due to future conversion to development by 2040. Closer comparison of the land cover map in 2040 as predicted by H-GAC and of the current marsh footprint shows that most of this loss will occur in small, isolated patches. No cohesive and compact area of marsh is projected to be lost because of this forecasted development.

As mentioned in Section 2.2, sea level rise is likely to be the main climate variable expected to impact coastal wetlands in Galveston Bay. This would result in the conversion (and sometimes net loss) of one wetland type to another (Warren Pinnacle Consulting (WPC) 2011). In this section, we investigate the response of salt marshes to sea level rise.

Salt marsh zonation in Galveston Bay is tightly linked to tidal elevation, with the low marsh dominated by *Spartina alterniflora* and the upper marsh by *Spartina patens*. Together these habitats colonize available substrate from mean sea level to the upper edge of the saltwater inundation zone (Feagin and Wu 2006). As sea levels rise, inundation frequency will change and marshes will be required to reestablish proper zonation to maintain equilibrium in flood regimes. In general, marsh that becomes permanently inundated as sea levels rise will likely die, and marshes in areas with projections of high sea level rise combined with small tidal ranges are most vulnerable (Craft 2009, Stevenson and Kearney 2009). Consequently, salt marshes in Galveston Bay, which is a microtidal bay with a relatively high rate of SLR, are projected to incur significant losses due to sea level rise (Warren Pinnacle Consulting (WPC) 2011).

Salt marshes may adapt to sea level rise through two mechanisms, vertical accretion of the marsh platform and inland migration. Areas of Galveston Bay with ample sediment supply may outpace sea level rise through accretion. However, in other areas, particularly West Galveston where inorganic sediment is limiting (Ravens et al. 2009) and accretion rates fall well below sea level rise rates (Ravens et al. 2009, Feagin et al. 2010), it might be harder for salt marshes to adapt, or keep up. In that case, marsh will migrate inland, and adjacent non-salt marsh areas will be converted to salt marsh. The potential for marsh to migrate inland depends on a number of biophysical factors including slope, substrate, and the robustness of the marsh plant. However, the biggest barrier to marsh migration is human development. Indeed, roads, agricultural lands, building and other types of development are either protected from the impacts of SLR in ways that exclude marsh migration, or cannot be converted to marsh habitat.

In this section, we use the InVEST Habitat Climate Adaptation model (beta version) to predict where salt marsh migration may be blocked or facilitated, based on tidal elevation, slope,

accretion rate, erosion, sea level rise and land use/land cover information. This tool was developed using the same principles as the SLAMM model, but focuses solely on salt marsh areas. Consequently, it predicts future marsh footprints in regions located between mean sea level and the 'salt boundary' (upper edge of tidal inundation) where the contiguous upper and lower saltmarsh grow. This region encompasses the SLAMM categories of regularly and irregularly flooded marsh.

First, we estimate the migration potential of salt marshes in Galveston Bay based on the land class present landward of the marsh habitats. For each land class, we determine a migration potential value, based on the likelihood that this land class can be converted to marsh habitat (Table B1). Results of this analysis are shown in Figure B-1, where we show regions where marsh is more likely than others.

Table B1: Marsh migration potential factors as a function of land class.

| General Classification	Migration Potential	Examples of land class used
Developed Land	0.0	Low to high intensity development, multiple family dwelling units, etc…
Open Water	0.0	Inland and marine water
Roads	0.0	Major highways, etc.
Agricultural Land	0.3	Cultivated land, etc.
Bare land	0.5	Bare land, grassland, open pasture, etc.
Forested Land	0.7	Evergreen forest, mixed forest, deciduous forest
Freshwater Wetland	1.0	Palustrine emergent wetland, palustrine forested wetland
Scrub/Shrub Wetland	1.0	Scrub/shrub wetland
Tidal Wetland	1.0	Estuarine emergent & brackish wetland, irregularly & regularly flooded marsh, etc.
Shoreline	1.0	Unconsolidated shoreline, beach
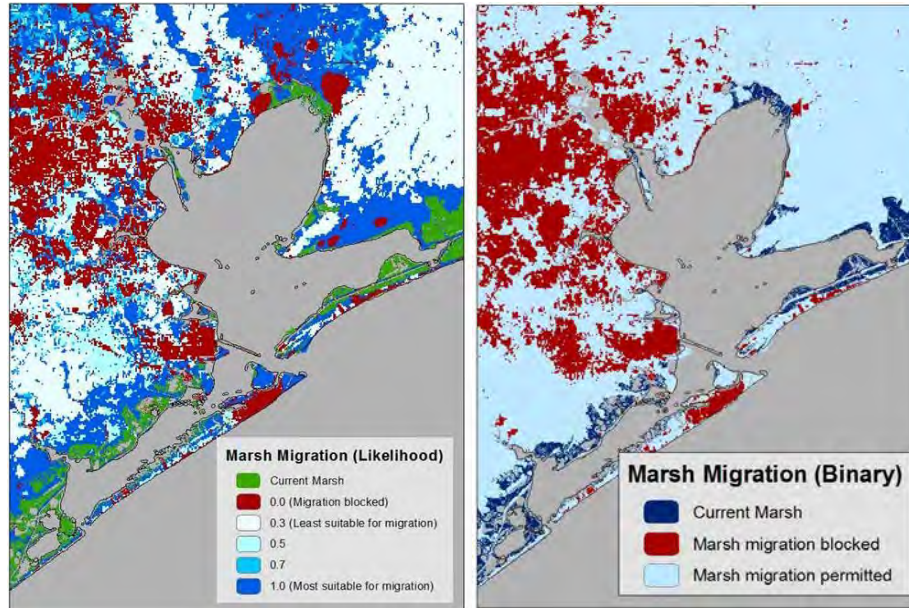
Figure B-1: Likelihood of marsh migration in Galveston Bay. Left: Outputs for land classes classified based on the likelihood that marshes will be permitted to migrate across different land classes. Right: Results where migration factors less than 1 are considered to be 0.

Next, we use the relationship between tidal elevation and marsh zonation to identify the region that marshes may inhabit under future sea level conditions (Figure B-2). Salt marsh may inhabit part or all of this region, depending on its vertical accretion rate. In addition to inland migration, vertical accretion of the marsh platform through sedimentation or biological accumulation of peat is a second adaptation strategy that may allow marshes to outcompete sea-level rise. Here, we entered site-specific accretion rates to offset inundation from sea-level rise. This represents a simplification over the SLAMM model, which incorporates the relationship between tidal elevation and marsh productivity to calculate a more dynamic representation of accretion rates. Results are shown in Figure B-2 and B-3.
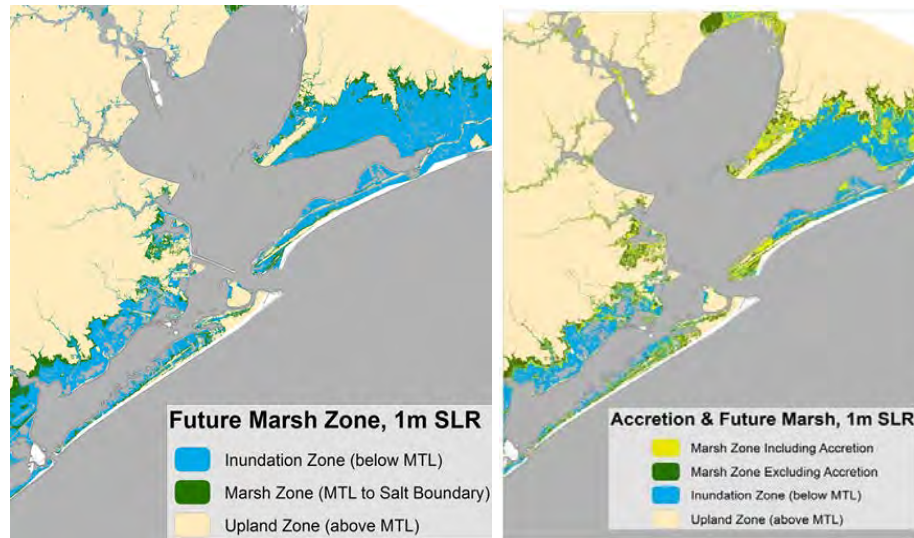
Figure B-2: Potential zone of future marsh location. Left: future marsh zone corresponding to the region between MSL and the 'salt boundary' given 1m SLR in Galveston Bay. Right: future marsh zone in the Bay computed by taking into account vertical accretion.

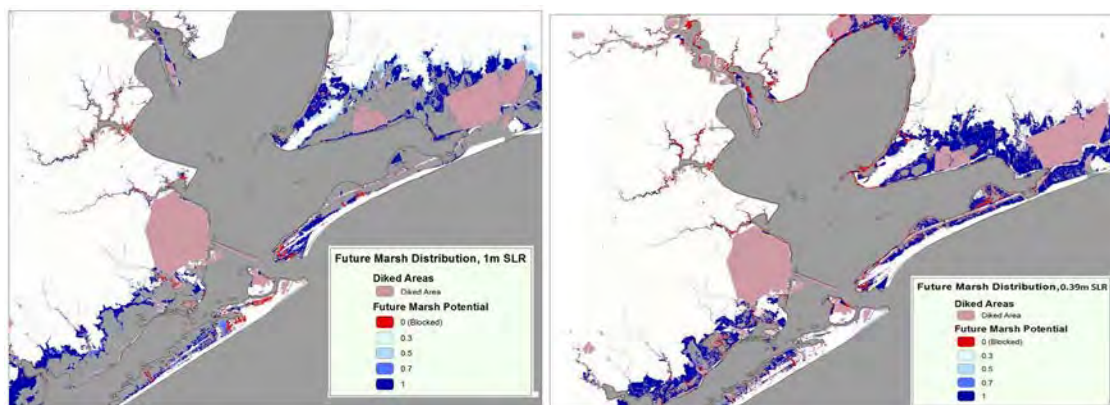**Future marsh distribution**

Figure B-3: Potential future marsh distribution in 2100 with (a) 1 meter SLR, (b) 1 meter SLR for a section of Galveston Island, (c) 2 meter SLR, and (d) the A1B mean (0.39 m) SLR scenarios. Marsh contained in diked areas (in pink) is assumed to remain unchanged until dikes are overtopped (>2m SLR). Marsh in red is blocked from migration by development, etc., and is assumed lost in 2100 (this marsh is automatically removed from the future marsh footprint). Marsh in varying shades of blue corresponds to 'likelihood' of marsh migration across the landscape based on land use (darker blue=increased likelihood). This depicts where marsh may be most imperiled by existing land use.

