Journal of Coastal Research	SI	63	211–233	Coconut Creek, Florida	Spring 2013

Potential Effects of Sea-Level Rise on Coastal Wetlands in Southeastern Louisiana

Patty Glick†, Jonathan Clough‡, Amy Polaczyk‡, Brady Couvillion§, and Brad Nunley††

†National Wildlife Federation
2100 Westlake Avenue North
Suite 107
Seattle, WA 98109, U.S.A.
glick@nwf.org

‡Warren Pinnacle Consulting
P.O. Box 315
Waitsfield, VT 05673, U.S.A.

§U.S. Geological Survey
National Wetlands Research Ctr.
c/o Livestock Show Office
Louisiana State University
Baton Rouge, LA 70803, U.S.A.

††11904 East Johnson Rd.
Ashland, KY 41102, U.S.A.

ABSTRACT

Glick, P.; Clough, J.; Polaczyk, A.; Couvillion, B., and Nunley, B., 2013. Potential effects of sea-level rise on coastal wetlands in southeastern Louisiana. *In:* Brock, J.C.; Barras, J.A., and Williams, S.J. (eds.), *Understanding and Predicting Change in the Coastal Ecosystems of the Northern Gulf of Mexico,* Journal of Coastal Research, Special Issue No. 63, pp. 211–233, Coconut Creek (Florida), ISSN 0749-0208.

Coastal Louisiana wetlands contain about 37% of the estuarine herbaceous marshes in the conterminous United States. The long-term stability of coastal wetlands is often a function of a wetland's ability to maintain elevation equilibrium with mean sea level through processes such as primary production and sediment accretion. However, Louisiana has sustained more coastal wetland loss than all other states in the continental United States combined due to a combination of natural and anthropogenic factors, including sea-level rise. This study investigates the potential impact of current and accelerating sea-level rise rates on key coastal wetland habitats in southeastern Louisiana using the Sea Level Affecting Marshes Model (SLAMM). Model calibration was conducted using a 1956–2007 observation period and hindcasting results predicted 35% versus observed 39% total marsh loss. Multiple sea-level-rise scenarios were then simulated for the period of 2007–2100. Results indicate a range of potential wetland losses by 2100, from an additional 2,188.97 km^2 (218,897 ha, 9% of the 2007 wetland area) under the lowest sea-level-rise scenario (0.34 m), to a potential loss of 5,875.27 km^2 (587,527 ha, 24% of the 2007 wetland area) in the highest sea-level-rise scenario (1.9 m). Model results suggest that one area of particular concern is the potential vulnerability of the region's baldcypress-water tupelo (*Taxodium distichum-Nyssa aquatica*) swamp habitat, much of which is projected to become permanently flooded (affecting regeneration) under all modeled scenarios for sea-level rise. These findings will aid in the development of ecosystem management plans that support the processes and conditions that result in sustainable coastal ecosystems.

ADDITIONAL INDEX WORDS: *Marsh accretion, river diversion, sediments, SLAMM, subsidence.*

INTRODUCTION

For generations, coastal Louisiana's marshes and swamps have been a linchpin for the regional economy, culture, and quality of life (Engle, 2011). These systems support the largest commercial fishery in the lower 48 states (NOAA, 2010). They are habitat for millions of wintering waterfowl and many other species of fish and wildlife, provide coastal communities with a critical line of defense against tropical storms and hurricanes (Costanza *et al.*, 2008; Feagin *et al.*, 2010; Gedan *et al.*, 2010), and support five of the nation's largest ports and over 20% of the nation's foreign waterborne commerce (Twilley, 2007). Yet, there has long been recognition that Louisiana's coastal wetlands are in peril. Today, Louisiana's wetlands represent about 37% of the estuarine herbaceous marshes of the conterminous United States, yet sustain more coastal wetland loss than all other states in the conterminous United States combined (Louisiana Department of Natural Resources, 2011). Recent analysis has determined that the region has lost more than 4,876 km^2 of land area between 1932 and 2010 due to a multitude of natural and anthropogenic factors, including: altered hydrology due to river levees and upstream dams; canal dredging; oil and gas extraction; invasive species; geological subsidence; storms; and sea-level rise (Couvillion *et al.*, 2011; Yuill, Lavoie, and Reed, 2009).

In addition, considerable marsh losses occurred in parts of the region during recent hurricane events, including Hurricanes Katrina and Rita in 2005 (Barras, 2007) and Gustav and Ike in 2008 (Barras *et al.*, 2010). While it is difficult to determine which of these losses are likely to be permanent and which are transitory, evidence of cumulative, long-term storm impacts across the Louisiana coast during the past century suggests that cumulative losses associated with these hurricanes will remain significant (Barras, 2009; Morton and Barras, 2011). Paradoxically, engineering in response to extreme events such as floods and coastal storms are making some coastal habitats even more vulnerable to their impacts as structures such as dikes and levees interfere with the processes that are necessary to sustain the function and extent of wetland ecosystems and reduce their natural capacity to mitigate flooding.

Research indicates that the overall rate of wetland loss in the region has slowed somewhat in recent decades (Barras, Bernier,

DOI: 10.2112/SI63-0017.1 received 1 November 2011; accepted 29 June 2012.

and Morton, 2008). Nevertheless, significant net losses are expected to continue into the future (Couvillion *et al.*, 2011). If the trend of land loss that has occurred between 1985 and 2010 (-42.92 km^2 per year) were to persist at a constant rate, the region would lose wetlands by an area the size of one football field per hour (Couvillion *et al.*, 2011).

The realized and potential consequences of these losses have galvanized a tremendous regional and national effort to restore and protect Louisiana's coastal wetland systems. Today we also face the extraordinary challenges brought on by climate change, including sea-level rise, more extreme precipitation events, and an increase in the intensity of tropical storms and hurricanes (Karl, Melillo, and Peterson, 2009). In particular, an accelerating rate of sea-level rise is a considerable added stressor that has only recently started to be factored into projected land loss trends.

Given this situation, state and federal agencies, nongovernmental organizations and others concerned with restoring and protecting coastal Louisiana are faced with designing and implementing projects that will maximize the effectiveness of our investments under current and expected future climate conditions. While the ecological systems associated with Louisiana's coastal wetlands are dynamic and complex, resource managers and planners need information about the existing and potential ecosystem and landscape changes to assist them in designing and implementing meaningful projects. As Twilley (2007) acknowledges, to assume a continuation of present conditions for relative sea-level change when evaluating proposed restoration efforts to sustain Louisiana's coastal wetlands over the long term likely overstates the potential effectiveness of such efforts given recent studies projecting accelerating sea-level rise as well as the intensification of hurricanes due to climate change.

ACCELERATING SEA-LEVEL RISE: A MATTER OF DEGREES

Rising sea levels have been a concern for coastal Louisiana for decades, as decreases in land elevation due to subsidence have combined with a rise in eustatic sea level due to the thermal expansion of ocean waters and the addition of water from melting glaciers and ice fields. As mentioned above, Louisiana will continue to lose land in the coming years even if the rate of relative sea-level rise were to stay constant into the future (Couvillion *et al.*, 2011). However, scientists now recognize that climate change is likely to cause eustatic sea levels to rise at an accelerating pace (*e.g.*, Cazenave and Llovel, 2010; Chen, Wilson, and Tapley, 2006; Rahmstorf, 2010; Rignot *et al.*, 2011).

Projections for future sea-level rise vary among different studies due to a host of factors, such as the choice of scenarios for global temperatures and the respective treatment of uncertainties such as ice flow dynamics. The most recent projections from the Fourth Assessment Report (AR4) of the Intergovernmental Panel on Climate Change (IPCC) suggest a possible range of increase of 18–59 cm from 1990 to the mid-2090s, with an additional rise of 10–20 cm or more possible when taking into consideration the observed 1993–2003 rate of ice sheet melt from Greenland and Antarctica (IPCC, 2007). There is compelling new evidence, however, that because these figures ignored the recent dynamic changes in Greenland and Antarctica ice sheets, it is likely that they significantly underestimate the rate of global sea-level rise that society will experience in the coming decades (Jevrejeva, Moore, and Grinsted, 2010; Otto-Bliesner *et al.*, 2006; Overpeck *et al.*, 2006; Overpeck and Weiss, 2009; Rahmstorf, 2007; Rignot and Kanagaratnam, 2006; Vermeer and Rahmstorf, 2009).

Rahmstorf (2007) proposes that, taking into account possible model error, a feasible range by 2100 is a rise of 50–140 cm. More recently, this work was updated and the ranges were increased to 75–190 cm (Vermeer and Rahmstorf, 2009). Importantly, Pfeffer, Harper, and O'Neel (2008) suggest that 2 m by 2100 is likely at the upper end of plausible scenarios over the period due to physical limitations on glaciological conditions.

Sea-Level Rise and Coastal Wetlands

One of the primary factors influencing the vulnerability of coastal wetlands to relative sea-level rise is tidal range (Kirwan and Guntenspergen, 2010). In general, the composition of coastal wetland vegetation is a factor of the duration and depth of tides (Martin *et al.*, 2009; Mitsch and Gosselink, 2000). Many coastal plant species are adapted to a certain level of salinity and tidal influence, so prolonged changes can make habitats more favorable for some species, less for others (Callaway *et al.*, 2007). For example, salt marshes are generally assumed to persist from mean tide level (MTL) up to an elevation greater than mean higher high water. Empirical studies support this relationship, although there are occasional site-specific differences. Sea-level rise can also contribute to the expansion of open water inland, where dry land can become saturated by an increase in the height of the water table.

Coastal wetlands may, at least to some extent, be able to accommodate moderate changes in sea level by migrating inland or increasing in elevation due to accretion (Morris *et al.*, 2002). Factors that contribute to marsh accretion are complex and dynamic, and they can vary considerably by wetland type, tidal range, and other variables. In some river deltas, for example, the deposition of sediments from upstream or upland sources can provide sufficient levels of soil for marshes to maintain elevation relative to sea level (Day *et al.*, 2000). Sediments can also be deposited via tidal currents and storms (Cahoon, 2006). In addition, marshes can build up their own organic matter through growth and decomposition of roots and leaves (Nyman *et al.*, 2006).

Research suggests significant feedbacks among coastal marsh productivity, organic matter accumulation, rates of sedimentation, and maintenance of elevation (Callaway *et al.*, 2007; Kirwan *et al.*, 2010; Nyman *et al.*, 2006). While these relationships may combine to allow coastal wetlands to adapt to a certain amount of sea-level rise in the future, studies suggest that there is likely a threshold elevation that will lead to greater plant stress and a reduction in organic matter accumulation and vertical accretion (DeLaune, Nyman, and Patrick, 1994; FitzGerald *et al.*, 2008; Kirwan *et al.*, 2010; Morris, 2006). This threshold appears to have already been reached for many of coastal Louisiana's wetlands, due to a number of anthropogenic and natural factors (Boesch, 2006; Day *et al.*, 2000, 2011b; DeLaune *et al.*, 2003; Hatton, DeLaune, and Patrick, 1983; Morton *et al.*, 2005). There are, however, some notable exceptions. Specifically, sediment inputs from river flows (avulsions) into the Atchafalaya and Wax Lake Deltas are contributing to land accumulation along the coast (Roberts *et al.*, 2003). These sediment inputs are the result of an engineered avulsion of the Mississippi, whereby structures were built in 1964 to divert water into the Atchafalaya River for sediment and flood control (Kim *et al.*, 2009). The Atchafalaya River itself partially avulsed into the man-made Wax Lake Outlet during a major flood event in 1973, and since then the Atchafalaya and Wax Lake Deltas have been actively building land even under current rates of subsidence and eustatic sea-level rise (Allen, Couvillion, and Barras, 2011). These examples have bolstered interest in developing diversions in other parts of the Mississippi Delta region as a way to restore land-building processes for coastal wetland restoration. While key questions remain, such as whether the resulting sediments would be sufficient to offset accelerating sea-level rise, the potential for controlled avulsions through river diversions is a strategy that shows some promise in coastal restoration (Boustany, 2010; Day *et al.*, 2009; Kim *et al.*, 2009).

Modeling the Coast

As the National Research Council (NRC) suggests in its 2006 report *Drawing Louisiana's New Map*, the region will need to find a balance between what is considered desirable and what is attainable. Indeed, the "new map" (NRC, 2006) for Louisiana's coastline will forever be a moving target, and our vision for the future will need to be less of a snapshot and more of a motion picture. Within this context, models have been and will continue to be important tools to help managers better understand how coastal Louisiana's wetland systems work and how they might respond to the numerous stressors they face in the coming decades, including sea-level rise.

Recent planning efforts led by the U.S. Army Corps of Engineers (USACE) for coastal Louisiana restoration [*e.g.*, the draft Louisiana Coastal Area, Louisiana–Ecosystem Restoration: Comprehensive Coastwide Ecosystem Restoration Study (USACE, 2003) and the final Louisiana Coastal Area (LCA), Louisiana–Ecosystem Restoration Study (USACE, 2004)] have relied on relatively sophisticated ecological simulation models and coarser scale desktop models to assist in decisionmaking (NRC, 2006). In addition, the Coastal Louisiana Ecosystem Assessment and Restoration (CLEAR) Program used a variety of modeling tools within a framework to develop an ecosystem forecasting system to evaluate potential restoration alternatives (Twilley *et al.*, 2008).

A number of models are available to assess the potential impacts of sea-level rise on coastal habitats and communities (Mcleod *et al.*, 2010). These models range considerably in terms of their complexity, and all have certain strengths and weaknesses. Some of these models have been applied in coastal Louisiana and other areas of the Gulf of Mexico (*e.g.*, Blum and Roberts, 2009; Doyle *et al.*, 2010; Martin *et al.*, 2002; Reyes *et al.*, 2000; Ritchie, Gawthorpe, and Hardy, 2004; Shirley and Battaglia, 2008; Smith *et al.*, 2010; Weiss, Overpeck, and Strauss, 2011).

This study investigates the potential impact of accelerating sea-level rise on key coastal wetland habitats in southeastern Louisiana using the Sea Level Affecting Marshes Model (SLAMM). The study was undertaken to provide another tool to help managers understand the risks at hand and identify strategies to minimize those risks and restore an ecologically functional wetland system to the region.

It is important to recognize that the SLAMM model is neither the simplest nor the most complex tool available for simulating the impacts of sea-level rise on coastal zones. As Mcleod *et al.* (2010) acknowledge, there are significant tradeoffs among different sea-level rise impact models due to multiple factors, such as the cost to acquire the model or commission experts to apply it, the amount of computer capacity required to run the model or interpret results, and the ability of the model to cover relevant impacts at spatial scales desirable for decisionmakers.

SLAMM accounts for the dominant processes involved in wetland conversion and shoreline modifications during long-term sea-level rise (Park *et al.*, 1989). The model provides greater detail than static coastal topography provides alone. For example, it can be used to assess the extent to which seawater inundation contributes to the conversion of one habitat type to another based on elevation, habitat type, slope, sedimentation, accretion, erosion rates, and the presence of dikes. In addition, SLAMM accounts for relative sea-level change, calculated as the sum of the historical eustatic (global average) trend, the site-specific rate of change of coastal elevation due to subsidence and other factors, and the accelerated rise in sea level (which is dependent on the future scenario chosen). Sea-level rise is also offset by sedimentation and accretion, again using site-specific values.