Validation with SLAMM model

Results from the InVEST tool were validated against those from the SLAMM model. We find good alignment between SLAMM and InVEST predicted marsh distributions for the 1m, 2m and A1B mean SLR scenarios (the latter two now shown) (Figure B-4). InVEST incorporates three SLAMM habitat types (regularly flooded, irregularly flooded and transitional salt marsh) into one habitat footprint (saltmarsh). There are some deviations between SLAMM and InVEST, which may be accounted for by slight differences in the DEM or SLAMM's more complex accretion feedback loop. Furthermore, while marsh erosion by waves will be incorporated into the InVEST tool, it was not included in this analysis.
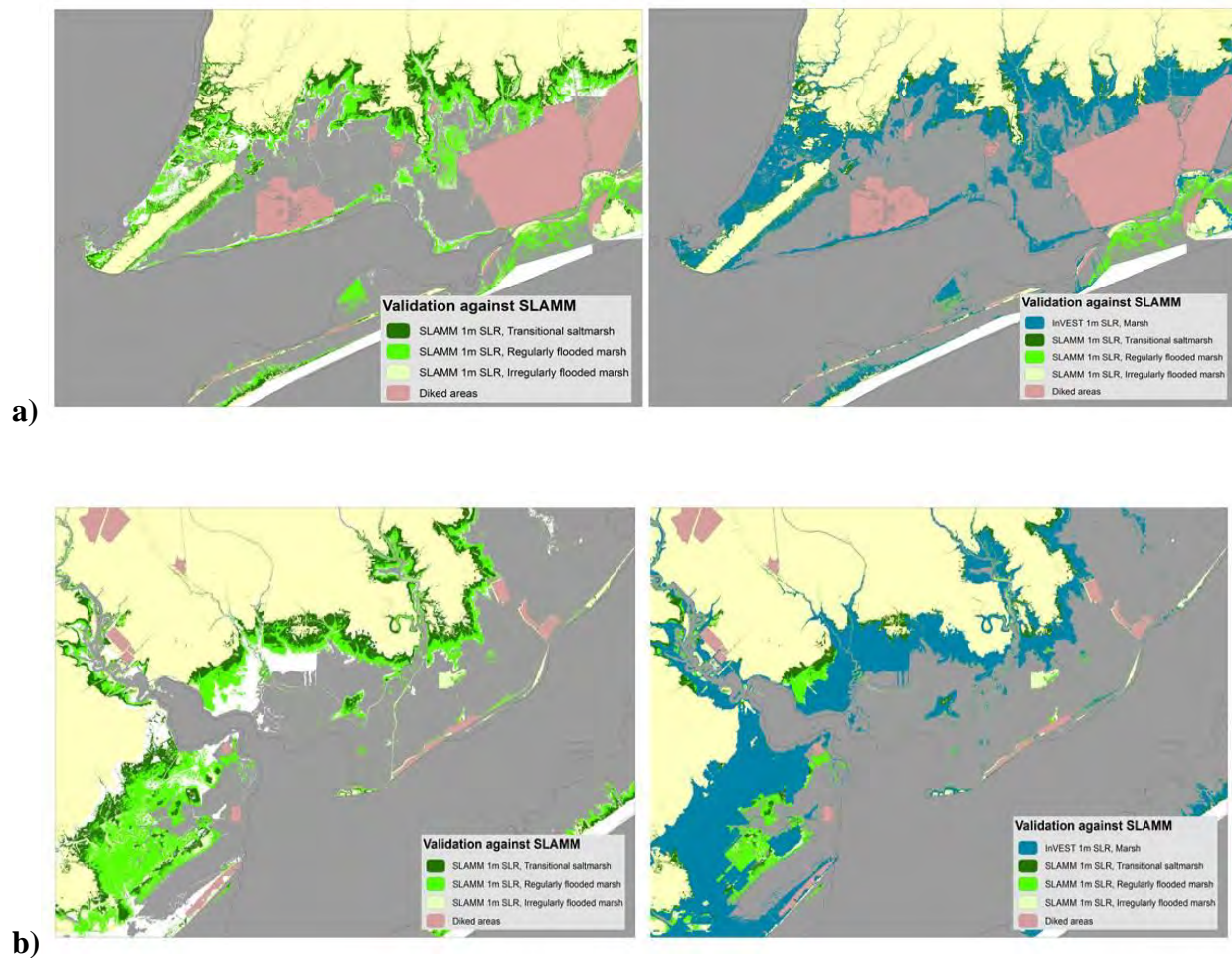

Figure B-4: Validation of InVEST climate adaptation model against SLAMM results in East Bay (a) and in West Bay (b).

SLAMM Model Outputs

WPC (2011) computed changes in habitat and wetland coverage in Galveston Bay, for different sea-level rise scenarios. Table B2 shows changes in land class types for the 1 m SLR scenario.

Table B2: SLAMM Outputs Depicting Changes in Land Classes for 1 m Sea-Level Rise

Land cover Category	SLAMM Code	2004	2050		2100	
		Area [thousands of Ha]	Area [thousands of Ha]	Change from 2004	Area [thousands of Ha]	Change from 2004
Developed Dry Land	1	43.8	43.3	-1%	41.3	-6%
Undeveloped Dry land	2	143.1	132.0	-8%	116.8	-18%
Nontidal Swamp	3	9.2	9.3	1%	8.6	-6%
Cypress swamp	4	0.0	0.0	-4%	0.0	-18%
Inland fresh marsh	5	47.5	44.7	-6%	35.0	-26%
Tidal fresh marsh	6	2.3	3.0	31%	1.7	-25%
Transitional salt marsh/Shrub	7	7.6	10.1	33%	12.7	69%
Regularly flooded salt marsh	8	8.7	13.2	52%	20.1	131%
Estuarine beach	10	1.4	0.7	-52%	0.2	-82%
Tidal flat	11	1.5	9.6	559%	16.5	1030%
Ocean beach	12	0.5	0.4	-7%	0.7	52%
Inland open water	15	10.5	8.6	-19%	8.1	-23%
Riverine tidal open water	16	1.3	0.6	-57%	0.4	-73%
Estuarine open water	17	162.0	170.5	5%	195.0	20%
Open ocean	19	348.7	348.7	0%	348.9	0%
Irregularly flooded salt marsh	20	24.3	18.6	-23%	8.4	-65%
Inland shore	22	2.0	2.0	-2%	1.9	-6%
Tidal swamp	23	2.4	1.6	-34%	0.4	-85%

Estuarine Wetlands		45.3	46.4	3%	43.4	-4%
Freshwater Wetlands		56.7	54.0	-5%	43.7	-23%
Open Water and Beaches		527.8	541.0	2%	571.6	8%
Land		186.9	175.3	-6%	158.1	-15%

Appendix C Storm Surge

Coastal vegetation has the capacity to reduce storm surge by increasing the amount of flow energy lost to bottom friction dissipation. Bottom friction over submerged lands is a function of the Manning coefficient associated with different land classes. We estimate the potential for wetlands to attenuate storm surge in West Bay by running the AdCirc storm surge model (Luettich et al. 1992) for three storms conditions (Table C-1): a tropical storm (storm "B0"), and a category 1 ("Z1") and 2 ("Z2") hurricane. All three storms had the same idealized track (Figure C-1), and the storm parameters were held constant throughout the storm simulation. Although these storms only impact West Bay and are idealized, they give an indication of the surge reduction potential of wetlands in the Bay. We find that wetlands have the capacity to reduce storm surge in the Bay, but that the amount of surge reduction is highly dependent on the local topography.

Table C-1: Storm Parameters.

Simulation ID	Central Pressure [mb]	Maximum Wind Speed [kts]	Radius of Maximum Wind [nm]	Storm Source*
B0	996	55	25	Eduardo 2008
Z1	982	75	25	Claudette 2003
Z2	970	85	25	Unnamed 1949

*The source indicates actual storms in the area from which central pressure and maximum wind speeds were obtained

Figure C-1: Storm track in Galveston Bay. The green dots indicate the eye path for the completed simulations. The hatched polygon and blue solid polygon represent the bands where the wind speeds are within 10 and 5% of the maximum offset from the storm path, respectively.

Figure C-2 shows the maximum water surface elevation generated Z2 storm induced surge, in the absence of vegetation. Comparisons of surge elevation in the presence and absence of wetlands in that region (Figure C-3) show that the wetlands have the potential to reduce surge by as much as 20 cm. However, the role of wetlands, captured through the range of computed attenuation values, is not uniformly distributed along the vegetated area. Wetlands are more effective at reducing storm surge on the west side of West Bay than on its east side.
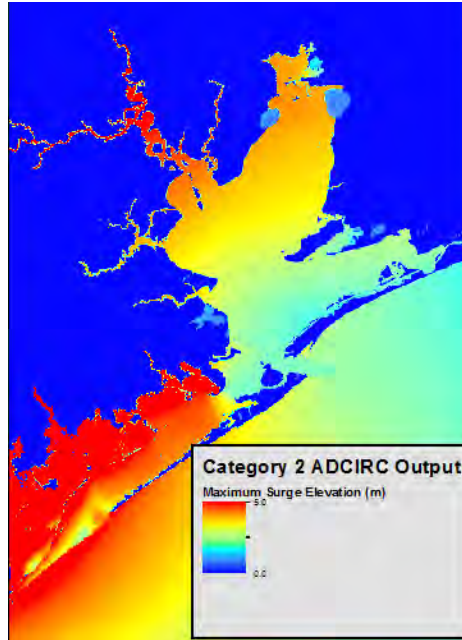
Figure C-2: Storm surge elevation in Galveston Bay generated by Storm Z2, in the absence of vegetation.
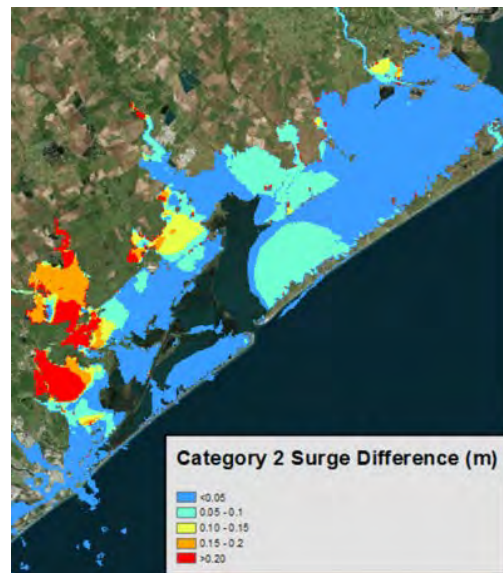

Figure C-3: Difference between maximum surge elevations in West Bay with and without habitat.

To compute the average rate of storm surge reduction by wetlands in West Bay, we extracted profiles of the overland surge along the different transects shown in Figure C-3, for the three storms modeled, B0, Z1 and Z2. Next, we calculated the slope of the difference between surge computed in the presence and absence of habitats plots in regions where such difference was approximately linear. However, because of the complex nature of the topography in that region, most of the profiles analyzed produced different profiles that we could not confidently use to compute a linear attenuation rate. Only a few transects (circled by a thick black oval in Figure C-3) captured a clean profile from which an attenuation rate could be inferred. Figure C-4 shows an example of such profiles, for storms Z1 and Z2.
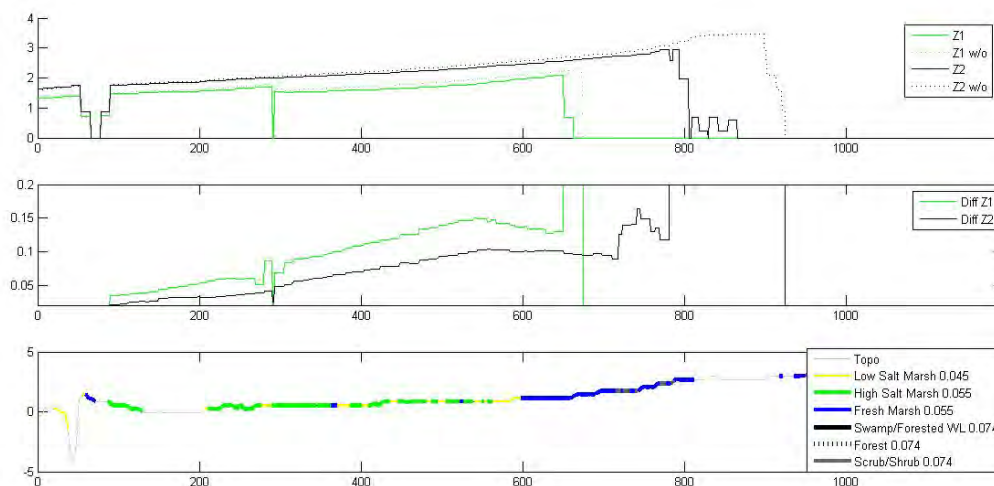

Figure C-4: Example profiles of storm surge in West Bay, in the presence and absence of wetlands. The value of maximum surge elevation associated with Z1 (green) and Z2 (black), with (solid) and without habitat (dashed). The maximum surge dips to 0 at topographic high points not inundated by the surge (top). The difference between with and without habitats for Z1 (green) and Z2 (black) (center). The topographic profile overlain with different colored segments representing the different habitats and corresponding Mannings N-values along the profile. The line representing forest is dashed because that is the only habitat not excluded in the without habitat simulation. Also, there is no forest present along this profile (bottom).

For the 5 transects used in our analysis, we estimated surge attenuation rates varying from 19.8 cm/km of wetland (storm B0) to 15 cm/km (storm Z2). Our results are on the higher side of the range of attenuation rates reported for other wetlands systems, but seem to be reasonable (Table C-2). Outputs also show the difficulty in coming up with simple rules of thumb and general guidelines about the role of wetlands at reducing storm surge.

Table C-2: Attenuation rates of storm surge by wetlands

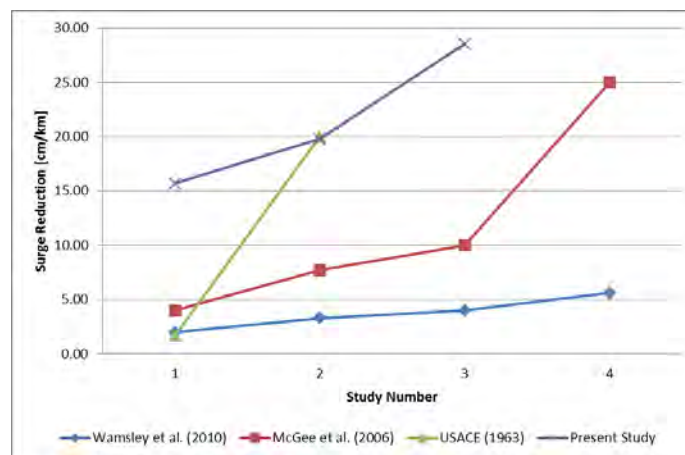| Study	Attenuation Rates [cm/km]	Forcing	Manning N	Method
Wamsley et al. (2010)	2.0, 3.3, 4, 5.6	960mb, 900mb (Cat 3, Cat 5?)	0.035 – 0.055	ADCIRC-
McGee et al. (2006)	4.0, 7.7, 10, 25	Hurricane Wilma (no direct hit)	NA	Field Observations
USACE (1963)	1.7 – 20.0	NA	NA	NA
Present Study	19.8, 28.5, 15.7	TS, Cat 1, Cat2	0.045 – 0.074	ADCIRC

Figure C-5: Rates of surge reduction as measured by different studies

Appendix D Description of Terrestrial and Blue Carbon Models' Inputs

InVEST Terrestrial Carbon model

We used the InVEST terrestrial carbon model (version 2.5.6) to estimate the amount of carbon stored at the initial time step in each land cover class within the project site boundaries. Model inputs include the 2004 land cover map from SLAMM and estimates of the amount of carbon in each land cover category. We included in our analysis the first three pools and excluded the detrital layer because of a lack of data. Additionally, because local data are difficult to obtain, we used data from other geographies in the northern Gulf of Mexico region (Figure D-1) to determine the amount of organic carbon in each of these pools.
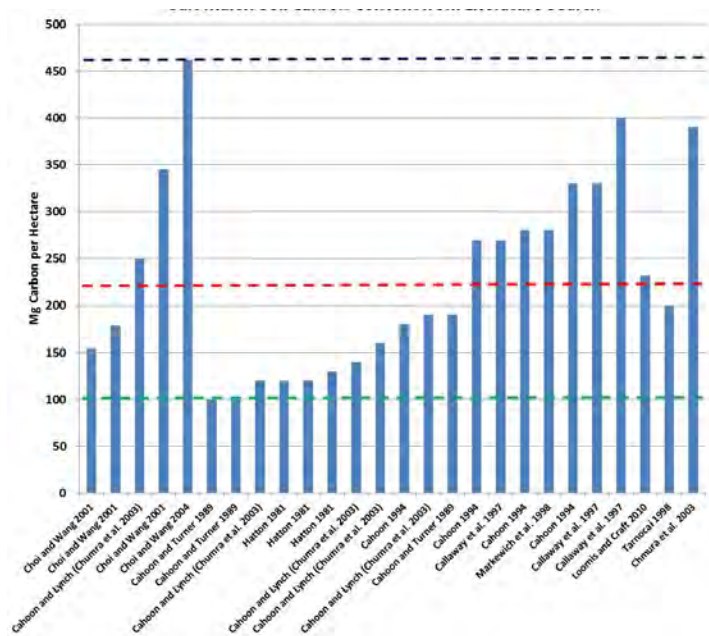

Figure D-1: Salt marsh soil carbon content values from published literature. The red horizontal line represents the value used herein. The blue and green horizontal lines represent maximum and minimum reported soil carbon values.

We assigned an average soil carbon content value of 222 Tons C/ha to the regularly flooded salt marsh landcover class. This value is below the global average value of 390 Tons

C/ha reported by Chmura et al. (2003), whose database was dominated by soil carbon values collected in tropical climates. We assigned soil carbon values to the remaining habitat types by using conversion factors between the regularly flooded marsh category and the remaining wetland habitats (Table C1). Consequently, for irregularly flooded and transitional marshes, which are considered high and mid marsh, respectively, we assigned a conversion factor based on the relationship between high, mid and low marsh carbon content found in Choi et al. (2001). Additionally, because of a lack of data for the different swamp classifications in Texas, we grouped all swamp landcover categories (non-tidal, cypress and tidal). We then computed a conversion factor for this group based on the relationship between the global average for salt marsh and the global average for all coastal wetlands values in Chmura et al. (2003). For fresh marsh landcover types, we assumed that the carbon content in fresh marshes is approximately 1.21 times that found in salt marshes (Craft et al. 2009). For tidal flats, we calculated the conversion factor using the relationship between the average carbon content of salt flats and wetland soils found in Yu et al. (2012). Although tidal flats have very little plant biomass they do store carbon more effectively than estuarine open water habitats. Finally, for estuarine beach and ocean beach, we allocated an organic carbon content of 10 Tons C/ha based on data published by Yu et al. (2012).