Successive versions of SLAMM have been used to estimate the impacts of sea-level rise along the coasts of the United States (*e.g.*, Craft *et al.*, 2009; Galbraith *et al.*, 2002; Geselbracht *et al.*, 2011; Glick and Clough 2006; Glick, Clough, and Nunley, 2007, 2008; Lee, Park, and Mausel, 1992; Park, Lee, and Canning, 1993; Titus *et al.*, 1991; Traill *et al.*, 2011). SLAMM is useful for a range of landscape scales (<1 km^2 to 100,000 km^2) at high resolution (cell sizes as low as 5 m x 5 m), and it enables managers to identify potential changes in both extent and composition of different wetland types.

The latest version of SLAMM (Version 6) also offers the ability to capture important feedback mechanisms between sea-level change and marsh accretion (Clough, Park, and Fuller, 2010). However, the model does not incorporate some of the other complex factors that will affect the way the region's marsh systems may respond to sea-level rise, such as localized geomorphology, hydrodynamics, and salinity tolerance (Burkett *et al.*, 2005); nor does it address other impacts associated with climate change, including altered river flows, higher average temperatures, and more intense hurricanes.

All model results are subject to uncertainty due to limitations in input data, incomplete knowledge about factors that control the behavior of the systems being modeled, and simplifications of the systems (Pascual, Steiber, and Sunderland, 2003). Being transparent about the sources and extent of uncertainty (quantitatively and/or qualitatively) in ecological assessments will facilitate conservation and management decisionmaking. However, one must be careful not to allow uncertainty to preclude consideration of climate impacts on species and habitats.

METHODS

Study Area

SLAMM 6 was applied to approximately 31,000 km^2 (3.1 million ha) of southeastern Louisiana (Figure 1), using 15-m by 15-m cells. The study area was split vertically into 2 even pieces (to the east and west) to bypass memory and computational limitations.

Model Summary

SLAMM Version 6.0 was released in 2009 and is based on and backward compatible with SLAMM 5. Within SLAMM, there are five primary processes that affect wetland fate under different scenarios of sea-level rise: inundation, erosion, overwash, saturation, and accretion. Increased inundation and the corresponding rise of the salt boundary (the elevation of the mean high water spring that delineates salt and fresh marshes) are tracked by reducing elevations of each cell as sea level rises, thus keeping MTL constant at zero. The effects of inundation on each cell are calculated based on the minimum elevation and slope of that cell. Erosion is triggered based on a threshold of maximum fetch (the distance a wave travels across water) and the proximity of marsh to estuarine water or open ocean. When the required conditions are met, horizontal erosion occurs. To account for the effects of large storms, a barrier island's width can be assumed to undergo overwash during each specified time interval. During overwash, beach migration and transport of sediments are calculated (Clough, Park, and Fuller, 2010).

Vertical accretion is considered in SLAMM simulations by using average or site-specific values for each wetland category. Accretion rates may be spatially variable within a given model domain or can be specified to respond to feedbacks based on frequency of flooding/wetland elevation, distance to channel, and salinity. Accretion feedbacks based on wetland elevation relative to the tide range were used in the simulations for southeastern Louisiana.

SLAMM 6 will summarize site-specific categorized elevation ranges for wetlands as derived from light detection and ranging (lidar) data or other high-resolution datasets. This functionality is used to test the SLAMM conceptual model at each study site. If site-specific data indicate that wetland elevation ranges are outside of SLAMM defaults, a different range may be specified within the interface.

SLAMM allows for the assumption that all currently developed areas, agricultural zones, and other areas categorized as dry land will be protected by dikes, levees, or other coastal armoring, so these areas do not show inundation under the various scenarios for sea-level rise used in the model. While it was assumed in the simulations presented in this work that all dry land (both developed and undeveloped) would be protected

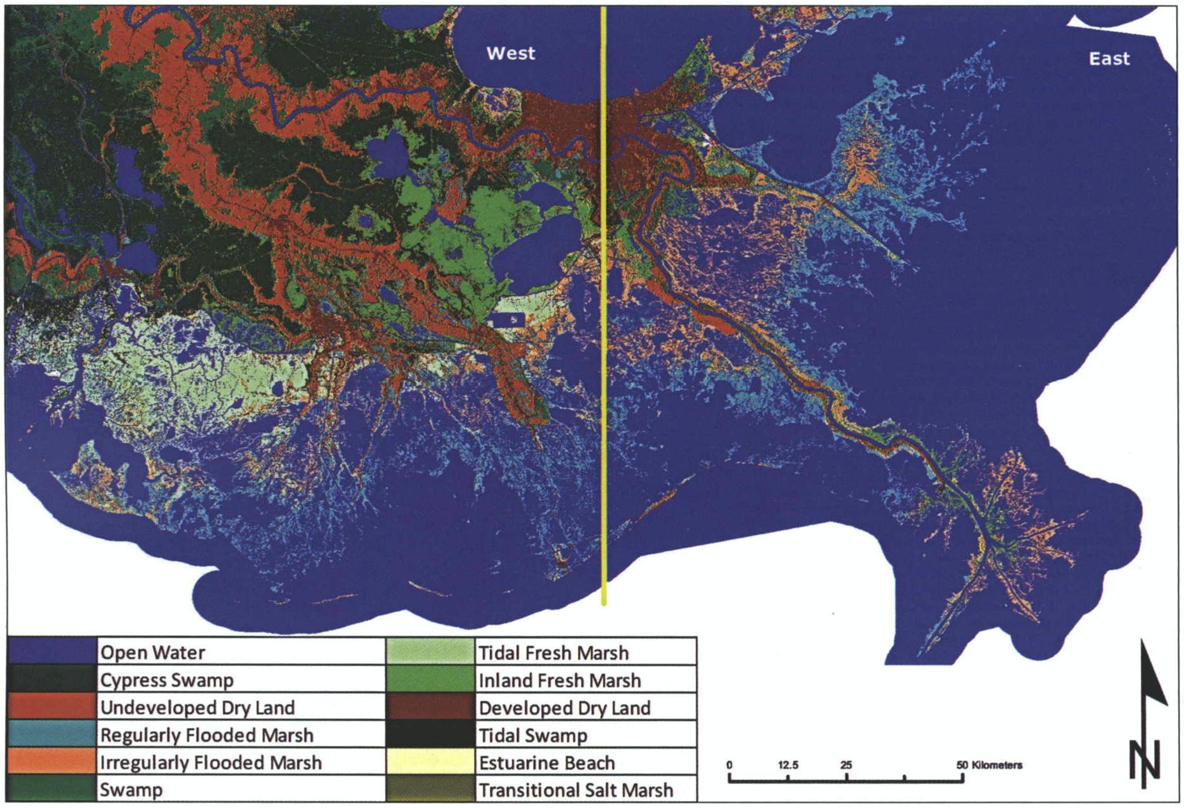

Figure 1. 2007 wetland coverage for geographic extent of study area based on Couvillion *et al.* (2011).

into the future, this does not mean that low-lying communities such as New Orleans are not also vulnerable to sea-level rise, only that the potential impacts are not captured here. While it is beyond the scope of this study to address impacts on the region's vulnerable developed areas, the potential for sea-level rise to cause significant and costly damage to property and infrastructure should not be ignored.

Sea-Level Rise Scenarios

Four eustatic sea-level rise scenarios were applied in this study. The first scenario represents a rapid climate stabilization (Stanton and Ackerman, 2007) case and assumes a linear sea-level rise rate of 3.1 mm/yr (or 0.34 m of sea-level rise by 2100 when starting projections in 1990). This rate of 3.1 mm/yr was based on historical trend data measured from 1993 to 2003, as reported by Grinsted, Moore, and Jevrejeva (2009). However, this is likely an overly optimistic scenario. As a number of recent studies have indicated, global emissions of greenhouse gases have been accelerating in recent years at a rate higher than the Intergovernmental Panel on Climate Change's (IPCC) most fossil-fuel intensive scenario (A1FI) (Betts *et al.*, 2011; Nakicenovic *et al.*, 2000; Raupach *et al.*, 2007). Even if emissions were to essentially halt immediately, the changes associated with the associated increase in the atmospheric concentration of carbon dioxide (CO_2) are expected to be largely irreversible for at least 1,000 years (Solomon *et al.*, 2009).

Barring a significant climate change policy breakthrough in the near future, the rapid climate stabilization scenario is unlikely. That said, this scenario was used to provide a baseline context against which to evaluate the added stressors associated with an accelerating rate of sea-level rise in coastal Louisiana's wetlands. The three remaining scenarios modeled the effects of 0.75, 1.22, and 1.90 m of eustatic sea-level rise by 2100. These represent the full range of possible sea-level rise modeled by Vermeer and Rahmstorf (2009). These scenarios did not assume the rate of sea-level rise to be linear, but to increase quasi-exponentially. Importantly, the purpose of this study is not to predict how much eustatic sea-level rise is possible or likely in the future. Rather, the choice of such scenarios is exogenous. The projections by Vermeer and Rahmstorf (2009) are within the range of several studies published since the IPCC AR4 (IPCC, 2007) (*e.g.*, Grinsted, Moore, and Jevrejeva, 2009; Jevrejeva, Moore, and Grinsted, 2010; Pfeffer, Harper, and O'Neel, 2008; Rahmstorf, 2007).

When the model was run to estimate past wetland changes (hindcasting), the global rate of sea-level rise from 1990 to present was estimated to be 3.0 mm/yr, slightly less than the figure used for the rapid stabilization scenario but reflecting recent evidence suggesting an accelerating rate of sea-level rise during the past several decades as compared to the longer-term trend over the 20th century (Cazenave and Llovel, 2010; Donoghue, 2011; Merrifield, Merrifield, and Mitchum, 2009). Prior to 1990, the global trend of 1.75 mm/yr was assumed in the hindcasting study, based on estimates by Church *et al.* (2001) and Church and White (2006). These global rates of sea-level rise were translated into estimated local rates of sea-level rise using the spatial subsidence data derived for the site as discussed below.

Prospective and Retrospective Analysis

This study included both prospective (forecasting) and retrospective (hindcasting) analyses. Hindcasting is performed by starting a simulation at the photo date of the oldest available wetlands data, running it through the present day, and comparing the output to present-day wetland cover data. The primary goals of hindcasting are to assess the predictive capacity of a model and potentially to improve model predictions through calibration. In the case of SLAMM, hindcasting is used to determine whether or not the model is correctly predicting the effect of the observed sea-level signal on the wetland types in a given study area.

Historical digital elevation model (DEM) data are usually not used since older technology generally produced low vertical resolution data. For this analysis, SLAMM utilized an elevation preprocessor to compensate for the lack of historical DEM data. The elevation preprocessor estimates elevation ranges as a function of tide ranges and estimated relationships between wetland types and tide ranges (Clough, Park, and Fuller, 2010). It is important to note, therefore, that the accuracy of hindcast results should be judged based on the fact that initial wetland elevations were roughly estimated rather than based on observed data.

Baseline Elevation Map

The elevation data used in this effort consisted of lidar data, where available, and estimated topography data where lidar was unavailable. The lidar data used for this investigation was collected on dates ranging from 2000–2003, though most data were collected during lidar surveys that were completed in 2003 (Cunningham, Gisclair, and Craig, 2002). The accuracy of elevation data is paramount to modeling efforts. Root-mean-square error (RMSE) is often used to estimate vertical accuracy and is defined as the square root of the average squared differences between lidar values and values from an independent, presumably higher accuracy source. In general, the lower the RMSE, the higher the accuracy of the dataset. The RMSE of the lidar data used in this analysis is 15–30 cm, which varied by land cover type (Cunningham, Gisclair, and Craig, 2002). Variable RMSE by land cover type was calculated for this dataset and in general, the RMSE of wetland areas was better (*i.e.*, lower) than that of upland, forested, and shrub/scrub areas, and were typically observed at the lower end of the published range (Watershed Concepts, 2009). Refer to Cunningham, Gisclair, and Craig (2002) for a coverage map of the circa 2003 lidar.

To facilitate the comparability of this dataset to the 2007 land cover data, areas that had undergone a change in land cover type between the date of acquisition of the lidar and 2007, as determined by Couvillion *et al.* (2011), and areas for which no lidar was available, used an estimated elevation value. To estimate elevation, this study applied a methodology that draws upon patterns observed in multiple dates of optical imagery. The concept at the foundation of this methodology relies upon hydrologic and biophysical characteristics represented in the optical imagery that are related to topography.

One of the primary characteristics upon which this methodology draws is the spatial variation in the wetting and drying cycles observed in the optical imagery. Coastal

Louisiana wetlands are generally considered to be a low-relief environment, meaning there is little variation in elevation values. The topographic variation that is present, however, has a substantial effect on many hydrologic parameters, including the distribution, duration, and frequency of inundation. The low-relief topography of coastal Louisiana actually intensifies the spatial and temporal variability in these parameters, as a relatively small change in the vertical dimension (*e.g.*, water level) can have large effects on inundation patterns in the horizontal dimension. It is the variability in these changes in the horizontal dimension that are detectable from optical imagery, upon which this methodology draws.

Other parameters that are also highly related to topography in many environments, including coastal Louisiana, relate to vegetation. Species composition, in particular, is often dictated by the conditions of the site. Many of these conditions are related to topography, and as such, the species composition of an area is often indicative of its elevation. While optical imagery does not directly sense elevation, it can contain within its bands information regarding species composition. It is this information that is used to estimate the elevation of an area.

A regression tree classifier was used to exploit patterns observed in the above-mentioned characteristics at sites of known elevation in order to assign an estimate at sites of unknown elevation (Figure 2). The regression tree classifier was chosen because these models can approximate complex, nonlinear relationships such as the relationships between inundation and elevation. For this effort, Cubist® software, developed by RuleQuest Research (2012), was used to construct the regression trees.

This methodology requires a dependent variable that consists of known data values for the parameter of interest. In this case, elevation data for areas in which lidar was available constituted the dependent variable. Secondly, the methodology requires multiple independent variables containing information related to the parameter of interest. For this effort, optical imagery constituted the primary independent variables. Specifically, this optical imagery consisted of every cloud-free date of Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM) available for the study area from 1984–2007. Multitemporal imagery was used to allow the classifier to detect temporal variation in wetting and drying patterns as well as intra-annual variation in vegetation. The exact dates and number of images varies across the study domain. More information regarding cloud-free dates of Landsat imagery is available from the U.S. Geological Survey (USGS; USGS, 2012a). A secondary independent variable used in this effort consisted of aerial imagery collected in 2008 (Photo Science, Inc., 2009).

Independent variables were first subset to a land mask developed for 2007 to exclude elevation estimation in open-water areas. A 5% random sample of the dependent variable (elevations for areas in which lidar did exist) was taken for each basin in which elevations were to be estimated. The Cubist® software automatically recognizes patterns in the dependent and independent variables at these sample sites and produces rule-based models for parameter estimation at sites for which the value of the parameter of interest, in this case elevation, is unknown. These rule-based models (regression trees) were then applied to obtain estimated elevations for sites at which no elevation data was available.