Table C1: Wetland carbon conversion factors

Land cover Category	Conversion Factor	Soil Carbon [Tons/ha]	Source
Regularly flooded salt marsh	1.00	222	
Nontidal swamp	1.10	244	Chmura et al., 2003
Cypress swamp	1.10	244	Chmura et al., 2003
Inland fresh marsh	1.21	269	Loomis and Craft, 2010
Tidal fresh marsh	1.21	269	Loomis and Craft, 2010
Transitional salt marsh	0.52	115	Choi et al., 2001
Tidal flat	0.25	56	Yu et al., 2012
Irregularly flooded salt marsh	0.45	100	Choi et al., 2001
Tidal swamp	1.10	244	Chmura et al., 2003

Table C2: Elemental carbon in carbon pools

Land cover Category	Carbon in Biomass [Tons/ha]		Soil Carbon [Tons/ha]
	Above Ground	Below Ground	
Developed dry land	0	0	0
Undeveloped dry land	0	0	0
Nontidal swamp	20	10	244
Cypress swamp	42	24	244
Inland fresh marsh	20	10	269
Tidal fresh marsh	20	10	269
Transitional salt marsh	20	10	115
Regularly flooded salt marsh	20	10	222
Estuarine beach	0	0	10
Tidal flat	0	0	55
Ocean beach	0	0	10
Inland open water	0	0	0
Riverine tidal open water	0	0	0
Estuarine open water	0	0	0
Open ocean	0	0	0
Irregularly flooded salt marsh	20	10	100
Inland shore	0	0	0
Tidal swamp	20	10	244

Blue Carbon Model

The InVEST blue carbon model (beta version) computes carbon accumulation and decay in soils over time. However, because this model was initially designed for salt marshes, the only

land cover classes that are modeled for accumulation and decay over time are the three marine

wetland or salt marsh categories: regularly flooded, irregularly flooded and transitional salt

marshes. Accumulation and decay that also occur in the fresh marsh and swamp land cover

classes were not incorporated into the analysis.

The accumulation rate used herein was set at 0.573 TonsC/ha/yr, which was converted

from the average rate of 210g CO_2/m^2/yr for tidal marshes in the coterminous United States

(Chmura et al., 2003). This rate is multiplied by the number of years between two time steps to

calculate the total amount of carbon sequestered at a given time horizon. Carbon stored in the

soil also decays through time. Here, we assigned a decay value of 0.75 TonsC/ha over 50 years

for all land cover types that convert to other land cover classes, except for when salt marsh

converts to tidal flats. For this latter type of transition, we assigned a decay value of 0.50. Since

the blue carbon model is not iterative and only calculates the accumulation and decay in one

step, these values represent a loss of 75% and 50%, respectively, of the total carbon in the

previous salt marsh class by the next time step.

From the above mentioned input layers, the model generates two maps, one depicting the

amount of accumulated carbon and the other the amount of carbon lost, at different time

horizons.

Appendix E Modeled Fisheries Species' Habitat and Environmental Conditions

Preferences

In this appendix, we describe the life histories, commercial importance and the habitat and environmental parameters of the fisheries species studied.

Blue Crab

The blue crab (*Callinectes sapidus*) commercial fishery is one of the most important commercial fisheries in Texas with landings in 2010 of 3.4 million pounds, or 3.1 million dollars (ex-vessel). Movements of blue crab within estuaries are directly related to life cycle stages, season and environmental factors (Van Den Avyle, M.J. & Fowler.. 1984; Perry & McIlwain 1986). Female crabs migrate from estuary waters to coastal waters or inlets releasing their eggs, which develop in nearshore Gulf waters through their later larval stages (Perry and McIlwain, 1986). Once they reach the megalopal stage they migrate back into estuarine waters settling as juveniles primarily into marsh habitats, but also into seagrass meadows and other shallow nearshore areas (Pardieck et al. 1999; Van Heukelem 1991; Ward 2012). Eventually, they vacate their primary nursery habitats and spread into the upper reaches of the estuary before moving into deeper water as adults (Ward 2012; Perry & McIlwain 1986).

Water depth does not influence juvenile distribution, and they can live in water temperatures between 2 and 35ºC (Tagatz 1969), with salinities ranging from 0.0 to well over 30 ppt (Pardieck et al. 1999; Hammerschmidt 1982). However, they prefer soft, muddy sediments and tend to prefer marsh and seagrass habitats to estuarine waters (Evink 1976; Perry & Stuck 1982).

Brown Shrimp

Brown shrimps (*Farfantepenaeus aztecus*) are also an important commercial fishery in Texas and, together with other commercial shrimp species, form the largest fishery in the state. Over 27 million pounds of shrimp caught in 2010, which is worth an ex-vessel price of over 91 million dollars (NMFS 2010). Brown shrimps occupy a wide range of habitat types and are found in both marine and estuarine habitats along the Texas coast. They typically migrate to offshore waters during spawning periods. After hatching, young larvae spend their early stages in offshore marine waters and migrate into shallow waters of estuaries and coastal wetlands (Turner & Brody 1983). After entering the estuary, they transform into juveniles where they settle into seagrass meadows and marsh edge zones (Turner and Brody, 1983). As the juveniles grow into adults they migrate into deeper estuarine waters, eventually returning to deeper offshore Gulf waters to spawn.

Juvenile brown shrimps can survive fluctuations in salinity from 10 to 20 ppt (Gunter, Christmas & Killebrew 1964; Zein-Eldin & Renaud 1986). They can also survive in waters as cold as 2 ºC and to over 32ºC (Venkataramaiah 1971; Christmas, J.Y., Langley & Devender 1976; NOAA 1998). However, post-larval and juvenile shrimps prefer shallow (< 2m) regions with mud or silt bottom substrates rich in organic matter. They can also be found in mixed, sandy and shell shallow substrate types (NOAA 1998; TPWD 2002). Finally, juvenile brown shrimps prefer seagrass meadows and salt and brackish marsh areas over other habitat types (Turner & Brody 1983; NOAA 1998; Zimmerman, R.J. & Zamora 1984), with annual populations of brown shrimp found to be directly correlated with temperature and the area of suitable marsh habitat (Barrett & Ralph 1976).

Red Drum

Red drums (*Sciaenops ocellatus*) support an important recreational fishery in Texas – commercial fishing for red drum in Texas was banned in 1981 to reduce fishing pressure. Recreational catches are not well documented in Texas and actual landings cannot be verified; however, the recreational fishery industry in Texas is said to contribute billions of dollars in yearly revenue to the Texas economy (NOAA 2012).

Red drums are an estuarine-dependent species that utilize estuarine habitats primarily during their larval and juvenile stages but can be found in deeper waters as adults (Buckley 1984). Adult red drums migrate from marine waters to areas adjacent to channels and passes for spawning, where their eggs and larval stages can be transported into their estuarine nursery habitats. After hatching, red drum larvae settle in both vegetated and un-vegetated shallow water estuarine habitats and along marsh edges (Rooker et al. 1998). Once they reach the juvenile stage they stay in these protected areas of the estuaries, preferring grassy clumps or muddy bottoms (Simmons & Breuer 1962). Older juveniles are found along shallow marsh edges or in marshes until they mature when they start spending more time in marine waters (Yokel 1966).

Juvenile red drum tolerate a wide range of salinities and temperatures. They have been found in salinities ranging from 0 to 50 ppt and temperatures ranging from 13 to 28 ºC (Perret, W.S., Weavers, J.E., Williams, R.O., Johansen, P.L., McIlwain, T.D., Raulerson & Tatum 1980; NOAA 1998). Juveniles tend to be found in waters swallower than 2 meters (Buckley, 1984) in a wide variety of bottom sediment types including mud, silt, clay and sand.

In Texas larval and juvenile stages of red drum have been found to be associated with seagrass meadows and marsh edge habitat (Yokel, 1966). Studies have also indicated that marsh

and seagrass habitat restoration can increase survival and growth rate of red drums (Levin & Stunz 2005; Stunz et al. 2002).

Southern Flounder

Southern flounder (*Paralichthys lethostigma*) is another important fishery species in Texas, both recreationally and commercially; however its economic impact is much less than the blue crab and brown shrimp. Commercial landings of unclassified flounder species were around 26,000 pounds in 2010, worth an ex-vessel value of around $62,000 (NMFS 2010, commercial landings).

Southern flounders recruit to estuaries early in their life cycle and are the dominant type of flounder in the Gulf of Mexico that use estuary nursery grounds (McEachran & Fechhelm 2006). Adult southern flounder are commonly found along the shores, bays and lagoon of coastal Texas. They become gravid and move offshore into the gulf during the fall and early winter to spawn, although some stay within the estuary all year without becoming gravid (Ginsburg 1952). Once released in the Gulf the eggs and resulting larval stages remain in marine waters for less than two months before migrating into the estuaries (Arnold et al., 1977) where juvenile southern flounder settle in shallow estuarine waters.