Following this effort, lidar data became available for a majority of the areas in which topography was estimated for this study, thereby enabling comparison of the estimated elevation values with those from the newly available lidar (Woolpert, 2011). These areas were flown between January and April of 2011, and the vertical accuracy RMSE of this lidar is listed as 7.0 cm. As this data represents an improvement in RMSE as compared to that of the circa 2003 lidar, it was chosen for comparisons to the estimated lidar. In areas of common coverage, a difference was calculated by subtracting the 2011 lidar from the estimated topography, and a mean difference of +7.3 cm (standard deviation: 16.7 cm) was observed. Though

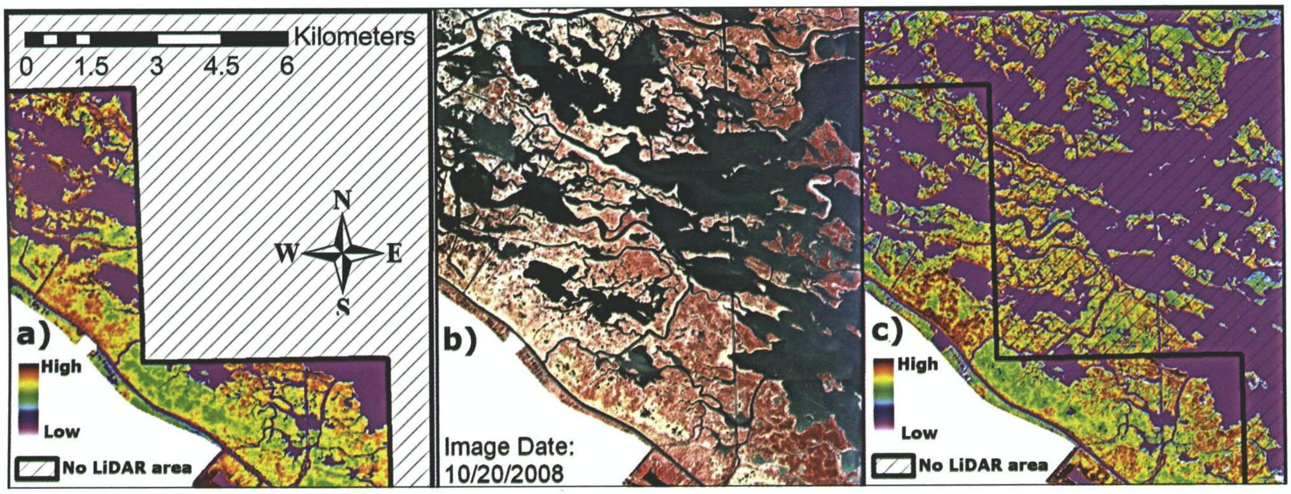

Figure 2. (A) Example of existing lidar used as training data; (B) Example of the imagery used to observe patterns in vegetation characteristics as well as wetting and drying cycles; and (C) The preliminary classified topography created based on the patterns observed in the training data, which are then applied to areas for which no data exists to create a pseudotopography dataset.

these differences are substantial, it is important to compare these values to other sources of elevation data. Comparing the circa 2003 and 2011 lidar datasets, a mean difference of +19.9 cm (standard deviation=84.8 cm) was observed. Finally, the National Elevation Dataset (NED), one of the only other sources of elevation available at the time of this study, was also compared to the 2011 lidar. A mean difference of +13.5 cm (standard deviation=42.4 cm) was observed between these two datasets. The overestimate of the previously available elevation sources as compared to the 2011 lidar is thought to result, in part, from increased point density, which would lead to an increased likelihood of penetrating obstructions and vegetation and obtaining accurate bare earth elevations.

Elevation accuracy remains a concern with models of coastal inundation and change and this effort is no exception. The estimated elevations used in portions of the study area served as an improvement over the best topography data available at the time of study implementation; however, interpretation of the results of this effort should proceed only with an understanding of the accuracy of each dataset and the potential implications of inaccuracy. It is important to note that the potential overestimate of various elevation data used for this analysis as compared to the higher accuracy 2011 data would likely result in a conservative estimate of potential wetland changes related to inundation. Furthermore, while the lowest eustatic SLR scenario run was 34 cm by 2100, the addition of estimated subsidence rates results in a local SLR estimate that is a minimum of 70 cm by 2100. These elevation datasets were deemed accurate enough to estimate the effects of changes of that magnitude. As elevation accuracy continues to improve, so too will coastal change models and our understanding of potential wetland change.

Land Cover Data

This analysis used a 2007 wetlands land cover dataset. This vegetation cover dataset was created using coastwide vegetation survey data from 2007 (Sasser *et al.*, 2008) as training data to classify marsh and shrub communities in the Louisiana coastal zone. This survey recorded species composition at 3,891 marsh and shrub stations throughout coastal Louisiana, which were then assigned to vegetation types using common dominants as well as species assemblages known to occur in the area. These data points were then used as training data for a remotely sensed land cover classification of the study area. The classification methodology used multiple Landsat TM images acquired throughout the 2007 growing season and a set of algorithms was created by training the model against existing wetland classes. The resulting 2007 layer was used both as the basis for the forecast model and also as the observed data to compare against the hindcast results. Model results were then simplified into the specific wetland categories suitable for SLAMM modeling.

One modification made to the land cover data was to demarcate a line between the two fresh marsh types (tidal fresh marshes and inland fresh marshes), based primarily on the most recent U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI) Data (USFWS, 1988). This demarcation was required as the SLAMM conceptual model for these two categories is quite different, with tidal fresh marsh existing lower in the tidal frame and subject to accretion feedbacks as a function of frequency of inundation, as discussed below.

The oldest available historical data from 1956 were used for hindcasting (Wicker, 1980, 1981). These older wetland data did not demarcate between high and low marsh (*i.e.*, irregularly and regularly flooded marshes) respectively, so salt marsh was modeled as a single category within model hindcasting estimates (as was done with SLAMM versions 4 and earlier). Aggregation of the 15-m cells used for modeling indicates the study area was composed of the land cover types shown in Table 1.

A number of diked or impounded areas are present in the study area. The location of dikes was determined through the use of designations in the 1988 NWI data layer (USFWS, 1988) and supplemented with information from the Levees GIS Database developed by the New Orleans District of the USACE (Waguespack, USACE, New Orleans District, *unpublished data*, 2010). The USACE data is considered to be the best available data regarding the locations of federal dikes and levees as of mid-October, 2009. It is important to note that levees maintained by the Louisiana levee districts or individual landowners may not be represented in either of these datasets.

As mentioned previously, dry land areas were classified as protected from inundation within all scenarios. Therefore this study was not a model of dike overtopping or dry land vulnerability under sea-level rise scenarios. In addition, dike maps may have been left incomplete with regards to dry land locations.

Spatially Variable Subsidence Rates

The Louisiana coastline is subject to several interrelated processes that lead to the high amount of subsidence observed in this area. These include tectonic subsidence, sediment compaction, sediment loading, isostatic adjustment, fluid withdrawal, and surface water drainage (Reed and Yuill, 2009). In order to create an accurate representation of the processes

Table 1. *Landcover in study area using 2007 data from Couvillion et al. (2011).*

	2007 Landcover (ha)	% of Study Area
Open Water	1,794,824	58%
Cypress Swamp	280,602	9%
Undeveloped Dry Land	226,801	7%
Regularly Flooded Marsh	180,772	6%
Irregularly Flooded Marsh	166,240	5%
Swamp	131,200	4%
Tidal Fresh Marsh	117,288	4%
Inland Fresh Marsh	101,551	3%
Developed Dry Land	85,241	3%
Tidal Swamp	17,532	1%
Estuarine Beach	1,034	< 1%
Transitional Salt Marsh	465	< 1%
Inland Open Water	395	< 1%

occurring in the study area, it was critical to incorporate the spatial variability of subsidence across the region into the modeling process. Several sources of subsidence data were considered, including the work of Shinkle and Dokka (2004), Ivins, Dokka, and Blom (2007), and data collected by the National Oceanic and Atmospheric Administration (NOAA) and National Ocean Service (NOAA, 2009). Shinkle and Dokka (2004) provided an accounting of subsidence using first-order leveling data and Global Positioning System (GPS) observations from the National Geodetic Survey (NGS). Ivins, Dokka, and Blom (2007) estimated the effects of Holocene sedimentary loading in the Gulf and Mississippi River Delta. Finally, water level (tide gage) data from the National Ocean Service (NOAA, 2012a) was considered along with long-term sea-level rise gages from NOAA (NOAA, 2009). Long-term sea-level rise trends measured at NOAA stations located at Eugene Island and Grand Isle were compared against global long-term trends to produce an estimate of land subsidence in these locations.

In our initial attempts to model historical marsh loss in Louisiana, the model results of Ivins, Dokka, and Blom (2007) were employed to represent subsidence. However, two problems were encountered with this calibration. First, the extent of predicted marsh loss from 1956 to the present date was far too low, even when accretion was calibrated to the low end of observed data and accretion feedbacks were minimized. Secondly, the spatial relationships of marsh loss within the study area, such as high rates of loss in the Bird's Foot Delta, were not present. Part of the reason that the model results of Ivins and colleagues may underestimate subsidence in the region is that their model only accounts for one of the several factors (the weight of loaded sediments) that produce subsidence effects in the Mississippi River Delta.

Subsidence rates included in the final model runs were estimated from point observations primarily derived from geodetic data (Shinkle and Dokka, 2004). These data were added to information obtained from the NOAA tide gages at Grand Isle (gage 8761724) and Eugene Island (gage 8764311) (NOAA, 2009). Subsidence rates at these tide gages were calculated by subtracting the 1.7 mm/yr eustatic sea-level rise trend from the observed at each location, resulting in subsidence rates of 7.5 and 8.0 mm per year, respectively. The full set of points was interpolated to produce a continuous map via kriging. Kriging is a method of interpolating a continuous dataset from observed data at discrete locations (Lang, 2000). The final raster obtained is shown in Figure 3.

While this appears to be the best spatial accounting of subsidence available, there remain several significant sources of uncertainty in this map. Producing a simple two-dimensional interpolation undoubtedly misses important local features such as the effects of diked locations or other subsidence hotspots. Furthermore, the assumption that subsidence rates will remain constant throughout the next 100 years adds additional uncertainty as significant temporal variations in subsidence have been observed (Meckel, 2008). Finally, the source data geodesy is uncertain, especially in the marshes of Louisiana in which many benchmarks are currently unmaintained and have uncertain stability (Reed and Yuill, 2009).

Using the derived subsidence map, an acceptable model calibration was achieved in most of the study area, with the exception of several areas where the subsidence map suggests rapid subsidence, but very little marsh loss was observed. As discussed in the next section, these areas are dominated by floating marshes and this required additional model modification and calibration.

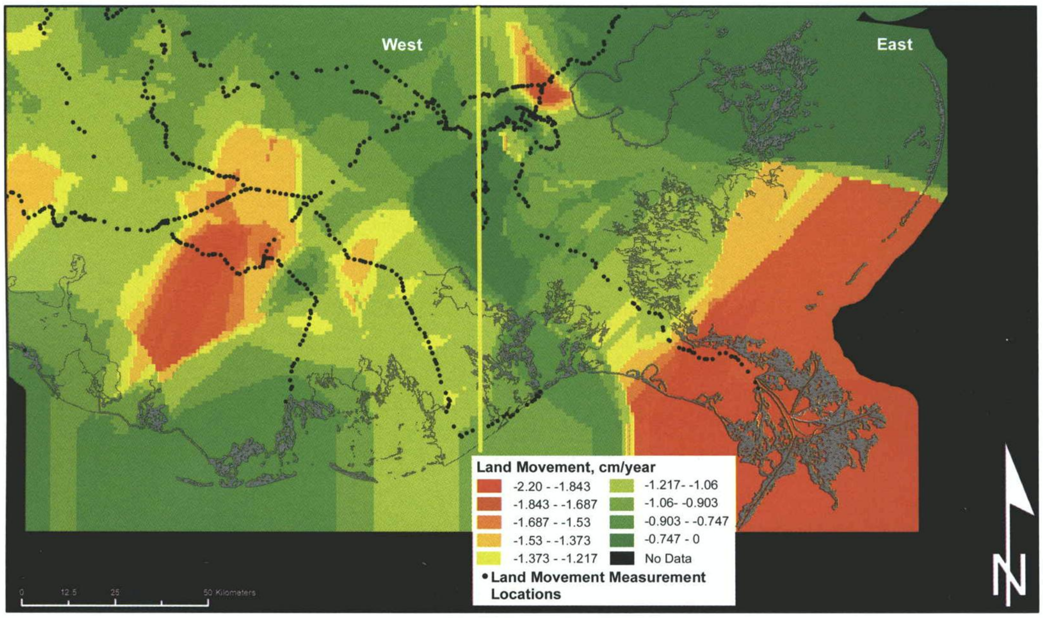

Figure 3. Kriged subsidence raster from Shinkle and Dokka (2004) and long-term NOAA tide gage data.

Cypress Swamp and Flooded Swamp

Much of southern Louisiana is populated by baldcypress-water tupelo (*Taxodium distichum-Nyssa aquatica*) swamps (classified as cypress swamp in this SLAMM application), although the extent and health of these systems has declined considerably over the past century due to a combination of factors, including logging and development, altered hydrology, nutrient and sediment deprivation, and saltwater intrusion (Faulkner *et al.*, 2007; Shaffer *et al.*, 2009). Studies indicate that an increase in the duration and depth of flooding and an increase in salinity stress in some areas, such as the Maurepas swamp of the Pontchartrain Basin, have significantly reduced the ability of these swamps to regenerate (Allen, Pezeshki, and Chambers, 1996; Faulkner *et al.*, 2007; Shaffer *et al.*, 2009; Thompson, Shaffer, and McCorquodale, 2002). Neither baldcypress nor water tupelo seeds can germinate while under water, and their seedlings are unable to survive prolonged periods of flooding (Keim *et al.*, 2006). Evidence also suggests that even established trees can experience a substantial reduction in growth rates and slowly die when subjected to extended deep water flooding (Brown, 1981; Conner and Day, 1988; Harms *et al.*, 1980). Accelerating sea-level rise is likely to exacerbate these stressors, which would lead to further conversion of Louisiana's coastal swamps to marsh and open water (Shaffer *et al.*, 2009).

For this project, SLAMM predicts that cypress swamps convert to flooded swamp when their elevation falls to a level below which nonflooded land will rarely be exposed (ranging from 0.3 to 0.5 m below MTL depending on the local tide range). These habitats often occur at elevations of 2 m above mean sea level and below (Allen, Pezeshki, and Chambers, 1996) and may be regularly inundated with standing water. In addition, site-specific data suggest that this is the lowest elevation inhabited by this wetland type.

The flooded swamp category was added to SLAMM specifically for this application. Cypress trees are considered to be highly tolerant of flooding, even at depths of 3 m or more (Allen, Pezeshki, and Chambers, 1996; Wilhite and Toliver, 1990), and mature trees have been observed to live for 18 years under permanently flooded conditions (Eggler and Moore, 1961). However, permanent flooding prevents seed germination and recruitment, ultimately leading to swamp decline (Shaffer *et al.*, 2009). Generally, at each time step, the SLAMM model estimates what will happen if a given habitat comes to equilibrium with the water levels predicted at that time. However, given the length of time that cypress trees can remain alive within flooded swamps, it was determined that assuming immediate conversion to open water would be potentially misleading. For this analysis, the flooded swamp category is not included in calculations of total coastal wetland losses. In the long run, however, it is likely that these swamps will eventually convert to open water.

Flotant Marsh

Southeastern Louisiana contains a significant amount of floating, or flotant, marsh. These marshes are characterized by a buoyant, 0.3–0.6-meter (1–2 feet) thick organic mat of densely intertwined roots that rise and fall in elevation with changing tides (Sasser *et al.*, 2007). Data collected in 1990 indicate that floating marshes covered more than 1,416.4 km^2 (141,640 ha) in the Barataria and Terrebonne basins (Sasser *et al.*, 2007). These marsh systems are complex and have important differences in mat buoyancy (Sasser *et al.*, 1996).

Floating marshes are difficult to model effectively using SLAMM because they are subject to marsh succession based on water quality (*i.e.*, salinity and organic content) rather than land elevation (Sasser *et al.*, 1996). Unless linked to a hydrodynamic model, SLAMM either uses cell elevation as a surrogate for salinity or employs a simple salt-wedge model. However, cell elevation is not an appropriate surrogate for salinity within floating marshes given that the marsh floats atop the water. Moreover, a salt-wedge model is likely not appropriate given the hydrological complexity of the Louisiana coastline.

Historical data used for hindcasting includes extensive regions of floating marsh. While nutria (*Myocastor coypus*) herbivory is considered to be a major factor in the loss of some floating marsh during the calibration period, these floating marshes have generally been noted to persist throughout much of the region, even within areas of high subsidence (Sasser *et al.*, 2009). To effectively calibrate the model, floating marsh areas were assigned relatively high rates of accretion (12–20 mm/yr) to allow them to keep up with the sea-level rise signal observed over the hindcast period. This approach has considerable implications for model forecasts, as SLAMM predicts marsh losses when local sea-level rise exceeds the rate of marsh accretion. Given the difficulties in predicting historical and future salinity within the floating marshes, the predicted timing of the loss of flotant marshes in current model forecasts is especially uncertain.

Tidal Parameters

Several NOAA tide gages were used to define the tide ranges for southeastern Louisiana (NOAA, 2012a). A gradient of decreasing tidal range from south to north was observed and applied in this simulation. Spatially variable tide range values were incorporated into the model through the use of input subsites. When several tide observations were available, an average value was applied.

Accretion Rates

Accretion data for coastal Louisiana were obtained from several studies published in peer-reviewed journals (Bryant and Chabreck, 1998; Cahoon and Turner, 1989; Nyman, DeLaune, and Patrick, 1990; Nyman *et al.*, 1993, 2006). A total of 40 averaged accretion rates were combined to determine the accretion value used in the Louisiana SLAMM model (Table 2). Each of these data points was based on several cores, and the average accretion value for coastal Louisiana was determined to be approximately 8.2 mm/yr.

For comparison, the average elevation change calculated from data in the Coastwide Reference Monitoring System (CRMS) database was 8.63 mm/yr (shallow subsidence was removed from these data to get a measure of elevation change). From this extensive array of 64 Surface Elevation Table (SET) tables placed throughout the study area in 2006, short-term accretion

Table 2. *Estimate of site-specific accretion rates from literature survey.*

Marsh Type	Average Accretion, mm/year	Samples	Reference
Brackish	8.53	21	Nyman, DeLaune, and Patrick, 1990; Nyman *et al.*, 2006
Brackish/ saline	9.60	155	Nyman *et al.*, 1993
Fresh	8.03	61	Nyman, DeLaune, and Patrick, 1990; Nyman *et al.*, 2006
Intermediate	6.40	3	Nyman, DeLaune, and Patrick, 1990; Nyman *et al.*, 2006
Saline	9.43	290	Nyman, DeLaune, and Patrick, 1990; Nyman *et al.*, 1993, 2006
Unspecified	6.10	474	Cahoon and Turner, 1989; Nyman *et al.*, 1993; Bryant and Chabreck, 1998
Grand Total	8.16	1004	

rates were measured. However, this dataset had considerable variability (ranging from negative 114 mm/yr to positive 60 mm/yr) likely due to the short observation period of the SET data. Analysis of CRMS data suggested a statistically significant relationship between accretion rates and cell elevations (with accretion rates tending to be higher in areas of lower elevation) but with extremely high variability.

Based on the observed relationship between cell elevations and accretion rates within the CRMS dataset, and also strong relationships between elevation and accretion encountered in other studies of marsh accretion (*e.g.*, Kirwan and Murray, 2007; Morris *et al.*, 2002; Mudd, Howell, and Morris, 2009), a negative relationship between cell elevation and the predicted accretion rate was used in this modeling analysis. Because observed accretion rates did not vary significantly as a function of marsh type and no spatial relationships regarding accretion rates were determined within the study area, the same elevation-to-accretion relationship was used throughout the majority of the study area. The exceptions were floating marsh, fan-shaped deltas, and the swamps of the Atchafalaya Delta region.

The extent of accretion feedbacks was used as a model calibration parameter by adjusting the maximum and minimum accretion values while maintaining a mean accretion rate of 8.5 mm/yr for each tidally influenced marsh category (regularly flooded, irregularly flooded, and tidal fresh marshes). The final model calibration resulted in a linear relationship between elevation and accretion with a maximum accretion rate of 11 mm/yr at the low elevation range and a minimum accretion rate of 6 mm/yr at the top of the tidal frame for each tidal marsh category.

Areas known to directly receive increased sediment loads from river inputs were assigned higher accretion values. These included the Atchafalaya River basin in the western portion of the study area and the Bird's Foot Delta in the eastern portion of the study area. In all three areas accretion feedback curves were used to allow marsh accretion rates to range from 12 to 16 mm/yr. In addition, the swamp and tidal swamp accretion rates in the Atchafalaya River basin increased to 8.2 mm/yr. Anthropogenic marsh restoration, such as freshwater diversions and beneficial use of dredge materials, occurring throughout the study area, can affect accretion rates but were not explicitly considered in this analysis. However, the potential effects of a freshwater diversion were examined in a separate model application.

Following successful calibration of the west side of the study area, the same accretion feedbacks were applied to the east side of the study area allowing for a limited model validation. As will be shown in the results section, model results for the east side validation were equally strong, as compared to the west side calibration. Although more accurate than applying a constant accretion rate, it is acknowledged that this model of accretion feedback is fairly simple and does not explicitly account for other mechanistic relationships that may affect marsh accretion rates such as biogenic production and soil compaction. For example, research suggests that elevated atmospheric CO_2 may stimulate marsh plant productivity, particularly belowground, which could enhance the ability of at least some marshes to increase in elevation (Langley *et al.*, 2009). Another factor important for marsh accretion that is not captured in this model is the role of nutrients (*e.g.*, Brantley *et al.*, 2008; Day *et al.*, 2004).

Erosion Rates

This modeling analysis did not calibrate results using beach or marsh erosion rates for several reasons. First, there are limited beaches within the study area and these are primarily on barrier islands. Barrier island outcomes were not emphasized due to complications in modeling overwash, storm effects, and barrier island migration using a nonhydrodynamic model such as SLAMM. Second, research by Morton *et al.* (2005, 2010) suggests that subsidence has been a greater contributor than erosion to historical marsh loss in the Mississippi River Delta with some localized exceptions (*e.g.*, parts of the lower deltaic plain). Third, a review of peer-reviewed literature provided little information about marsh erosion rates within the study area. Finally, SLAMM sensitivity analyses have found the model to be considerably less sensitive to erosion rate parameters as opposed to accretion parameters (Chu-Agor *et al.*, 2010).

The erosion rates applied were the default values for SLAMM. Marsh erosion was set to 1.8 horizontal m/yr, swamp erosion was set to 1 m/yr, and tidal flat erosion was set to 2 m/yr. It is also important to note that erosion only occurs in SLAMM if (1) the land is in contact with open water, and (2) the maximum wave fetch requirement of 9 km is met (Clough, Park, and Fuller, 2010).

MTL to NAVD 88 Correction

Available elevation data were based on the North American Vertical Datum of 1988 (NAVD 88). In order to estimate the frequency of inundation for each model cell for use within

SLAMM, these data were converted from a fixed vertical datum to a tidal datum. The Louisiana study area is covered by VDATUM, the vertical datum transformation product available from NOAA. Given limited variability within subsites, the NAVD 88 to MTL correction was applied on a subsite basis rather than a cell-by-cell basis using VDATUM results. The standard deviation of the VDATUM transformation in this area is 14.8 cm (NOAA, 2012b).

The NAVD 88 to MTL correction was used for model calibration near Lake Maurepas in the northwestern portion of the study area. During the calibration process this area was immediately converting to open water due to low initial-condition elevations. To prevent this, NAVD 88 correction values were modified to allow SLAMM output to more closely match the observed initial wetland coverage. This was warranted due to a lack of adequate data on historical elevations (in 1956) and extremely limited information about the extent of tides in the Maurepas area.

Elevation Analysis

The cell size used for this analysis was 15 m by 15 m; SLAMM also tracks partial conversion of cells as a function of their elevation and slope (*i.e.*, one 15-m x 15-m cell can contain more than one wetland category). SLAMM assumes that tidal wetlands inhabit a range of vertical elevations that is a function of the tide range. The relationships between wetland type and elevation generally align with those in the SLAMM conceptual model; however, there are occasional site-specific differences, especially in microtidal regimes (Clough, Park, and Fuller, 2010). In this application, the SLAMM conceptual model was modified to more accurately reflect the unique elevation characteristics of southern Louisiana. In particular, the lower bounds for tidal fresh and regularly flooded marsh were decreased to -0.05 m and -0.1 m, respectively. The tidal swamp and cypress swamp categories were also adjusted to allow these land cover types to extend lower into the tidal frame. The lower bound of tidal swamp was decreased to the mean tide level and the cypress swamp was allowed to extend to 0.5 m below mean tide level. These values were determined based on a literature review and the site-specific elevation data shown in Tables 3 and 4.

Temporal Aspect

The SLAMM simulation was run starting from the wetland cover data photo date as the initial condition. In this study a 25-year time step was selected for forecasting and a 10-year time step was selected for hindcasting.

RESULTS

Hindcast Results

Several metrics may be employed to judge the accuracy of the model based on the hindcast results. Because of the high potential for local uncertainty, the method used for evaluation focuses on assessing the capability of the model to predict overall trends in land cover over time.

The primary metric used to evaluate SLAMM hindcast results in this study is the percent of the land cover lost during the model simulation for the primary wetland/vegetation types. The percentage loss predicted by the model is compared to the percentage of the actual land cover lost determined by comparing the historical to contemporary wetland coverage datasets.

Historical records indicate approximately 10.9 cm of eustatic sea-level rise occurred during the hindcast period (1956–2007) (Grinsted, Moore, and Jevrejeva, 2009; IPCC, 2007). SLAMM converts these eustatic trends into a spatial map of relative sea-level rise using the map of subsidence rates derived for the region.

Numerical results are reported in Table 5 for the west and east sides of the study area along with the total results for the entire

Table 3. *SLAMM assumptions (conceptual model) compared to land-cover elevations derived from data for the western part of the study area.*

	SLAMM Conceptual Model: West Side						
SLAMM Category	Min (m)	Max (m)	n cells	5th Pct. (m)	95th Pct. (m)	mean (m)	st. dev. (m)
Cypress Swamp	-0.50	0.50	12103810	0.07	1.24	0.43	0.46
Undeveloped Dry Land	0.27	0.50	8231101	-0.70	5.34	1.90	1.85
Swamp	0.27	0.50	4930273	-0.59	3.82	1.09	1.44
Tidal Fresh Marsh	0.11	0.27	4313143	-0.05	0.53	0.25	0.22
Inland Fresh Marsh	-0.05	0.27	4116016	-0.10	1.47	0.36	0.88
Irregularly Flooded Marsh	0.08	0.38	3169151	-0.16	0.57	0.21	0.32
Regularly Flooded Marsh	-0.10	0.38	3012121	-0.10	0.42	0.18	0.18
Developed Dry Land	0.27	0.50	1714204	-0.13	6.05	2.34	1.90
Tidal Swamp	0.00	3.05	779672	-0.03	0.65	0.33	0.26
Estuarine Beach	-0.16	0.27	22101	-0.27	4.21	0.76	1.51
Transitional Salt Marsh	0.16	0.27	17311	0.11	0.55	0.30	0.16
Inland Open Water	0.27	3.05	15666	-0.63	0.30	-0.06	0.48

Table 4. *SLAMM assumptions (conceptual model) compared to land-cover elevations derived from data for the eastern part of the study area.*

	SLAMM Conceptual Model: East Side						
SLAMM Category	Min (m)	Max (m)	n cells	5th Pct. (m)	95th Pct. (m)	mean (m)	st. dev. (m)
Regularly Flooded Marsh	-0.10	0.31	4565639	-0.15	0.59	0.26	0.24
Irregularly Flooded Marsh	0.07	0.31	3462801	-0.16	0.65	0.27	0.30
Tidal Fresh Marsh	-0.05	0.22	753891	-0.12	0.87	0.28	0.40
Developed Dry Land	0.22	0.50	504198	-1.62	2.31	0.16	1.49
Undeveloped Dry Land	0.22	0.50	473091	-1.57	8.00	1.07	2.52
Swamp	0.22	0.50	161616	-1.40	1.60	0.03	0.97
Cypress Swamp	-0.50	0.50	152788	-0.51	1.12	0.20	0.57
Inland Fresh Marsh	0.22	0.50	54058	-1.30	2.62	0.27	1.06
Estuarine Beach	-0.13	0.22	18818	-0.24	0.95	0.07	0.45
Transitional Salt Marsh	0.13	0.22	3261	0.10	0.82	0.40	0.23
Inland Open Water	0.22	0.50	198	-0.22	1.24	0.20	0.72

study extent. The west-side model was calibrated as to the extent of accretion-rate feedbacks, floating-marsh accretion rates, and the relationship between tide ranges and cell elevations for Maurepas swamp. The west-side parameters were then applied to the east side of the simulation with the only calibration being an adjustment of accretion rates for a small portion of tidal-fresh floating marsh on the east side. Results shown in Table 5 suggest the calibrated SLAMM model for Louisiana closely predicts the amount of salt marsh loss observed, predicting 25% loss when 25% of marsh loss was observed. The fresh marsh loss model was stronger on the calibrated west side of the simulation than the noncalibrated eastern side of the model. Fresh marsh losses are underpredicted on the east side of the model. This may be partially due to the effects of recent major hurricanes on the east-side model domain, as will be discussed below.

Some model calibration was performed on the west side of the model to limit the overprediction of swamp loss. However, this analysis did not attempt to replicate numeric metrics precisely due to uncertainties in historical elevation data and lack of historical information about the locations of cypress swamps. In addition, it was assumed that some predicted swamp loss remains pending due to the time it takes for trees to die and woody debris to break down. Because of the uncertainty between flood time and swamp loss, in model forecasts cypress swamps are predicted to convert to flooded swamps rather than to open water. On the east side of the study area, where cypress swamps are less pervasive, the hindcast predicted 60% swamp loss as opposed to an observed loss of 56%. Table 5 indicates that model forecasts may be conservative in predicting the loss of fresh marsh (*i.e.*, less marsh loss will be predicted), while salt marsh predictions are likely to be accurate when averaged over the entire landscape.

Impacts of Hurricanes on Coastal Louisiana Marshes

The primary purpose of SLAMM is to predict land cover changes based on long-term sea-level rise effects; however, not all the changes observed throughout the study period were due to relative sea-level rise. In Louisiana, hurricanes can and have had a profound effect on the configuration of marsh communities (Morton and Barras, 2011). For example, following the 2005 hurricane season, approximately 562 km^2 (56,200 ha) of newly created open water areas were observed (Barras, 2007), particularly in low salinity wetlands such as those in Breton Sound and Bird's Foot Delta (Howes *et al.*, 2010). As hurricanes were not explicitly modeled within SLAMM, it is plausible that

Table 5. *Hindcast results. Percentage of land cover lost between 1979 and 2007 as predicted and observed. Salt Marsh category includes regularly and irregularly flooded marsh; Fresh Marsh includes inland and tidal fresh marsh; Swamp includes the 'swamp' and 'cypress swamp' categories.*

	WEST		EAST		TOTAL	
Land cover type	Predicted	Observed	Predicted	Observed	Predicted	Observed
Salt Marsh	12%	19%	33%	28%	25%	25%
Fresh Marsh	41%	44%	63%	79%	49%	57%
Total Marsh	28%	33%	43%	44%	35%	39%
Swamp	41%	13%	60%	56%	46%	24%
Beach	99%	56%	94%	84%	96%	76%

the underprediction of fresh marsh loss on the east side of the study area in the hindcast could be attributed in part to the recent hurricane-induced failure of those wetlands.