Juvenile southern flounder can withstand salinities ranging from 0.5 to 30 ppt. They have been observed in water temperatures ranging from 5 to over 30 ºC. Juveniles prefer mud or silt substrates but also can be found occasionally on sandy substrates (Enge & Mulholland 1985; Packer et al. 1999). They are known to settle in marsh and seagrass habitats but are also found in non-vegetative areas close to the water marsh interface (Glass et al. 2008). Because of this

association with marsh and seagrass habitats juvenile southern flounder in Texas are generally

found in waters shallower than 2 meters (Gutherz 1967; Glass et al. 2008; Packer et al. 1999).

Appendix F Fishery Habitat Change

We assess changes in the location and quantity of land and marine habitats to estimate current and future amounts of preferred habitat for four different commercially and recreationally important fish species in Galveston Bay: (1) Blue Crab (*Callinectes sapidus*), (2) Brown Shrimp (*Farfantepenaeus aztecus*), (3) Southern Flounder (*Paralichthys lethostigma*), and (4) Red Drum (*Sciaenops ocellatus*). Since these species are present in the Bay's wetlands primarily during their larval and juvenile stages, we only consider these life stages in our analysis (Appendix E).

Figure F-1 shows the modeling framework adopted herein. The most important and common habitat and environmental parameters that define where these larval and juvenile fish can live are sediment type, minimum and maximum depth, temperature, salinity and turbidity. To apply the InVEST fisheries habitat model in Galveston Bay, we collected information about the environmental and habitat preference ranges for each species of interest (Appendix E). However, because of the poor resolution of publicly available data, we did not include sea surface temperature, salinity and turbidity in our analysis.

From bathymetric and topographic, sediment and living habitat location information, the model first extracts the maximum livable area for a given species' life-stage based on depth preferences. Those are waters between -36 m and MHW for blue crab, and between -2 meters and MHW for the three remaining species. Next, the model extracts regions within the defined area that have the types of sediment preferred by that species (Table F-1). However, because the sediment dataset that we used is static and does not contain any future projection of sediment types in the Bay, it was only used to delineate the maximum offshore extent of habitat suitable for the different fisheries listed previously; sediment preference was not used to compute the amount of suitable habitats in the future. Then the model incorporates information about location

and quantity of available living habitats for those species by generating habitatat suitability maps based on the species preferences (Table F-2).

We obtained changes in the areal extent of estuarine and wetlands areas from SLAMM outputs (Warren Pinnacle Consulting (WPC) 2011). Lastly, using information from the different bathymetric and environmental layers, the model computes the amount of suitable habitat for that species. We conducted this process for the different time horizons considered, based on updated land cover maps.
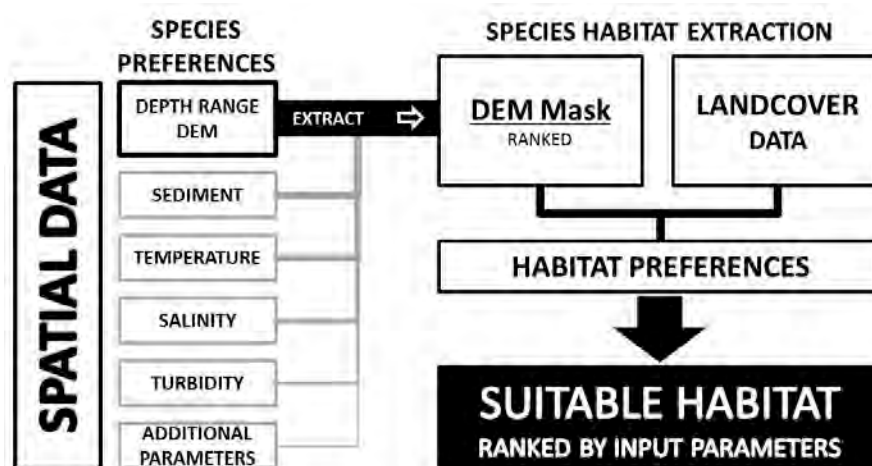
Figure F-1: Fishery Habitat model framework

Table F-1: Surficial sediment preferences by species used for the spatial fishery modeling in Galveston Bay, Texas.

Common Name	Sediment Classes and Species Preferences							
	Clayey Silt	Gravel	Gravelly Sediment	Sand	Sand, Clay, Silt	Sandy Silt	Silt	Silty Clay
Brown Shrimp	x						x	x
Red Drum	x			x	x	x	x	x
Southern Flounder	x			x	x	x	x	x
Blue Crab	x					x	x	x

Table F-2: SLAMM habitat preferences by species used as parameters in the spatial fishery modeling in Galveston Bay, Texas.

Land cover Category	SLAMM Code	Species Habitat Preference			
		Blue Crab	Brown Shrimp	Red Drum	Southern Flounder
Swamp	3				
Cypress swamp	4				
Inland fresh marsh	5				
Tidal fresh marsh	6	x			
Transitional salt marsh	7	x	x	x	x
Regularly flooded salt marsh	8	x	x	x	x
Estuarine beach	10				
Tidal flat	11	x		x	
Ocean beach	12				
Ocean flat	13	x			
Inland open water	15				
Riverine tidal	16	x		x	
Estuarine water	17	x		x	
Tidal creek	18	x		x	x
Open ocean	19				
Irregularly flooded salt marsh	20	x	x	x	x
Inland shore	22				
Tidal swamp	23				
Vegetated tidal flat	25	x	x	x	x
Back shore	26				

Results from the model provide a first estimate of the changes in habitat for each key species under different sea-level rise scenarios (Figure F-2, Figure F-3, Figure F-4). Under a 1 m SLR scenario, the total amount of habitat, and marsh habitat in particular, increases for all fisheries (Figure F-5), with the greatest increases occurring between 2050 and 2100. If wetlands are allowed to migrate, riverine tidal areas are the only habitat that will shrink; they will lose slightly more than two-thirds (-67%) of their surface area by 2100. The area of tidal fresh marsh and regularly flooded marsh will more than double in 2100, and the amount of irregularly flooded and transitional salt marsh will increase by 31 and 95%, respectively. In addition, blue

crabs will benefit from a modest increase in estuarine area (+5% by 2050 and +21% by 2100), while red drums will see modest changes (-4% by 2050 and +6% by 2100).
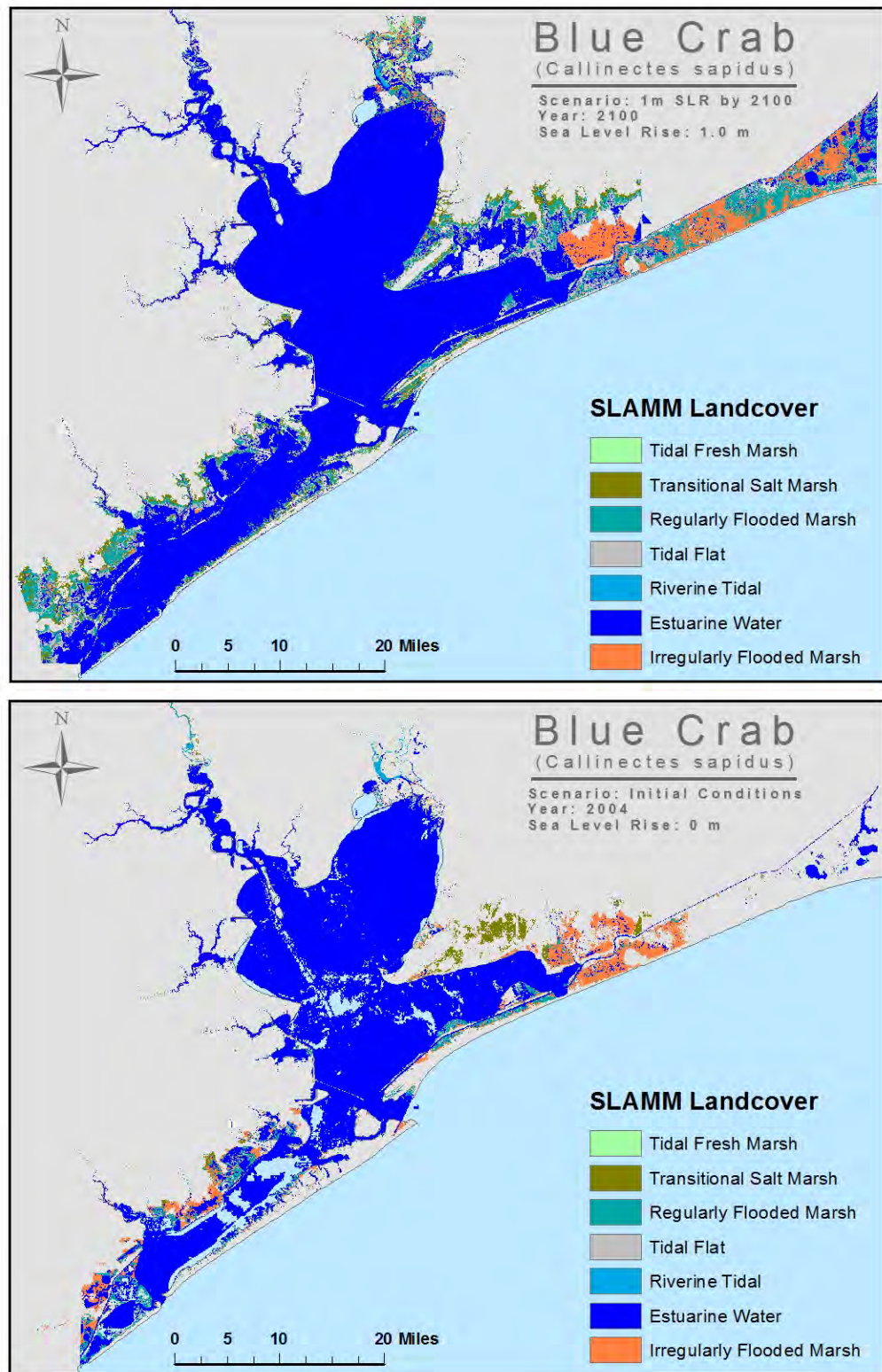
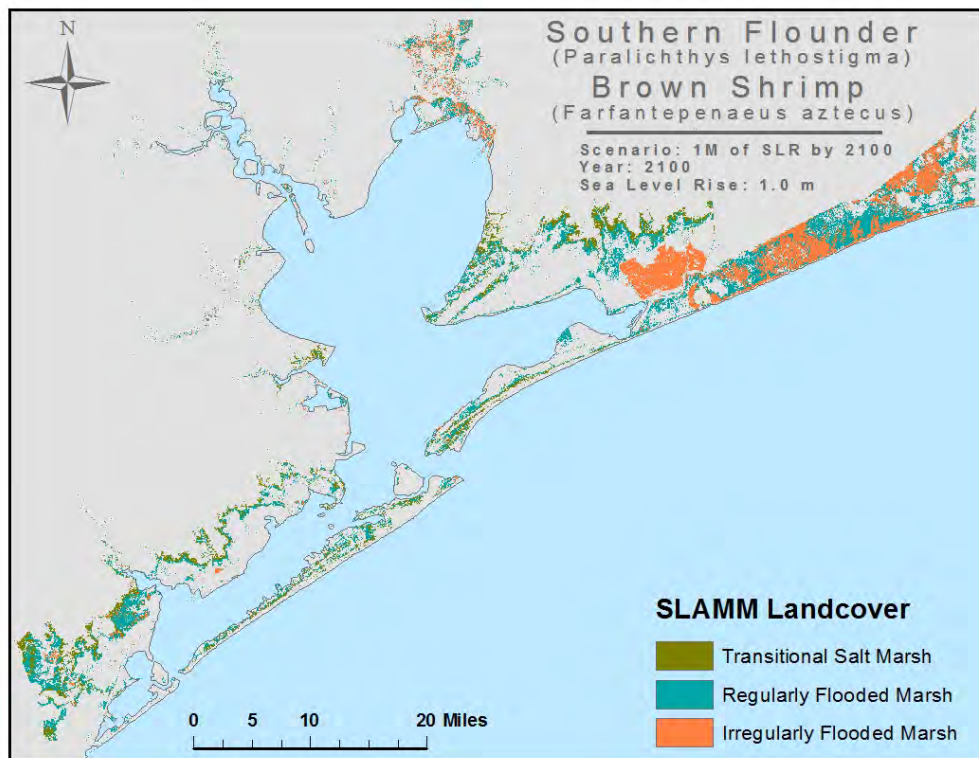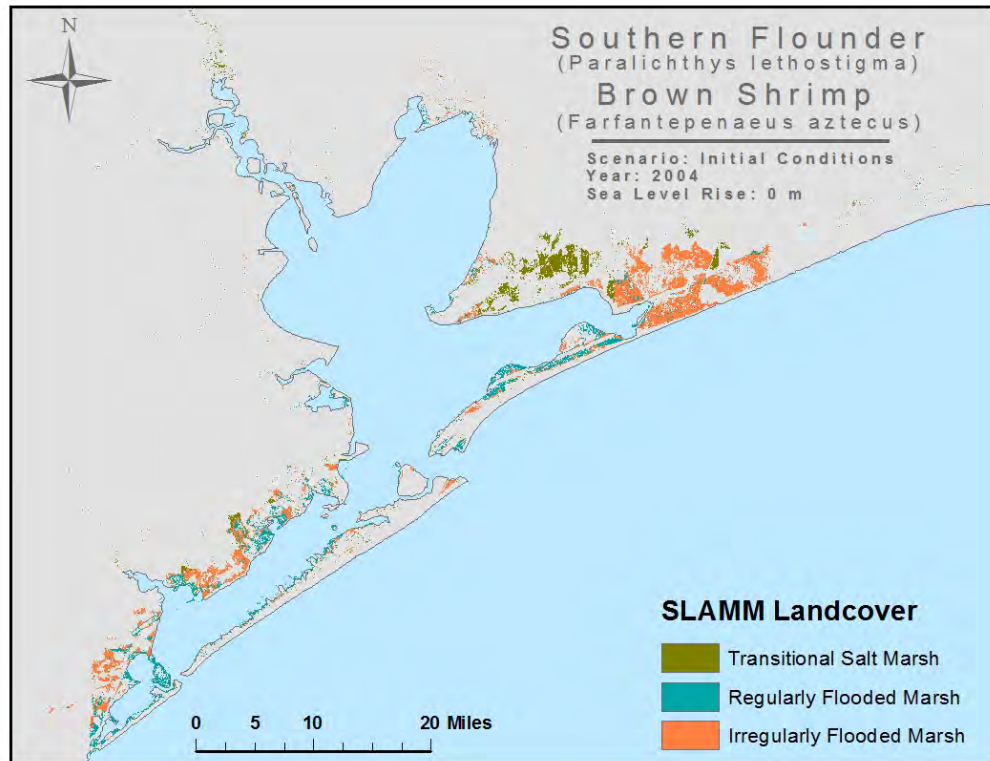Figure F-2: Habitat for juvenile blue crab in 2004 and 2100 under a 1 meter of SLR scenario.

Figure F-3: Habitat for juvenile southern flounder and brown shrimp in 2004 and 2100 under a 1 meter of SLR scenario.
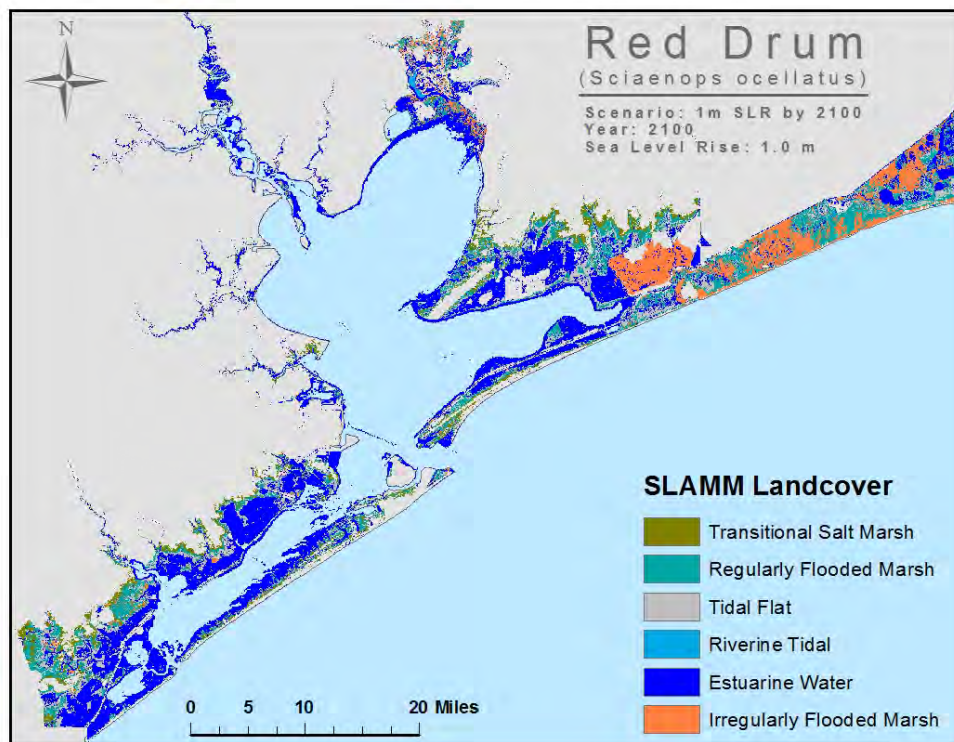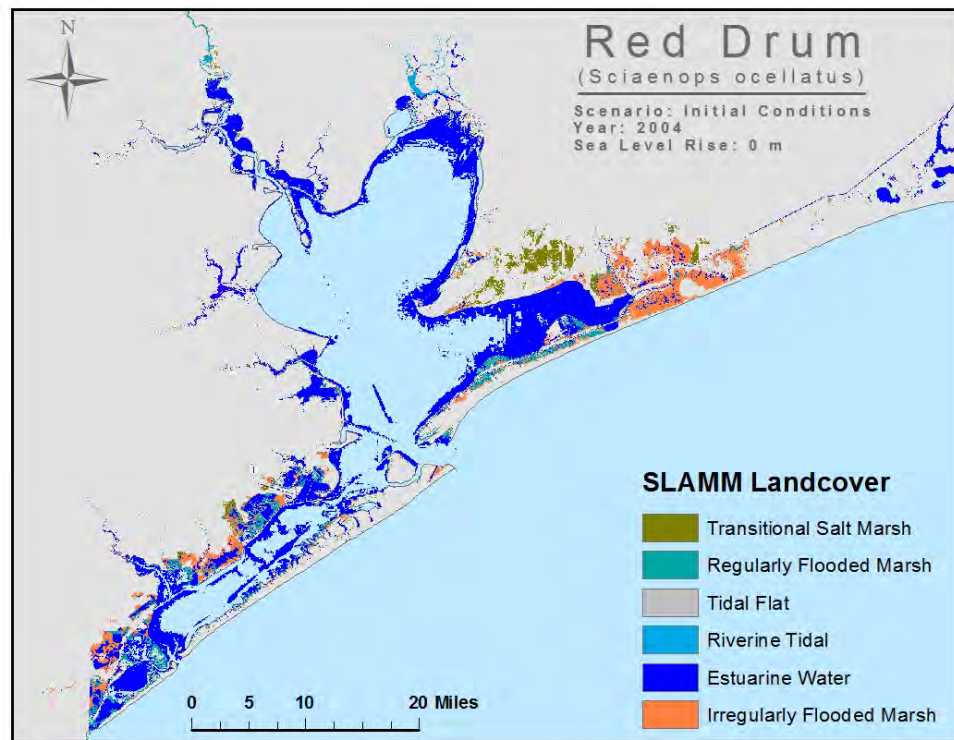
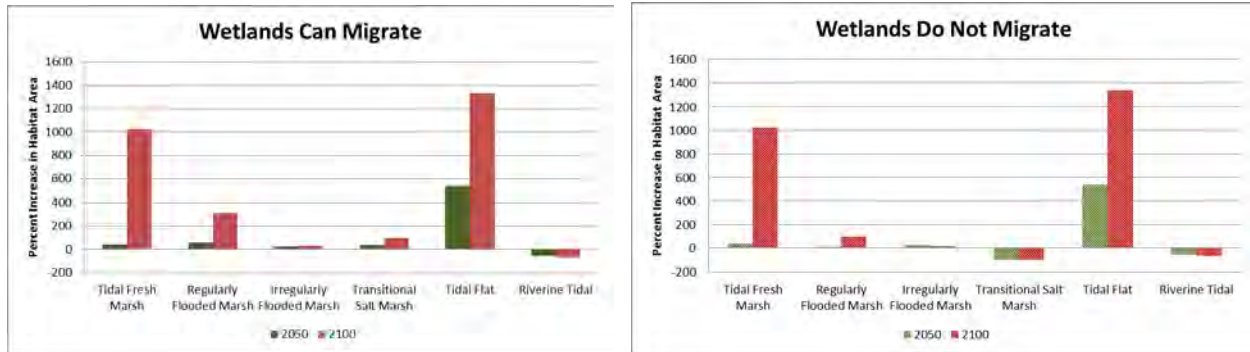Figure F-4: Habitat for juvenile red drum in 2004 and 2100 under a 1 meter of SLR scenario.