Forecast Results

Following the model calibration and validation, the model was applied using four different sea-level rise scenarios through 2100. Land cover and elevation input data for 2007 was more recent and therefore more reliable than input data from 1956, which should result in more accurate model predictions.

To assist in comparisons with historical losses and other model projections, the total area of wetland loss was calculated for each time step (Table 6) and in percentage losses (Table 7). Swamp, cypress swamp, tidal swamp, regularly and irregularly flooded marsh, tidal and inland fresh and transitional salt marsh, and estuarine beach SLAMM categories were included in these total wetland loss metrics; conversion to permanently flooded swamp was not considered a wetland loss.

Overall, model results for the rapid climate stabilization case (3.1 mm/year) suggest a vastly changed landscape within southeastern Louisiana—a continuation of marsh losses that have been occurring for the past 50 years. One exception is a projected 6% increase in the area of regularly flooded marsh, likely due to the conversion of irregularly flooded and fresh marshes across the site.

Table 6. *Hectares of wetland loss by each timestep simulated (excludes permanently-flooded swamp).*

	Predicted Sea-Level Rise by Year			
2100 (m)	2025 (ha)	2050 (ha)	2075 (ha)	2100 (ha)
0.34 (linear)	30,326	80,247	147,784	218,897
0.75	33,010	134,972	295,951	425,769
1.22	43,477	223,975	382,944	524,635
1.90	64,624	288,743	429,049	587,527

Table 7. *Estimated percent loss in landcover categories by 2100 for each sea-level rise scenario (the lone negative value indicates a category gain).*

	Predicted Sea-Level Rise by 2100			
Land cover type	0.34 (m)	0.75 (m)	1.22 (m)	1.9 (m)
Cypress Swamp[1]	66%	86%	93%	96%
Regularly Flooded Marsh	-6%	42%	55%	71%
Irregularly Flooded Marsh	52%	75%	82%	92%
Swamp	53%	61%	68%	74%
Tidal Fresh Marsh	30%	66%	91%	96%
Inland Fresh Marsh	42%	73%	86%	88%
Tidal Swamp	95%	98%	99%	99%
Estuarine Beach	82%	86%	88%	89%

[1]Cypress Swamp loss includes cypress swamp that becomes permanently flooded.

Predictions for higher rates of sea-level rise are understandably more dramatic. What Vermeer and Rahmstorf (2009) suggested to be the most likely sea-level rise scenario (1.22 m by 2100) results in loss predictions of over 55% for all wetland categories, with swamp loss predicted at 68% or greater.

Tables 8 and 9 show numerical results for projected habitat changes across all time steps for the 0.34 m by 2100 and 1.22 m by 2100 scenarios, respectively. Figures 4 and 5 show map-based results for projected habitat changes for the year 2050 under the 0.34 m by 2100 and 1.22 m by 2100 scenarios, respectively.

Among the habitats that appear particularly vulnerable to accelerating sea-level rise are the region's irregularly flooded (primarily brackish and intermediate) marshes. These marshes, which comprised approximately 1,662.4 km^2 (166,240 ha) in the initial condition year (2007), are projected to decline by 52% under the linear 0.34 m by 2100 scenario and 82% under the accelerating 1.22 m by 2100 scenario.

Another vulnerable habitat is the region's baldcypress-water tupelo (*Taxodium distichum-Nyssa aquatica*) swamps. Under the rapid climate stabilization case (3.1 mm/year), nearly all cypress swamp in the study is predicted to convert to flooded swamp by 2100. Cypress swamps in the Atchafalaya Delta and the Lake Maurepas region are both predicted to be affected, despite the physical differences between the two. In fact, this trend has already been noted by scientists. In a seven-year study of the Maurepas swamp, Shaffer *et al.* (2009) noted that nearly 20% of the trees in their study plots suffered mortality, and recruitment of both baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*) saplings were essentially absent.

The modeled predictions under the scenario of 0.34 m by 2100 suggest that floating marsh will be resilient to sea-level rise while the swamp and marshlands behind them will be inundated and convert to open water. However, as described elsewhere, there is a high degree of uncertainty associated with predictions regarding floating marsh in this analysis.

Under higher sea-level rise scenarios, marshes that are not diked are predicted to convert to tidal flat and open water. In the 0.75 m sea-level rise by 2100 scenario (not shown), the outer marshes of the Terrebonne and Barataria basins are predicted to convert to open water by 2100. For example, in the western portion of the study area, Point au Fer Island is predicted to be completely inundated by 2100. In the eastern portion, nearly all the marsh that is not protected by dikes is predicted to convert to tidal flat by 2100. Moreover, the marsh areas separating Lake Borgne and Lake Pontchartrain are predicted to be lost by 2100 under the 0.75 m and higher sea-level rise scenarios.

Under the 1.22 m and 1.9 m by 2100 sea-level rise scenarios, only small amounts of marsh are predicted to remain. These are predominantly in areas designated as floating marsh, and their persistence is therefore subject to considerable uncertainty. Under the 1.9 m scenario (not shown), the Atchafalaya Delta area is predicted to become open water.

Estimating the Potential Impact of River Diversions

One reason for marsh loss in southern Louisiana is the

Table 8. *Projected habitat coverage in hectares under scenario of .34 m of sea-level rise.*

	2007	2025	2050	2075	2100
Open Water	1,794,824	1,808,527	1,831,580	1,886,451	1,953,484
Cypress Swamp	280,602	229,786	180,725	130,972	95,724
Undeveloped Dry Land	226,801	226,801	226,801	226,801	226,801
Regularly Flooded Marsh	180,772	234,370	228,573	215,024	192,158
Irregularly Flooded Marsh	166,241	124,501	113,290	100,086	79,732
Swamp	131,200	98,786	81,778	69,957	62,120
Tidal Fresh Marsh	117,287	107,905	99,392	88,955	81,678
Inland Fresh Marsh	101,551	78,843	70,199	62,802	58,403
Developed Dry Land	85,241	85,241	85,241	85,241	85,241
Tidal Swamp	17,532	13,961	7,869	2,472	906
Estuarine Beach	1,034	529	336	254	187
Transitional Salt Marsh	465	26,872	34,407	28,763	22,015
Inland Open Water	394	103	92	87	86
Flooded Swamp	0	50,806	99,868	149,613	184,861
Tidal Flat	0	16,914	43,794	56,465	60,547
Total (incl. water)	3,103,944	3,103,944	3,103,944	3,103,944	3,103,944

isolation of wetlands from inputs of riverine sediments due to flood-control infrastructure (Martin *et al.*, 2002). Because of the reduction of sediment inputs, marshes succumb to subsidence and are rapidly converted to open water (Day *et al.*, 2011a). Providing marshes with additional river water may enhance vertical accretion both directly (through sediment deposition) and indirectly (through enhanced root growth), and this can produce marshes that are more resilient to sea-level rise (Day *et al.*, 2011a). Therefore, one of the major strategies for preserving coastal wetlands in Louisiana is to restore sediment inputs through the reintroduction of river flow from the Mississippi River.

One of the largest restoration-oriented diversions off the Mississippi in southern Louisiana is the Caernarvon diversion. The Caernarvon is located on the east bank of mile 81.5 of the river, just east of New Orleans. The structure was completed in 1991 and has flow capacity of approximately 226 m^3 per second (Lane *et al.*, 2002). Flows through the diversion are highly dependent on the time of year. Over the last decade, the flow through Caernarvon has averaged 57 m^3 per second.

Table 9. *Projected habitat coverage in hectares under scenario of 1.22 m of sea-level rise.*

	2007	2025	2050	2075	2100
Open Water	1,794,824	1,810,043	1,864,446	2,036,332	2,189,695
Cypress Swamp	280,602	216,332	115,125	41,202	20,638
Undeveloped Dry Land	226,801	226,801	226,801	226,801	226,801
Regularly Flooded Marsh	180,772	240,784	175,513	154,141	81,923
Irregularly Flooded Marsh	166,241	114,217	87,786	52,953	29,545
Swamp	131,200	94,719	68,622	52,298	41,973
Tidal Fresh Marsh	117,287	103,821	59,249	25,115	10,047
Inland Fresh Marsh	101,551	72,750	31,852	17,323	14,644
Developed Dry Land	85,241	85,241	85,241	85,241	85,241
Tidal Swamp	17,532	12,499	1,433	288	173
Estuarine Beach	1,034	481	271	165	128
Transitional Salt Marsh	465	33,340	67,389	30,863	13,021
Inland Open Water	394	102	86	73	71
Flooded Swamp	0	64,261	165,469	239,391	259,955
Tidal Flat	0	28,551	154,663	141,758	130,089
Total (incl. water)	3,103,944	3,103,944	3,103,944	3,103,944	3,103,944

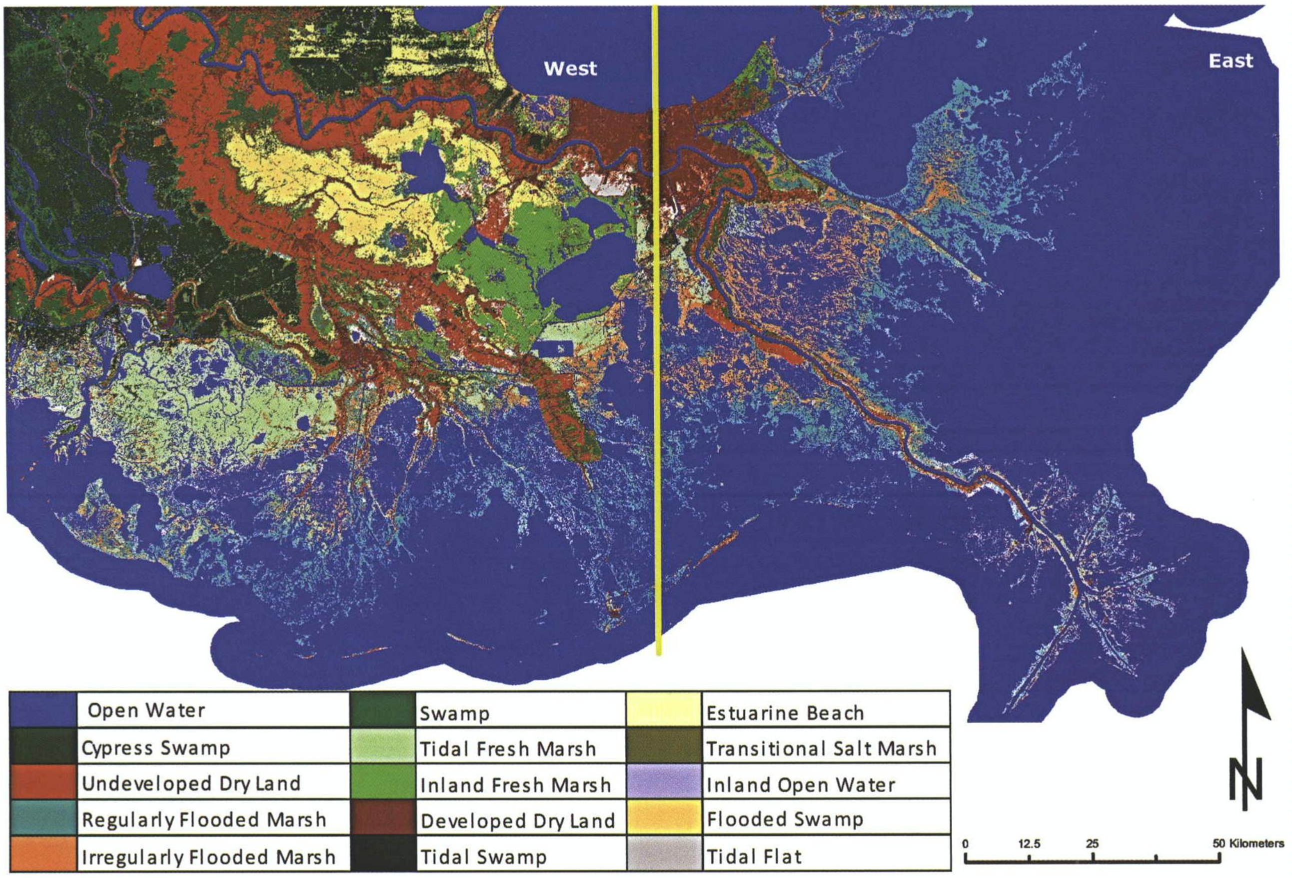

Figure 4. Projected spatial distributions of land-cover categories for entire study area in 2050, under scenario of .34 m sea-level rise.

In the model calibration and forecast presented above, the Caernarvon diversion is not explicitly included. However, an additional case study was created to test the ability of SLAMM to capture potential impacts of diversions such as this to provide an illustrative example of possible management responses. While SLAMM is not a hydrodynamic model, the effect of the Caernarvon diversion can be represented through a modified accretion to elevation relationship as a consequence of the increased sediment load. One important caveat to this approach, however, is the fact that the relationship between sediment load and accretion is not linear and is dependent on many biotic and abiotic factors. Furthermore, this analysis does not consider the many other impacts that such diversions can have on the ecological systems of the Louisiana coast, such as nutrient budgets, phytoplankton responses, and food web dynamics (Day *et al.*, 2009). Nevertheless, even a simplified application of possible management responses may provide a useful tool.

Accordingly, a new accretion relationship was determined for the area affected by the Caernarvon diversion by estimating the new sediment input and using this number as an input for the accretion model developed by Morris *et al.* (2002). Sediment loading was calculated following the approach of Lane, Day, and Day (2006). Average yearly flow through the diversion was determined from data collected at a USGS water monitoring station at the Caernarvon Outfall Channel (station 295124089542100) (USGS, 2012b). The suspended sediment concentration in the diverted water was assumed to be equal to the concentration of suspended sediment measured at the USGS monitoring station at St. Francisville, Louisiana (station 07373420) (USGS, 2012b). By assuming the area of influence of the Caernarvon diversion is 1100 km^2 (Lane, Day, and Day, 2006), sediment loading was estimated to be 0.187 g/cm^2 per year. Results of this preliminary analysis suggested that this addition of sediment to the marshes beyond the Caernarvon diversion could increase accretion rates by nearly a factor of two at lower elevations, but marshes at the upper end of the tidal frame might be largely unaffected. To represent this, accretion rates ranging from 18 mm/yr to 6 mm/yr were applied to the area with the diversion rather than from 11 mm/yr to 6 mm/yr, which were applied to this area not assumed to be affected by the diversion.

Model results suggest that the Caernarvon diversion could make an inportant difference in this region, especially under moderate scenarios of sea-level rise. For example, within the area of Breton Sound assumed to be influenced by the diversion, given a scenario of 0.75 m of sea-level rise by 2100, 90% of

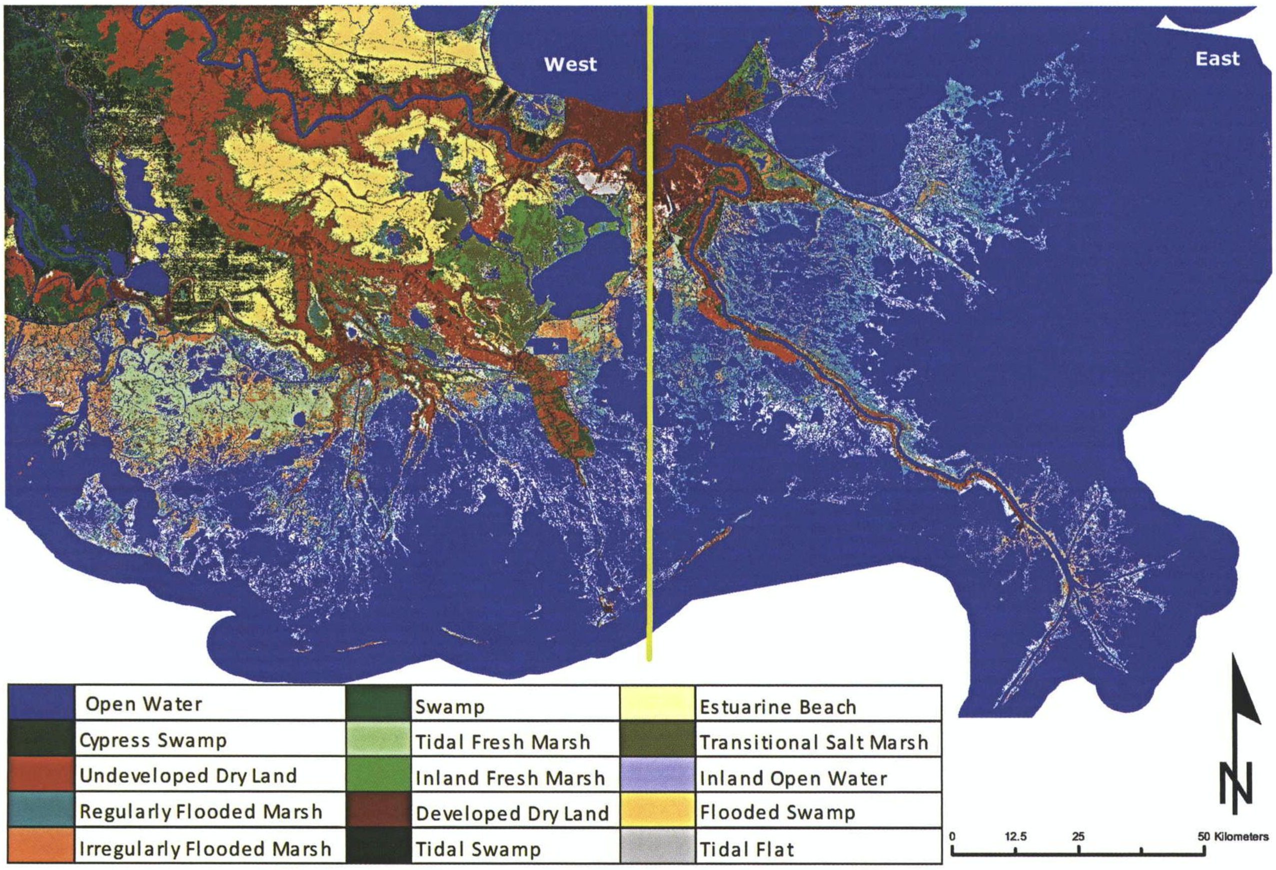

Figure 5. Projected spatial distributions of land-cover categories for entire study area in 2050, under scenario of 1.22 m sea-level rise.

irregularly flooded marsh is predicted to be retained in 2050 post diversion, but only 60% without the diversion (Figure 6). However, under a scenario of 1.22 m of sea-level rise by 2100, the diversion is predicted to retain only 36% of irregularly flooded marsh as compared with 27% retained marsh with no diversion. These results suggest that there is a breakpoint in terms of sea-level rise scenarios beyond which sediment introduction is not capable of protecting marshes within the area of influence of the Caernarvon diversion. Based on the scenarios examined in this study, this breakpoint occurs near 1.0 m of eustatic sea-level rise by 2100. Figure 7 suggests that a similar relationship holds true when examining predictions of marsh losses by 2100.

DISCUSSION AND CONCLUSIONS

The goal of this study was to model the potential effects of sea-level rise on southeastern Louisiana using a relatively simple nonhydrodynamic model. The western half of the study area was initially calibrated to historical data, closely matching trends in marsh and swamp losses from 1956 to 2007. Feedbacks between accretion rates and marsh elevations, and the rate of accretion in floating marshes, were the primary calibration parameters. The model was then run for the eastern half of the study area with the same parameter set to provide a limited model validation. This calibrated model was applied to predict effects of future sea-level rise on coastal Louisiana given global sea-level rise scenarios ranging from 0.34 m to 1.90 m by 2100. Results suggest continued overall marsh losses for the region even under the most optimistic rapid stabilization sea-level rise scenario. Marsh and swamp losses range from 42% to 99% by 2100 (depending on land cover category) when global sea-level rise exceeds 0.75 meters by that date.

One significant wildcard for the future of coastal Louisiana's marshes is the role of hurricanes. In Louisiana, hurricanes can and have had a profound effect on the configuration of marsh communities (Morton and Barras, 2011). As highlighted previously, major hurricanes in 2005 and 2008 damaged extensive areas of coastal marshes in southeastern Louisiana (Barras, 2009). Many of these losses occurred in low salinity wetlands, particularly on the eastern side of the study area. Although this analysis does not account for potential future damage from storms, such impacts should be a significant consideration for coastal restoration efforts in Louisiana. In particular, given that many of the marshes supported by freshwater diversion will be low salinity wetlands, those areas may be subject to the same types of failures observed following the recent hurricane events (Howes *et al.*, 2010).

Additionally, it is important to recognize that this study

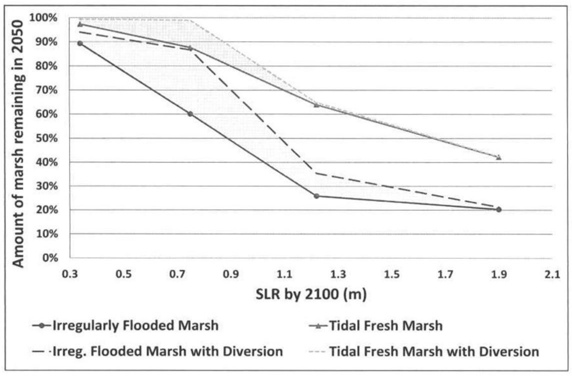

Figure 6. Percent of remaining regularly-flooded and tidal-fresh marsh in the Breton Sound in 2050 with and without the effects of the Caernarvon diversion.

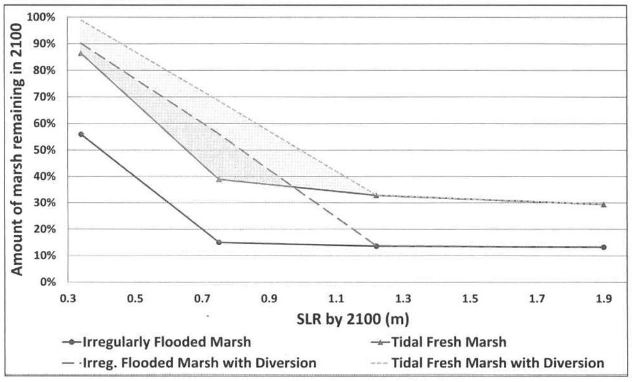

Figure 7. Percent of remaining regularly-flooded and tidal-fresh marsh in the Breton Sound in 2100 with and without the effects of the Caernarvon diversion

presumes that all currently developed areas, agricultural zones, and other areas categorized as dry land will be protected by dikes, levees, or other coastal armoring. Accordingly, these areas do not show inundation under the various scenarios for sea-level rise used in the model. This does not mean, however, that low-lying communities such as New Orleans are not also vulnerable to sea-level rise, only that the potential impacts are not captured here. While it is beyond the scope of this study to address impacts on the region's vulnerable developed areas, the potential for sea-level rise to cause significant and costly damage to property and infrastructure should not be ignored. Coastal managers will also want to consider possible areas (*e.g.*, less developed land currently used for agriculture) where selected dike removal may enable some wetland habitat migration.

This study suggests that scientists and managers will need to find ways to balance near-term restoration goals for species and habitats with longer-term goals for sustaining functional ecological systems that are likely to persist under future conditions.

State and federal agencies, nongovernmental organizations, and others concerned with coastal Louisiana restoration are challenged with designing and implementing projects that take plausible sea-level rise scenarios into consideration in order to maximize the effectiveness of restoration investments. This study is intended to provide coastal resource managers and other relevant decisionmakers with information about the potential impacts of sea-level rise on the region's coastal wetland habitats to help them assess the risks and identify reasonable steps to manage those risks.

Appropriate response strategies will necessarily vary for different areas, and site-specific studies will be warranted to supplement these findings and address factors that have not been effectively characterized by the model. However, the results of this analysis can be used to inform a number of important decisions regarding coastal restoration and management. In particular, managers should consider the following actions:

(1) Prioritize project sites based on ecological importance and vulnerability to sea-level rise.

Given limited conservation resources, it will be important for governmental and nongovernmental decisionmakers to consider where and what coastal restoration and management efforts will be most effective in supporting specific conservation goals, such as protecting important ecosystem services, rare species, iconic places, and human communities, given the added threat of sea-level rise.

Model results suggest that one area of particular concern is the apparent vulnerability of the region's baldcypress-water tupelo (*Taxodium distichum-Nyssa aquatica*) swamp habitat, much of which is projected to convert to flooded swamp even under the lower scenarios for sea-level rise in the coming decades. Given that permanently flooded swamps are unable to regenerate, management decisions regarding restoration and harvest of these trees will need to take these impacts into consideration. Furthermore, the degradation of cypress swamp is likely to affect the potential benefits these habitats would provide for regional carbon sequestration efforts.

(2) Restore and protect a diverse array of habitat types to better support ecosystem functions and improve the resilience of fish and wildlife.

It will be important to develop strategies that restore or maintain a diverse array of habitat types, and habitat connectivity, to better support ecosystem functions and improve the resilience of fish and wildlife. This study can help in this effort by identifying potential changes in habitat composition and diversity, as well as increased fragmentation due to sea-level rise.

(3) Identify areas that may warrant specific adaptation strategies such as natural and artificial replenishment of sediments.

In general, it is possible for managers to help coastal habitats in some areas adapt to sea-level rise by restoring natural processes that supply sediments to estuaries, coastal marshes, and beaches, or by considering strategies such as the managed avulsion of rivers and assisted accretion (using dredged materials to replenish coastal marshes).

The predicted persistence of marsh in the Atchafalaya River Delta under all but the highest scenario for sea-level rise by 2100 suggests that adequate supplies of sediment can boost marsh resilience. This potential is further illustrated by the simulation of additional sediment input from the Caernarvon diversion into Breton Sound, which model results suggest could significantly reduce marsh loss under moderate (*e.g.*, 0.75 m by 2100) sea-level rise when compared to the same scenario without additional sediment input.

(4) Expand restoration areas and coastal protection strategies to accommodate habitat migration.

With additional research on land use trends and infrastructure vulnerability to sea-level rise, managers can identify areas where there is potential to protect habitat buffers and enable inland migration, such as capitalizing on opportunities to protect land where there is currently little or no development (*e.g.*, marginal agricultural lands).

For example, the region could broaden considerations and opportunities for targeted land acquisition, rolling easements, tax incentives, and other strategies to discourage additional development in vulnerable areas, which will protect people and habitats.

(5) Expand monitoring and adaptive management practices.

Finally, there will always be uncertainty about how, when, and where climate change (including associated sea-level rise) will affect natural systems. Increased monitoring and research on the known and potential consequences of climate change on species and habitats will help close the gap in knowledge, but we will never know exactly when and where we will experience the impacts until they occur. Taking such uncertainty into consideration, it is prudent to consider actions we can take now to reduce our vulnerability while building in the flexibility to revise strategies as we learn more.

ACKNOWLEDGEMENTS

We sincerely thank the following individuals for providing invaluable assistance and advice in the development of this study: John Barras, John Brock, Virginia Burkett, Greg Steyer, Cindy Thatcher, and Hongqing Wang of USGS; Brian Czech, Leo Miranda, and Bill Wilen of USFWS; Don Boesch of the University of Maryland Center for Environmental Science; Cindy Brown of The Nature Conservancy of Louisiana; Syed Khalil and James Pahl of the Coastal Protection and Restoration Authority of Louisiana; Jay Martin of Ohio State University; Dick Park of Eco Modeling; Jay Ratcliff of the U.S. Army Engineering Research and Development Center; and Robert Twilley of the University of Louisiana at Lafayette. We also appreciate the assistance of Marco Propato of Warren Pinnacle Consulting, Inc., for his assistance with formatting figures. This project was made possible through a generous grant from the Walton Family Foundation. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

Allen, J.A.; Pezeshki, S.R., and Chambers, J.L., 1996. Interaction of flooding and salinity stress on baldcypress (*Taxodium distichum*). *Tree Physiology*, 16(1-2), 307–313.

Allen, Y.C.; Couvillion, B.R., and Barras, J.A., 2011. Using multitemporal remote sensing imagery and inundation measures to improve land change estimates in coastal wetlands. *Estuaries and Coasts*, 35(1), 190–200.

Barras, J.A., 2007. Land area changes in coastal Louisiana after Hurricanes Katrina and Rita. *In*: Farris, G.S.; Smith, G.J.; Crane, M.P.; Demas, C.R.; Robbins, L.L., and Lavoie, D.L. (eds.), Science and the Storms: The USGS Response to the Hurricanes of 2005. *U.S. Geological Survey Circular 1306*, pp. 98–113. URL: http://pubs.usgs.gov/circ/1306/; accessed on August 16, 2012.

Barras, J.A., 2009. Land Area Change and Overview of Major Hurricane Impacts in Coastal Louisiana, 2004–2008. *U.S. Geological Survey Scientific Investigations Map 3080*, scale 1:250,000, 1 sheet, 6 p. pamphlet. URL: http://pubs.usgs.gov/sim/3080/; accessed on August 16, 2012.

Barras, J.A.; Bernier, J.C., and Morton, R.A., 2008. Land Area Change in Coastal Louisiana–A Multidecadal Perspective (from 1956 to 2006). *U.S. Geological Survey Scientific Investigations Map 3019*, scale 1:250,000, 1 sheet. URL: http://pubs.usgs.gov/sim/3019/; accessed on August 16, 2012.

Barras, J.A.; Brock, J.C.; Morton, R.A., and Travers, L.J., 2010. Satellite Images and Aerial Photographs of the Effects of Hurricanes Gustav and Ike on Coastal Louisiana. *U.S. Geological Survey Data Series 566*. URL: http://pubs.usgs.gov/ds/566/; accessed on August 16, 2012.

Betts, R.A.; Collins, M.; Hemming, D.L.; Jones, C.D.; Lowe, J.A., and Sanderson, M., 2011. When could global warming reach 4ºC? *Philosophical Transactions of the Royal Society A*, 369(1934), 67–84.

Blum, M.D. and Roberts, H.H., 2009. Drowning of the Mississippi Delta due to insufficient sediment supply and global sea-level rise. *Nature Geoscience*, 2, 488–491.

Boesch, D.F., 2006. Scientific requirements for ecosystem-based management in the restoration of Chesapeake Bay and Coastal Louisiana. *Ecological Engineering*, 26(1), 6–26.

Boustany, R.G., 2010. Estimating the benefits of freshwater introduction into coastal wetland ecosystems in Louisiana: Nutrient and sediment analyses. *Ecological Restoration*, 28(2), 160–174.

Brantley, C.G.; Day, J.W., Jr.; Lane, R.R.; Hyfield, E.; Day, J.N., and Ko, J.-Y., 2008. Primary production, nutrient dynamics, and accretion of a coastal freshwater forested wetland assimilation system in Louisiana. *Ecological Engineering*, 34(1), 7–22.