Figure F-5: Change in habitat area between 2004 and 2050 and 2004 and 2100 for a 1 m SLR scenario, assuming that wetlands can migrate (left) or cannot migrate (right).

Overall, all fisheries will experience net gains in habitats if wetlands are allowed to migrate. Blue crabs will experience a net increase of 13% in total habitat area by 2050 and 44% by 2100. More importantly, they will experience a 37% gain of wetland habitat, or "preferred habitat", by 2050 and a 117% gain by 2100 (Figure F-6). Southern flounder and brown shrimp, which depend mostly on wetlands, will also benefit from a 37% in habitat area by 2050 and 117% increase by 2100 (Figure F-7). Finally, red drums will benefit from a 16% increase in total habitat area and 37% increase in marsh habitat by 2050 and a 57% increase in total habitat and an 11% increase in marsh habitat by 2100 (Figure F-8). The gains in marsh habitat are greater than gains in total habitat for those fisheries because the amount of open water area decreases.
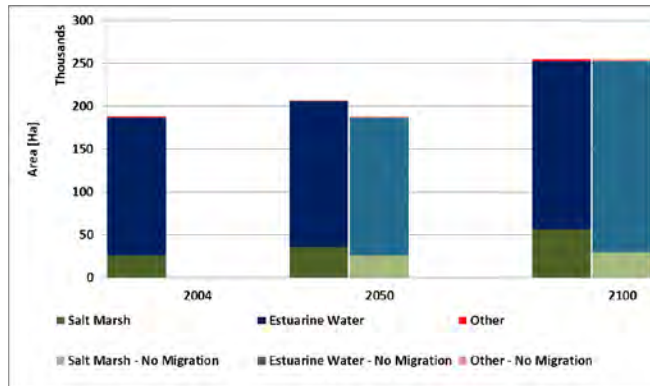
Figure F-6: Gains in habitat by type for juvenile blue crab in Galveston Bay, Texas under a 1 meter of SLR scenario.
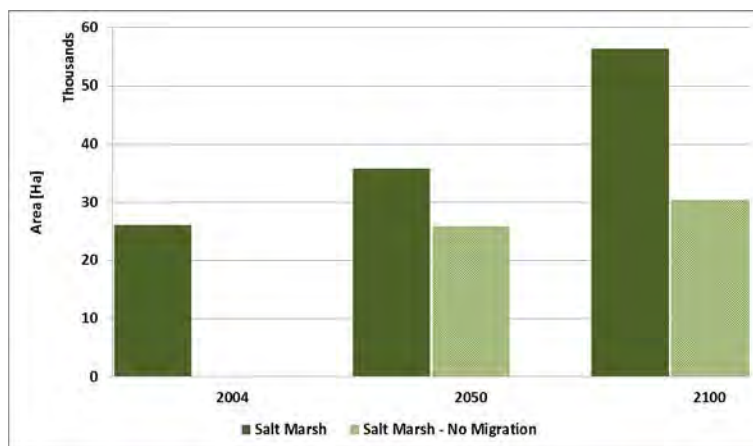

Figure F-7: Gains and Losses in habitat by type for juvenile southern flounder and brown shrimp in Galveston Bay, Texas under a 1 meter of SLR scenario.
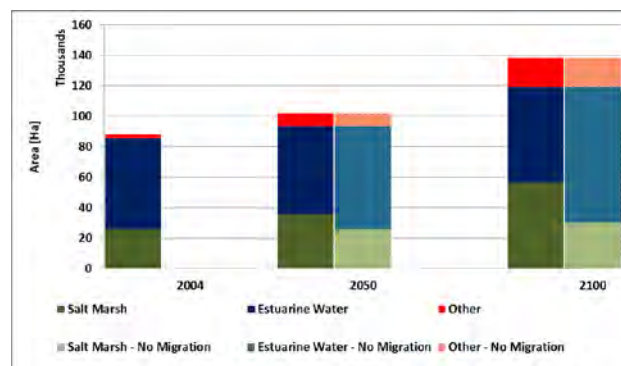

Figure F-8: Gains and Losses in habitat by type for juvenile red drum in Galveston Bay, Texas under a 1 meter of SLR scenario.

If wetlands are not allowed to migrate, the key species we examined will still experience the same net gains in total habitat area, but those gains will mostly come from additional gains in estuarine and open water areas because they will experience losses or little gain in marsh areas. In particular, blue crabs will experience a 17% decrease in wetland habitat by 2050 and a modest 17% gain by 2100 (Figure F-6). Southern flounder, brown shrimp and red drum will see almost no change in marsh habitat by 2050 (-1%) and a modest 17% gain by 2100 (Figure F-7 and Figure F-8).

Consequently, most fisheries are likely to experience positive gains in habitat as sea level rises. More importantly, they are likely to experience net gains in wetland areas by 2050 and even greater gains by 2100, if marshes are allowed to migrate. If marshes are not allowed to migrate, fisheries will mostly benefit from even wider open areas. They will mostly experience little positive change or small losses in the amount of marsh habitat that they can utilize.

Appendix G Marsh Habitat Captured in the Fishery Habitat Model

In the Fishery Habitat model, we determine the change in salt marsh habitats area (regularly flooded, irregularly flooded and transitional) with sea level rise. To do so, we capture the total area of each of the three salt marsh categories withindifferent elevation ranges. Since marsh habitat is the preferred habitat for most of the important fisheries in the Bay (see Section 3.2.5), the ability of those masks to capture salt marsh habitat is important.

We created two masks, for the different species studied herein. For Blue Crab, we created a mask, referred to as the *max* mask, that extends from -36 m to MHW. For all other species, the mask, referred to as the *min* mask, extends from -2 m to MHW. In this section, we compare the relative effectiveness of both masks.

First, comparison of the area of the two different masks (Figure G-1) shows that the -36 mask (*max* mask) is more than five times larger than the -2 m mask (*min* mask). The area of the -36 m mask increases under each time step, for all SLR scenarios. While the -2 meter DEM mask area also increases under each time step, overall, it decreases in size by 13% under the two meter SLR scenario during the final period between 2050 and 2100 This indicates that this mask best captures the rapid SLR experienced under the 2 m scenario, with the nearshore areas becomes deeper as low lying, flatter areas become inundated.

Area Comparison Between -36 meter (max) and -2 meter (min) to MHW DEM Masks

Legend:
- 2050 MAX
- 2100 MAX
- 2050 MIN
- 2100 MIN
- 2004 MAX
- 2004 MIN

Y-axis: HECTARES (X1000)
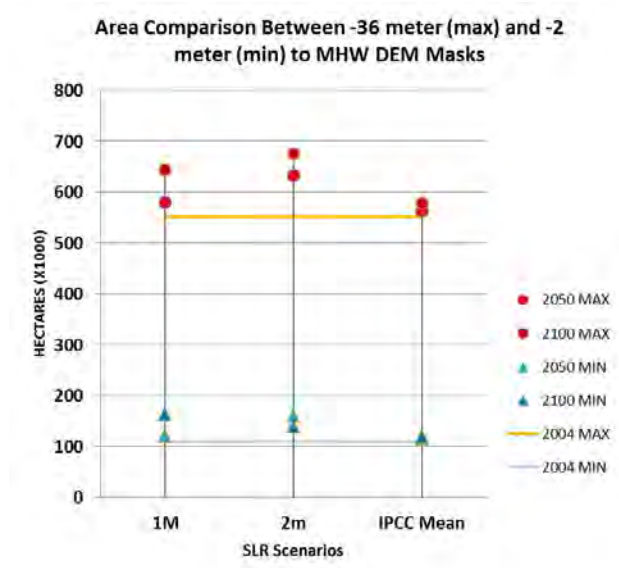X-axis: SLR Scenarios (1M, 2m, IPCC Mean)

Figure G-1: Area comparison between the -36 meter (max) and the -2 meter (min) DEM masks under 1 meter, 2 meter and IPCC mean SLR scenarios