Brown, S., 1981. A comparison of the structure, primary productivity, and transpiration of cypress ecosystems in Florida. *Ecological Monographs*, 51(4), 403–427.

Bryant, J.C. and Chabreck, R.H., 1998. Effects of impoundment on vertical accretion of coastal marsh. *Estuaries*, 21(3), 416–422.

Burkett, V.R.; Wilcox, D.A.; Stottlemyer, R.; Barrow, W.; Fagre, D.; Baron, J.; Price, J.; Nielson, J.L.; Allen, C.D.; Peterson, D.L.; Ruggerone, G., and Doyle, T., 2005. Nonlinear dynamics in ecosystem response to climate change: Case studies and policy implications. *Ecological Complexity*, 2(4), 357–394.

Cahoon, D.R., 2006. A review of major storm impacts on coastal wetland elevations. *Estuaries*, 29(6), 889–898.

Cahoon, D.R. and Turner, R.E., 1989. Accretion and canal impacts in a rapidly subsiding wetland II. Feldspar marker horizontal techniques. *Estuaries*, 12(4), 260–268.

Callaway, J.C.; Parker, V.T.; Vasey, M.C., and Schile, L.M., 2007. Emerging issues for the restoration of tidal marsh ecosystems in the context of predicted climate change. *Madroño*, 54(3), 234–248.

Cazenave, A. and Llovel, W., 2010. Contemporary sea level rise. *Annual Review of Marine Science*, 2(1), 145–173.

Chen, J.L.; Wilson, C.R., and Tapley, B.D., 2006. Satellite gravity measurements confirm accelerated melting of Greenland ice sheet. *Science*, 313(5795), 1958–1960.

Chu-Agor, M.L.; Muñoz-Carpena, R.; Kiker, G.; Emanuelsson, A., and Linkov, I., 2010. Global sensitivity and uncertainty analysis of SLAMM for the purpose of habitat vulnerability assessment and decision making. *Proceedings of the World Environmental and Water Resources Congress* (Providence, Rhode Island), pp. 4702–4709.

Church, J.A. and White, N.J., 2006. A 20th century acceleration in global sea-level rise. *Geophysical Research Letters*, 33, L01602, doi:10.1029/2005GL024826.

Church, J.; Gregory, J.M.; Huybrecht, P.; Kuhn, M.; Lambeck, K.; Nhuan, M.T.; Qin, D., and Woodworth, P.L., 2001. Changes in sea level. *In*: Houghton, J.T.; Ding, Y.; Griggs, D.J.; Noguer, M.; van der Linden, P.J.; Dai, X.; Maskell, K., and Johnson, C.A. (eds.), *Climate Change 2001: The Scientific Basis*. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press, pp. 639–694.

Clough, J.S.; Park, R.A., and Fuller, R., 2010. *SLAMM 6 Beta Technical Documentation*. Warren, Vermont: Warren Pinnacle Consulting, Inc., 48p.

Conner, W.H. and Day, J.W., Jr., 1988. Rising water levels in coastal Louisiana: Implications for two coastal forested wetland areas in Louisiana. *Journal of Coastal Research*, 4(4), 589–596.

Costanza, R.; Pérez-Maqueo, O.; Martinez, M.L.; Sutton, P.; Anderson, S.J., and Mulder, K., 2008. The value of coastal wetlands for hurricane protection. *Ambio*, 37(4), 241–248.

Couvillion, B.R.; Barras, J.A.; Steyer, G.D.; Sleavin, W.; Fischer, M.; Beck, H.; Trahan, N.; Griffin, B., and Heckman, D., 2011. Land Area Change in Coastal Louisiana from 1932 to 2010. *U.S. Geological Survey Scientific Investigations Map 3164*, scale 1:265,000, 1 sheet, 12 p. pamphlet.

Craft, C.; Clough, J.; Ehman, J.; Joye, S.; Park, R.; Pennings, S.; Guo, H., and Machmuller, M., 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. *Frontiers in Ecology and the Environment*, 7(2), 73–78.

Cunningham, R.; Gisclair, D., and Craig, J., 2002. The Louisiana Statewide LIDAR Project. URL: http://atlas.lsu.edu/central/la_LiDAR_project.pdf; accessed on June 28, 2012.

Day, J.W., Jr.; Britsch, L.D.; Hawes, S.R.; Shaffer, G.P.; Reed, D.J., and Cahoon, D., 2000. Pattern and process of land loss in the Mississippi Delta: A spatial and temporal analysis of wetland habitat change. *Estuaries*, 23(4), 425–438.

Day, J.W., Jr.; Ko, J.-Y.; Rybczyk, J.; Sabins, D.; Bean, R.; Berthelot, G.; Brantley, C.; Cardoch, L.; Conner, W.; Day, J.N.; Englande, A.J.; Feagley, S.; Hyfield, E.; Lane, R.; Lindsey, J.; Mitsch, W.J.; Reyes, E., and Twilley, R.R., 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: A review. *Ocean and Coastal Management*, 47(11-12), 671–691.

Day, J.W.; Cable, J.E.; Cowan, J.H., Jr.; DeLaune, R.; de Mutsert, K.; Fry, B.; Mashriqui, H.; Justic, D.; Kemp, P.; Lane, R.R.; Rick, J.; Rick, S.; Rozas, L.P.; Snedden, G.; Swenson, E.; Twilley, R.R., and Wissel, B., 2009. The impacts of pulsed reintroduction of river water on a Mississippi Delta coastal basin. *Journal of Coastal Research*, Special Issue No. 54, pp. 225–243.

Day, J.; Ibánez, C.; Scarton, F.; Pont, D.; Hansel, P., and Lane, R., 2011a. Sustainability of Mediterranean deltaic and lagoon wetlands with sea-level rise: The importance of river input. *Estuaries and Coasts*, 34(3), 483–493.

Day, J.W.; Kemp, G.P.; Reed, D.J.; Cahoon, D.R.; Boumans, R.M.; Suhayda, J.M., and Gambrell, R. 2011b. Vegetation death and rapid loss of surface elevation in two contrasting Mississippi Delta salt marshes: The role of sedimentation, autocompaction, and sea-level rise. *Ecological Engineering*, 37(2), 229–240.

DeLaune, R.D.; Nyman, J.A., and Patrick, W.H., 1994. Peat collapse, pending and wetland loss in a rapidly submerging coastal marsh. *Journal of Coastal Research*, 10(4), 1021–1030.

DeLaune, R.D.; Jugsujinda, J.; Peterson, G.W., and Patrick, W.H., Jr., 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana estuary. *Estuarine, Coastal, and Shelf Science*, 58(3), 653–662.

Donoghue, J.F., 2011. Sea level history of the northern Gulf of Mexico coast and sea level rise scenarios for the near future. *Climatic Change*, 107(1), 17–33.

Doyle, T.W.; Krauss, K.W.; Conner, W.H., and From, A.S., 2010. Predicting the retreat and migration of tidal forests along the northern Gulf of Mexico under sea-level rise. *Forest Ecology and Management*, 259(4), 770–777.

Eggler, W.A. and Moore, W.G., 1961. The vegetation of Lake Chicot, Louisiana, after eighteen years impoundment. *The Southwestern Naturalist*, 6(3-4), 175–183.

Engle, V.D., 2011. Estimating the provision of ecosystem services by Gulf of Mexico wetlands. *Wetlands*, 31(1), 179–193.

Faulkner, S.P.; Chambers, J.L.; Conner, W.H.; Keim, R.F.; Day, J.W.; Gardiner, E.S.; Hughes, M.S.; King, S.L.; McLeod, K.W.; Miller, C.A.; Nyman, J.A., and Shaffer, G.P., 2007. Conservation and use of coastal wetland forests in Louisiana. *In:* Conner, W.H.; Doyle, T.W., and Krauss, K.W. (eds.), *Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States*. New York: Springer, pp. 447–460.

Feagin, R.A.; Mukherjee, N.; Shanker, K.; Baird, A.H.; Cinner, J.; Kerr, A.M.; Koedam, N.; Sridhar, A.; Arthur, R.; Jayatissa, L.P.; Lo Seen, D.; Menon, M.; Rodriguez, S., and Dahdouh-Guebas, F., 2010. Shelter from the storm? Use and misuse of coastal vegetation bioshields for managing natural disasters. *Conservation Letters*, 3(1), 1–11.

FitzGerald, D.M.; Fenster, M.S.; Argow, B.A., and Buynevich, H.V., 2008. Coastal impacts due to sea-level rise. *Annual Review of Earth and Planetary Sciences*, 36, 601–647.

Galbraith, H.; Jones, R.; Park, R.; Clough, J.; Herrod-Julius, S.; Harrington, B., and Page, G., 2002. Global climate change and sea level rise: Potential losses of intertidal habitat for shorebirds. *Waterbirds*, 25(2), 173–183.

Gedan, K.B.; Kirwan, M.L.; Wolanski, E.; Barbier, E.B., and Silliman, B.R., 2010. The present and future role of coastal wetland vegetation in protecting shorelines: Answering recent challenges to the paradigm. *Climatic Change*, 106(1), 7–29.

Geselbracht, L.; Freeman, K.; Kelly, E.; Gordon, D.R., and Putz, F.E., 2011. Retrospective and prospective model simulations of sea level rise impacts on Gulf of Mexico coastal marshes and forests in Waccasassa Bay, Florida. *Climatic Change*, 107(1), 35–57.

Glick, P. and Clough, J., 2006. *An Unfavorable Tide: Global Warming, Coastal Habitats, and Sportfishing in Florida.* Reston, Virginia: National Wildlife Federation, Tallahassee, Florida: Florida Wildlife Federation, 56p.

Glick, P.; Clough, J., and Nunley, B., 2007. *Sea-Level Rise and Coastal Habitats in the Pacific Northwest: An Analysis for Puget Sound, Southwestern Washington, and Northwestern Oregon*. Seattle, Washington: National Wildlife Federation, 94p.

Glick, P.; Clough, J., and Nunley, B., 2008. *Sea-Level Rise and Coastal Habitats in the Chesapeake Bay Region: Technical Report*. Reston, Virginia: National Wildlife Federation, 121p.

Grinsted, A.; Moore, J.C., and Jevrejeva, S., 2009. Reconstructing sea level from paleo and projected temperatures 2000–2100 A.D. *Climate Dynamics*, 34(4), 461–472.

Harms, W.R.; Schreuder, H.T.; Hood, D.D., and Brown, C.L., 1980. The effects of flooding on the swamp forest in Lake Ocklawah, Florida. *Ecology*, 61(6), 1412–1421.

Hatton, R.S.; DeLaune, R.D., and Patrick, W.H., Jr., 1983. Sedimentation, accretion and subsidence in the marshes of Barataria Basin, Louisiana. *Limnology and Oceanography*, 28(3), 494–502.

Howes, N.C.; FitzGerald, D.M.; Hughes, Z.J.; Georgiou, I.Y.; Kulp, M.A.; Miner, M.D.; Smith, J.M., and Barras, J.A., 2010. Hurricane-induced failure of low salinity wetlands. *Proceedings of the National Academy of Sciences*, 107(32), 14014–14019.

IPCC (Intergovernmental Panel on Climate Change), 2007. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom: Cambridge University Press, 996p.

Ivins, E.R.; Dokka, R.K., and Blom, R.G., 2007. Post-glacial sediment load and subsidence in coastal Louisiana. *Geophysical Research Letters*, 34, L16303, doi: 10.1029/2007GL030003.

Jevrejeva, S.; Moore, J.C., and Grinsted, A., 2010. How will sea level respond to changes in natural and anthropogenic forcings by 2100? *Geophysical Research Letters*, 37, L07703, doi:10.1029/2010GL042947.

Karl, T.R.; Melillo, J.M., and Peterson, T.C. (eds.), 2009. *Global Climate Change Impacts in the United States*. A report of the U.S. Global Change Research Program (USGCRP). Cambridge, United Kingdom: Cambridge University Press, 196p.

Keim, R.F.; Chambers, J.L.; Hughes, M.S.; Nyman, J.A.; Miller, C.A.; Amos, J.B.; Conner, W.H.; Day, J.W., Jr.; Faulkner, S.P.; Gardiner, E.S.; King, S.L.; McLeod, K.W., and Shaffer, G.P., 2006. Ecological consequences of changing hydrological conditions in wetland forests of coastal Louisiana. *In:* Xu, Y.J. and Singh, V.P. (eds.), *Coastal Environment and Water Quality*. Highlands Ranch, Colorado: Water Resources Publications, pp. 383–396.

Kim, W.; Mohrig, D.; Twilley, R.; Paola, C., and Parker, G., 2009. Is it feasible to build new land in the Mississippi River Delta? *EOS, Transactions American Geophysical Union* 90(42), 373–374.

Kirwan, M.L. and Guntenspergen, G.R., 2010. Influence of tidal range on the stability of coastal marshland. *Journal of Geophysical Research*, 115, doi:10.1029/2009JF001400.

Kirwan, M.L. and Murray, A.B., 2007. A coupled geomorphic and ecological model of tidal marsh evolution. *Proceedings of the National Academy of Sciences*, 104(15), 6118–6122.

Kirwan, M.L.; Guntenspergen, G.R.; D'Alpoas, A.; Morris, J.T.; Mudd, S.M., and Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. *Geophysical Research Letters* 37, doi:10.1029/2010GL045489.

Lane, R.R.; Day, J.W.; Marx, B.; Reves, E., and Kemp, G.P., 2002. Seasonal and spatial water quality changes in the outflow plume of the Atchafalaya River, Louisiana, U.S.A. *Estuaries and Coasts*, 25(1), 30–42.

Lane, R.R.; Day, J.W., and Day, J.N., 2006. Wetland surface elevation, vertical accretion, and subsidence at three Louisiana estuaries receiving diverted Mississippi River water. *Wetlands*, 26(4), 1130–1142.

Lang, C.-Y., 2000. Kriging Interpolation. URL: http://www.nbb.cornell.edu/neurobio/land/oldstudentprojects/cs490-94to95/clang/kriging.html; accessed on January 11, 2011.

Langley, J.A.; McKee, K.L.; Cahoon, D.R.; Cherry, J.A., and Megonigal, J.P., 2009. Elevated CO_2 stimulates marsh elevation gain, counterbalancing sea-level rise. *Proceedings of the National Academy of Sciences*, 106(15), 6182–6186.

Lee, J.K.; Park, R.A., and Mausel, P.W., 1992. Application of geoprocessing and simulation modeling to estimate impacts of sea-level rise on the northeast coast of Florida. *Photogrammetric Engineering and Remote Sensing*, 58(11), 1579–1586.

Louisiana Department of Natural Resources, 2011. Louisiana Coastal Facts. Office of Coastal Management. Coastal Protection and Restoration Authority of Louisiana. URL: http://www.ocpr.louisiana.gov/coastalfacts.asp; accessed on June 28, 2012.

Martin, J.F.; Reyes, E.; Kemp, G.P.; Mashriqui, H., and Day, J.W., Jr., 2002. Landscape modeling of the Mississippi Delta. *BioScience*, 52(4), 357–365.

Martin, S.B.; Hitch, A.T.; Purcell, K.M.; Klerks, P.L., and Leberg, P.L., 2009. Life history variation along a salinity gradient in coastal marshes. *Aquatic Biology*, 8(1), 15–28.

Mcleod, E.; Poulter, B.; Hinkel, J.; Reyes, E., and Salm, R., 2010. Sea-level rise impact models and environmental conservation: A review of models and their applications. *Ocean and Coastal Management*, 53(9), 507–517.

Meckel, T.A., 2008. An attempt to reconcile subsidence rates determined from various techniques in southern Louisiana. *Quaternary Science Reviews*, 27(15-16), 1517–1522.

Merrifield, M.A.; Merrifield, S.T., and Mitchum, G.T., 2009. An anomalous recent acceleration of global sea level rise. *Journal of Climate*, 22(21), 5772–5781.

Mitsch, W.J. and Gosselink, J.G., 2000. *Wetlands*. New York: Wiley, 920p.

Morris, J.T., 2006. Competition among marsh macrophytes by means of geomorphological displacement in the intertidal zone. *Estuarine, Coastal and Shelf Science*, 69(3-4), 395–402.

Morris, J.T.; Sundareshwar, P.V.; Nietch, C.T.; Kjerfve, B., and Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. *Ecology*, 83(10), 2869–2877.

Morton, R.A. and Barras, J.A., 2011. Hurricane impacts on coastal wetlands: A half-century record of storm-generated features from southern Louisiana. *Journal of Coastal Research*, 27(6A), 27–43.

Morton, R.A.; Bernier, J.C.; Barras, J.A., and Ferina, N.F., 2005. Rapid Subsidence and Historical Wetland Loss in the Mississippi Delta Plain: Likely Causes and Future Implications. *U. S. Geological Survey Open-File Report 2005-1216*, 116p.

Morton, R.A.; Bernier, J.S.; Kelso, K.W., and Barras, J.A., 2010. Quantifying large-scale historical formation of accommodation in the Mississippi Delta. *Earth Surface Processes and Landforms*, 35(14), 1625–1641.

Mudd, S.M.; Howell, S.M., and Morris, J.T., 2009. Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. *Estuarine and Coastal Shelf Science*, 82(3), 377–389.

Nakicenovic, N.; Alcamo, J.; Davis, G.; deVries, B.; Fenhann, J.; Gaffin, S.; Gregory, K.; Grübler, A.; Jong, T.Y.; Kram, T.; Lebra La Rovere, E.; Michaelis, L.; Mori, S.; Morita, T.; Pepper, W.; Pitcher, H.; Price, L.; Riahi, K.; Roehrl, A.; Rogner, H.-H.; Sankovski A.; Schlesinger, M.; Shukla, P.; Smith, S.; Swart, R.; von Rooijen, S.; Victor, N., and Dadi, Z., 2000. *Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios*. Cambridge, United Kingdom: Cambridge University Press, 599p.

NOAA, 2009. Sea Level Variations of the United States 1854–2006. Silver Spring, Maryland: National Oceanic and Atmospheric Administration, *Technical Report NOS CO-OPS 053*. xiv, 192p. URL: http://www.co-ops.nos.noaa.gov/publications/Tech_rpt_53.pdf; accessed on January 3, 2013.

NOAA, 2010. Annual Commercial Landing Statistics. National Oceanic and Atmostpheric Administration. URL: http://www.st.nmfs.noaa.gov/st1/commercial/landings/annual_landings.html; accessed on August 16, 2012.

NOAA, 2012a. NOAA Tides & Currents. URL: http://tidesandcurrents.noaa.gov/; accessed on January 4, 2013.

NOAA, 2012b. Estimation of Vertical Uncertainties in VDATUM–Last Revised August 2012. URL: ftp://tidepool.nos.noaa.gov/pub/outgoing/HPT/ORR_Inundation_delivery/Procedures_and_SOPs/VDatum_%20Estimation_of_Vertical_Uncertainties_in_VDatum_Last_revised_%20August%202012.pdf; accessed on January 10, 2013.

NRC (National Research Council). 2006. *Drawing Louisiana's New Map: Addressing Land Loss in Coastal Louisiana*. Washington, D.C.: The National Academies Press, 190p.

Nyman, J.A.; DeLaune, R.D., and Patrick, W.H., 1990. Wetland soil formation in the rapidly subsiding Mississippi River Deltaic Plain: Mineral and organic matter relationships. *Estuarine, Coastal, and Shelf Science*, 31(1), 57–69.

Nyman, J.A.; DeLaune, R.N.; Roberts, H.H., and Patrick, W.H., Jr., 1993. Relationship between vegetation and soil formation in a rapidly submerging coastal marsh. *Marine Ecology Progress Series*, 96(3), 269–279.

Nyman, J.A.; Walters, R.J.; Delaune, R.D., and Patrick, W.H., Jr., 2006. Marsh vertical accretion via vegetative growth. *Estuarine, Coastal, and Shelf Science*, 69(3-4), 370–380.

Otto-Bliesner, B.L.; Marshall, S.J.; Overpeck, J.T.; Miller, G.H.; Hu, A., and CAPE Last Interglacial Project members, 2006. Simulating Arctic climate warmth and icefield retreat in the last interglaciation. *Science*, 311(5768), 1751–1753.

Overpeck, J.T. and Weiss, J.L., 2009. Projections of future sea level becoming more dire. *Proceedings of the National Academy of Sciences*, 106(51), 21461–21462.

Overpeck, J.T.; Otto-Bliesner, B.L.; Miller, G.H.; Muhs, D.R.; Alley, R.B., and Kiehl, J.T., 2006. Paleoclimatic evidence for future ice-sheet instability and rapid sea-level rise. *Science*, 311(5768), 1747–1750.

Park, R.A.; Trehan, M.S.; Mausel, P.W., and Howe, R.C., 1989. The effects of sea level rise on U.S. coastal wetlands. *In:* Smith, J.B. and Tirpak, D.A. (eds.), The Potential Effects of Global Climate Change on the United States. Appendix B: Sea Level Rise. Washington, D.C.: *U.S. Environmental Protection Agency, EPA 230-05-89-052*, pp. 1–1 to 1–55.

Park, R.A.; Lee, J.K., and Canning, D., 1993. Potential effects of sea level rise on Puget Sound wetlands. *Geocarto International*, 8(4), 99–110.

Pascual, P.; Steiber, N., and Sunderland, E., 2003. *Draft Guidance on the Development, Evaluation, and Application of Regulatory Environmental Models*. Washington, D.C.: National Council for Regulatory Environmental Modeling (CREM), Office of Science Policy, Office of Research and Development, U.S. Environmental Protection Agency, 60p.

Pfeffer, W.T.; Harper, J.T., and O'Neel, S., 2008. Kinematic constraints on glacier contributions to 21st century sea-level rise. *Science*, 321(5894), 1340–1343.

Photo Science, Inc., 2009. *2008 Digital Orthophoto Quarter Quadrangles (DOQQs) for the coastal region of Louisiana.* Produced by Photo Science, Inc. for the U.S. Geological Survey. Lafayette, Louisiana: National Wetlands Research Center.

Rahmstorf, S., 2007. A semiempirical approach to projecting future sea-level rise. *Science*, 315(5810), 368–370.

Rahmstorf, S., 2010. A new view on sea level rise. *Nature Reports Climate Change*, 4, 44–45, doi:10.1083/climate.2010.29.

Raupach, M.R.; Marland, G.; Ciais, P.; Le Quéré, C.; Canadell, J.G.; Klepper, G., and Field, C.B., 2007. Global and regional drivers of accelerating CO_2 emissions. *Proceedings of the National Academy of Sciences*, 104(24), 10288–10293.

Reed, D.J. and Yuill, B., 2009. *Understanding Subsidence in Coastal Louisiana*. New Orleans, Louisiana: Pontchartrain Institute for Environmental Sciences, University of New Orleans, 69p.

Reyes, E.; White, M.L.; Martin, J.F.; Kemp, G.P.; Day, J.W., and

Aravamuthan, V., 2000. Landscape modeling of coastal habitat change in the Mississippi Delta. *Ecology*, 81(8), 2331–2349.

Rignot, E. and Kanagaratnam, P., 2006. Changes in the velocity structure of the Greenland ice sheet. *Science*, 311(5763), 986–990.

Rignot, E.; Velicogna, I.; van den Broeke, M.R.; Monaghan, A., and Lenaerts, J., 2011. Acceleration of the contribution of the Greenland and Antarctic ice sheets to sea level rise. *Geophysical Research Letters*, 38, L05503, doi:10.1029/2011GL046583.

Ritchie, B.D.; Gawthorpe, R.L., and Hardy, S., 2004. Three-dimensional numerical modeling of deltaic depositional sequences 1: Influence of the rate and magnitude of sea level change. *Journal of Sedimentary Research*, 74(2), 203–220.

Roberts, H.H.; Coleman, J.M.; Bentley, S.J., and Walker, N., 2003. An embryonic major delta lobe: A new generation of delta studies in the Atchafalaya–Wax Lake Delta system. *Gulf Coast Association of Geological Societies Transactions*, 53, 690–703.

RuleQuest Research, 2012. Cubist 2.07 GPL Edition. URL: http://www.rulequest.com/cubist-info.html; accessed on October 9, 2012.

Sasser, C.E.; Gosselink, J.G.; Swenson, E.M.; Swarzenski, C.M., and Leibowitz, N.C., 1996. Vegetation, substrate, and hydrology in floating marshes in the Mississippi River Delta plain wetlands, U.S.A. *Plant Ecology*, 122(2), 129–142.

Sasser, C.E.; Materne, M.D.; Visser, J.M.; Holm, G.O., Jr., and Evers, E., 2007. Restoring freshwater floating marsh in coastal Louisiana. *Louisiana Agriculture*, 50(2), 14–17.

Sasser, C.E.; Visser, J.M.; Mouton, E.; Linscombe, J., and Hartley, S.B., 2008. Vegetation Types in Coastal Louisiana in 2007. *U.S. Geological Survey Open-File Report 2008-1224*, scale 1:550,000, 1 sheet.

Sasser, C.E.; Gosselink, J.G.; Holm, G.O., Jr., and Visser, J.M., 2009. Tidal freshwater wetlands of the Mississippi River deltas. *In*: Barendregt, A.; Whigham, D., and Baldwin, A. (eds.), *Tidal Freshwater Wetlands*. Weikersheim: Margraf Publishers, 320p.

Shaffer, G.P.; Wood, W.B.; Hoeppner, S.S.; Perkins, T.E.; Zoller, J., and Kandalepas, D., 2009. Degradation of baldcypress-water tupelo swamp to marsh and open water in southeastern Louisiana, U.S.A.: An irreversible trajectory? *Journal of Coastal Research*, Special Issue No. 54, pp. 152–165.

Shinkle, K.D. and Dokka, R.K., 2004. *Rates of Vertical Displacement at Benchmarks in the Lower Mississippi Valley and the Northern Gulf Coast*. Silver Spring, Maryland: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, 147p.

Shirley, L.J. and Battaglia, L.L., 2008. Projecting fine resolution land cover dynamics for a rapidly changing terrestrial-aquatic transition in Terrebonne Basin, Louisiana, U.S.A. *Journal of Coastal Research*, 24(6), 1545–1554.

Smith, J.M.; Cialone, M.A.; Wamsley, T.V., and McAlpin, T.O., 2010. Potential impact of sea level rise on coastal surges in southeast Louisiana. *Ocean Engineering*, 37(1), 37–47.

Solomon, S.; Plattner, G.-K.; Knutti, R., and Friedlingstein, P., 2009. Irreversible climate change due to carbon dioxide emissions. *Proceedings of the National Academy of Sciences*, 106(6), 1704–1709.

Stanton, E.A. and Ackerman, F., 2007. *Florida and Climate Change: The Costs of Inaction*. Medford, Massachusetts: Tufts University Global Development and Environment Institute, 104p.

Thompson, D.A.; Shaffer, G.P., and McCorquodale, J.A., 2002. A potential interaction between sea-level rise and global warming: Implications for coastal stability on the Mississippi River Deltaic Plain. *Global Planetary Change*, 32(1), 49–59.

Titus, J.G.; Park, R.A.; Leatherman, S.P.; Weggel, J.R.; Greene, M.S.; Mausel, P.W.; Trehan, M.S.; Brown, S.; Grant, C., and Yohe, G.W., 1991. Greenhouse effect and sea level rise: The cost of holding back the sea. *Coastal Management*, 19(2), 171–204.

Traill, L.W.; Perhans, K.; Lovelock, C.E.; Prohaska, A.; McFallan, S.; Rhodes, J.R., and Wilson, K.A., 2011. Managing for change: Wetland transitions under sea-level rise and outcomes for threatened species. *Biodiversity Research*, 17(6), 1225–1233. doi:10.1111/j.1472-4642.2011.00807.x.

Twilley, R.R., 2007. *Coastal Wetlands and Global Climate Change: Gulf Coast Wetland Sustainability in a Changing Climate*. Arlington, Virginia: Pew Center on Global Climate, 18p.

Twilley, R.R.; Couvillion, B.R.; Hossain, I.; Kaiser, C.; Owens, A.B.; Steyer, G.D., and Visser, J.M., 2008. Coastal Louisiana Ecosystem Assessment and Restoration Program: The role of ecosystem forecasting in evaluating restoration planning in the Mississippi River deltaic plain. *American Fisheries Society Symposium*, 64, 29–46.

USACE (U.S. Army Corps of Engineers), 2003. *Louisiana Coastal Area, Louisiana–Ecosystem Restoration: Comprehensive Coastwide Ecosystem Restoration Study* (draft LCA Comprehensive Study). New Orleans, Louisiana: U.S. Army Corps of Engineers.

USACE (U.S. Army Corps of Engineers), 2004. *Louisiana Coastal Area, Louisiana–Ecosystem Restoration Study*. New Orleans District, Louisiana, vol.1, 506p.

USFWS (U.S. Fish and Wildlife Service), 1988. National Wetlands Inventory. Washington, D.C.: U.S. Department of the Interior, Fish and Wildlife Service. URL: http://www.fws.gov/wetlands/; accessed on June 29, 2012.

USGS, 2012a. EarthExplorer. URL: http://earthexplorer.usgs.gov; accessed on January 4, 2012.

USGS, 2012b. USGS Water Data for the Nation. URL: http://waterdata.usgs.gov/nwis/; accessed on January 3, 2013.

Vermeer, M. and Rahmstorf, S., 2009. Global sea level linked to global temperature. *Proceedings of the National Academy of Sciences*, 106(51), 21527–21532.

Watershed Concepts, 2009. *Louisiana Phase 1,2,3 LIDAR Project, Quality Control Evaluation of LIDAR Data For Louisiana*. Louisiana Federal Emergency Management Agency (FEMA), Contract: EMT–2002–CO–0048.

Weiss, J.L.; Overpeck, J.T., and Strauss, B., 2011. Implications of recent sea level rise science for low-elevation areas in coastal cities of the conterminous U.S.A. *Climatic Change*, 105(3–4), 635–645.

Wicker, K.M., 1980. The Mississippi Deltaic Plain ecological characterization: A habitat mapping study. A user's guide to the habitat maps. Washington, D.C.: U.S. Fish and Wildlife

Service, Office of Biological Services, *FWS/OBS 79/07*, 91p.
Wicker, K.M., 1981. *Chenier Plain region ecological characterization: A habitat mapping study. A user's guide to the habitat maps*. Baton Rouge, Louisiana: Louisiana Department of Natural Resources, 102p.
Wilhite, L.P. and Toliver, J.R., 1990. *Taxodium distichum* (L.) Rich. Baldcypress. *In*: Burns, R.M. and Honkala, B.H. (eds.), *Silvics of North America*, 1, 563–572.
Woolpert, Inc., 2011. Louisiana LiDAR Regions 1 and 2. Charleston, South Carolina: *U.S. Geological Survey Contract No: G10PC00057*, Task Order No: G10PD02781.
Yuill, B.; Lavoie, D., and Reed, D.J., 2009. Understanding subsidence processes in coastal Louisiana. *Journal of Coastal Research*, Special Issue No. 54, pp. 23–36.