The spatial extent of salt marsh habitat that is captured in the DEM mask under the 1 m SLR scenario are shown for the initial conditions, 2050 and 2100 (Figure G-3, Figure G-2, and Figure G-4). We find that only a portion of the marsh habitat falls within the footprints of the DEM mask for the initial time period (Figure G-3). However, for the 2050 and 2100 time horizons, the proportion of salt marsh habitat that lies outside the mask decreases.
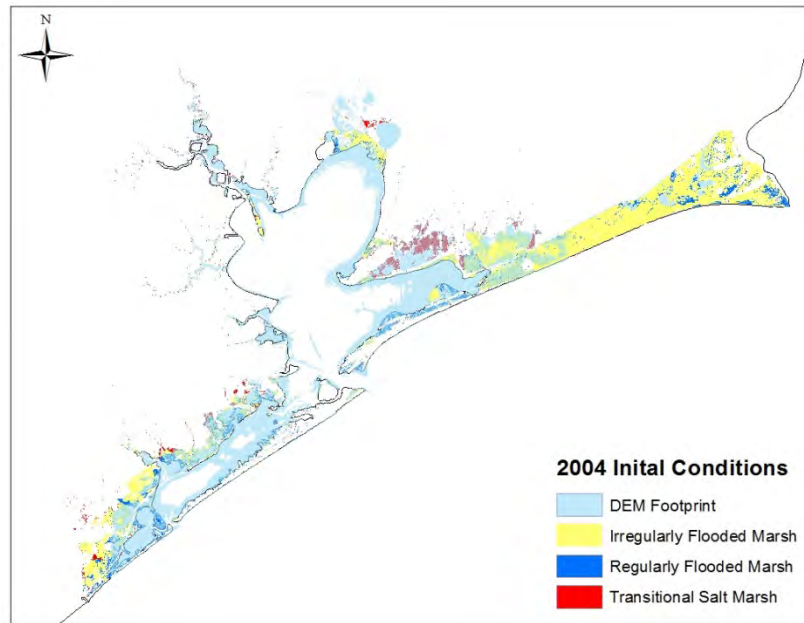
Figure G-2: Salt marsh habitats in 2004 that fall inside and outside of the -2 meter to MHW DEM spatial extent.
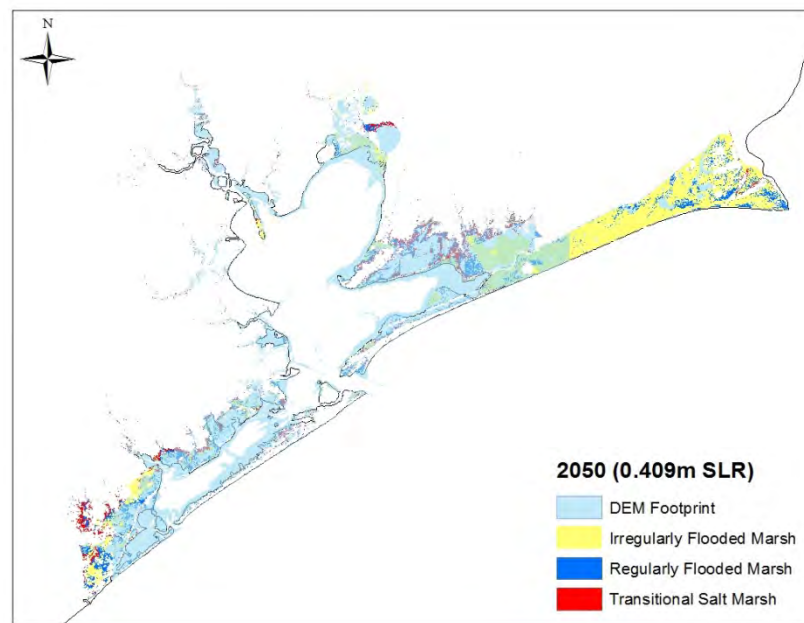

Figure G-3: Salt marsh habitats in 2050 that fall inside and outside of the -2 meter to MHW DEM spatial extent under a 1 meter of SLR by 2100 scenario.
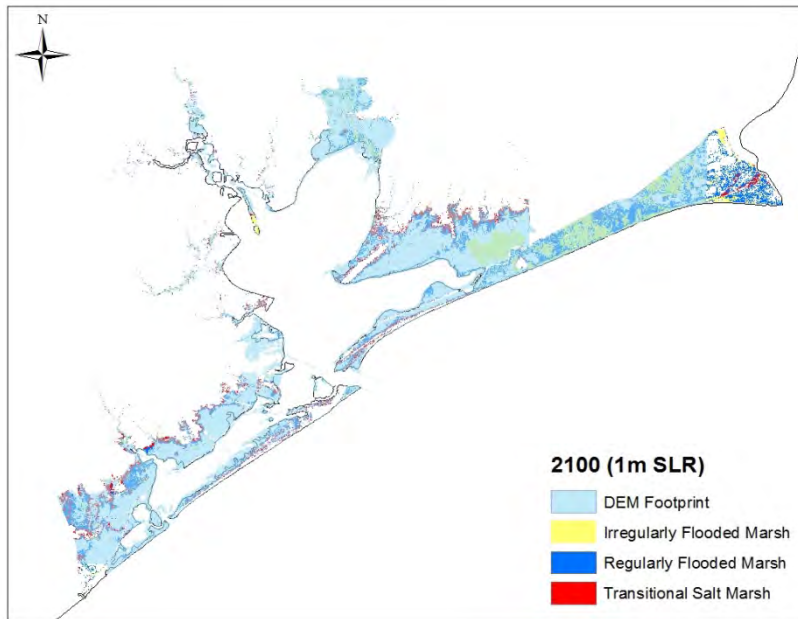
Figure G-4: Salt marsh habitats in 2100 that fall inside and outside of the -2 meter to MHW DEM spatial extent under a 1 meter of SLR by 2100 scenario.

The full results for the DEM marsh habitat analysis are shown in Figure G-5. Overall, under the initial 2004 conditions, less than 36% of the total area of the three SLAMM salt marsh landcover types fall within the DEM mask. Irregularly flooded marsh, which occupies the highest elevations of the three marsh categories, has 26% of its total area within the bounds of the DEM mask, but its initial area is also much larger than the other salt marsh types. Fifty-five percent (55%) of the regularly flooded marsh, which lies lower in elevation, is captured, while 71% of transitional salt marsh area is captured.

As sea level rises, more of the marsh habitat is captured by the mask. For the 2m SLR scenario, over 91% of all salt marsh area in Galveston Bay is covered by the mask in 2100. For the 1 m SLR scenario, this number is reduced to 83%. If we assume that marshes are not allowed to migrate (no new marsh created as sea level rises), the final percentages of salt marsh covered by the DEM mask are very similar to the marsh migration results with 87%, 82% and 41% under

the 2 m, 1 m and IPCC mean SLR scenarios, respectively. Finally, comparison of salt marsh area covered with the -36 m and -2 m mask shows that under the 1 m and IPCC mean SLR scenarios, the percent of marsh that falls within the two different DEM masks are the same. Under the 2 m SLR scenarios the -2 meter DEM mask contained 88% (migration) and 41% (no migration) of the total salt marsh area compared to 91% and 45% under the -36 meter mask.

In conclusion, although both masks do not capture all the habitats present in the Bay, they are effective in capturing a large majority of the lower salt marsh boundaries found within the study area.
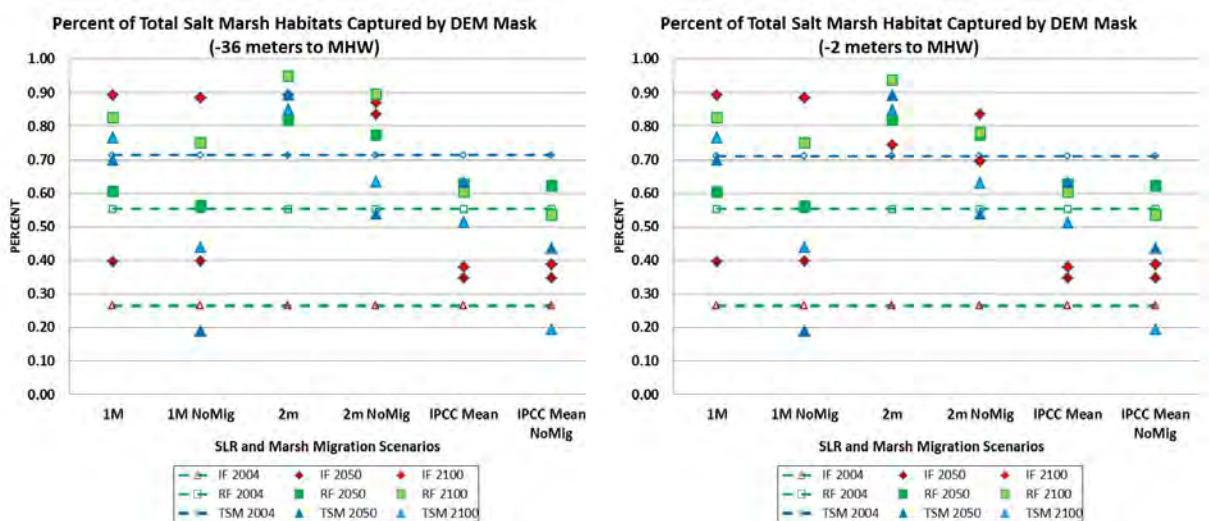

Figure G-5: Salt marsh habitats that fall within the DEM masks for the -36 meter to MHW (A) and the -2 meters to MHW (B) depth ranges under six SLR and marsh migration scenarios for regularly flooded (RF), irregularly flooded (IF) and transitional salt marsh (TSM).