Sediment and Plant Dynamics in a Degrading Coastal Louisiana Landscape

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SEDIMENT AND PLANT DYNAMICS IN A DEGRADING COASTAL LOUISIANA LANDSCAPE

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by
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This work is dedicated to my wife Rachelle G. Suir and my daughter Calleigh Grace Suir whose love, patience, and support made this dissertation possible. I would also like to dedicate this work to my parents, Doris T. Suir and the late Glenn R. Suir, for their eternal support and selfless devotion to my education.

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ABSTRACT

Alterations to Louisiana's river systems and local hydrology have resulted in reduced freshwater, sediment, and nutrient inputs into wetland landscapes, causing significant negative impacts on marsh productivity and stability. To combat these losses many restoration projects have been constructed or planned throughout coastal Louisiana. Typical goals of wetland restoration efforts are to conserve, create, or enhance wetland form, and to achieve wetland function that approaches natural conditions. Failure to adequately maintain wetland elevation and hydrology can have serious implications on sedimentation and vegetation processes, which significantly reduces the likelihood of reaching structural and functional targets. Measures of wetland condition have been used to monitor and assess project performance, resilience, and adaptive management needs. This study assessed the use of remotely sensed and *in situ* data, in addition to landscape metrics (i.e., marsh area, edge density, and aggregation index) and vegetative indices (i.e., vegetation cover, normalized difference vegetative index, and floristic quality index) to evaluate changes and trends in restored wetland condition, function, and resilience, and compare those to naturally occurring reference wetlands. Results show that restoration measures (i.e., hydrologic restoration, wetland restoration, and beneficial use of dredge material) significantly increased wetland function (i.e., vegetation productivity, carbon sequestration, floristic quality), stability (i.e., increased marsh area, reduced loss rates, and increased spatial integrity), and resilience to disturbance events. Though many structural and functional measures (i.e., vegetation and landscape indices) of restored wetlands rapidly achieved equivalency to reference wetlands (approximately 3 to 5 years after construction), others, like some fundamental soil functions (i.e., carbon accumulation) required several decades to reach equivalency. These results demonstrated the importance of river connectivity and sedimentation for wetland productivity and overall spatial integrity. These studies show remotely sensing data and applications can significantly supplement traditional methods and provide critical

knowledge elements for more efficient inventorying and monitoring of wetland resources, forecasting of resource condition and stability, and formulating adaptive management strategies.

CHAPTER 1 – SEDIMENT IN A DEGRADING ECOSYSTEM: A SYNTHESIS

INTRODUCTION

Background

Since the 1930s, Louisiana's coastal landscapes have experienced dramatic loss of wetlands (Day *et al*. 2000; Couvillion *et al*. 2011; Suir *et al*. 2014). Between 1932 and 2010, approximately 4,880 square kilometers (km^2) of Louisiana's 19,544 km^2 of wetlands were lost, and an additional 842 km^2 are expected to be lost by the year 2050 (Barras *et al*. 2004; Couvillion *et al*. 2011). These wetland losses are due in large part to a complex interaction of spatial and temporal factors, including reduced riverine inputs, flood control measures, altered wetland hydrology, saltwater intrusion, wave erosion, and reduced river sediment load (Day *et al*. 2000). These factors also have significant implications on long-term wetland resilience and function, as well as ecosystem goods and services (Craft *et al*. 2009). At risk are the wetland-derived benefits that range from regulating services (i.e., floods, drought, and wetland degradation), supporting services (i.e., soil formation and nutrient cycling), provisioning services (i.e., food and freshwater); cultural services (i.e., recreational and aesthetics), to ecosystem services (i.e., high biological productivity and critical habitat) (Millennium Ecosystem Assessment 2003; United States Army Corps of Engineers [USACE] 2013).

The stability of coastal ecosystems and wetland landscapes remains a function of the balance between sedimentation, subsidence, sea-level rise, and episodic events (Mitsch and Gosselink 2000; Steyer *et al*. 2007; Barras 2009). In Louisiana, channel training and flood protection measures have offset this balance by depriving adjacent landscapes of the sediment that was historically provided during seasonal overbank flooding and other episodic events (i.e., storm surge). Even where hydrologic connectivity between rivers and wetlands remain, the potential for land building through naturally delivered sediment is limited due to the Mississippi River's progressively declining sediment loads (Thorne *et al*. 2008; Allison *et al*. 2012). As a consequence, ecosystem uses of dredged sediments present increasingly valuable applications for re-introduction of sediments into degrading and subsiding landscapes. Though previous studies have evaluated wetland loss, goods, and services, few have evaluated and compared wetland structure, function, and resilience as related to sediment delivery (naturally or mechanically).

Mississippi River Discharge and Sediment Yield

The Mississippi River basin, which consists of thirty-one States (41% of the continental United States) and two Canadian Provinces, drains approximately 3.1 million km^2 (Figure 1.1) (USACE 2004; National Park Service 2018). This drainage basin supplies the Mississippi River with large volumes of sediment and water, ranking it eighth worldwide in average annual suspended load (more than 200,000,000 metric tons), ninth worldwide in discharge (approximately 16,000 cubic meters per second [m^3 s^{-1}]), and first in North America in sediment load and discharge (Holeman 1968; Thorne *et al*. 2008). A sediment budget of the lower Mississippi River shows prior to human modifications the upper portion of the Mississippi River basin acted as the dominant source of sediment (bank caving was single primary source), and the lower River basin operated as a sediment sink (Kesel *et al*. 1992). The sediment budget showed a downriver fining of sediments, with a decrease in mean sediment diameter from approximately 3.6 mm near Cairo, Illinois to 0.1 mm at the mouth of the river (Kesel *et al*. 1992). This fining in the lower Mississippi River system is largely caused by long- and short-term storage of sediment, accounting for 24 and 38 percent of the total sediment load, respectively (Kesel *et al*. 1992).

Sediment Transport and Flux

The majority of Mississippi River system sediments that are not confined to long-term storage are eventually transported to the Gulf of Mexico where the coarser fragments are deposited near the shore and the finer particles at a distance (Brady and Weil 1999). The transport and deposition of

Mississippi River sediments at low elevations create intertidal flats that ultimately transition to vegetated deltas. Over time, delta formation and land building constrains river flow and triggers a shift in river course to a path of least resistance.

Figure 1.1. Primary rivers and drainage area of the Mississippi River basin. Courtesy of National Park Service (2018).

For millennia the unrestrained Mississippi River changed courses, building multiple delta lobes that collectively resulted in the formation of approximately 1.6 million hectares of coastal Louisiana wetlands (USACE 2004). Figure 1.2 shows the approximate location and extent of six historical inactive delta lobes and two active lobes (Balize and Atchafalaya) in south Louisiana. For many thousands of years, the Mississippi River and distributaries have delivered sediments that were vital for delta formation. This delta building process resulted in the creation of Louisiana's Deltaic Plain.

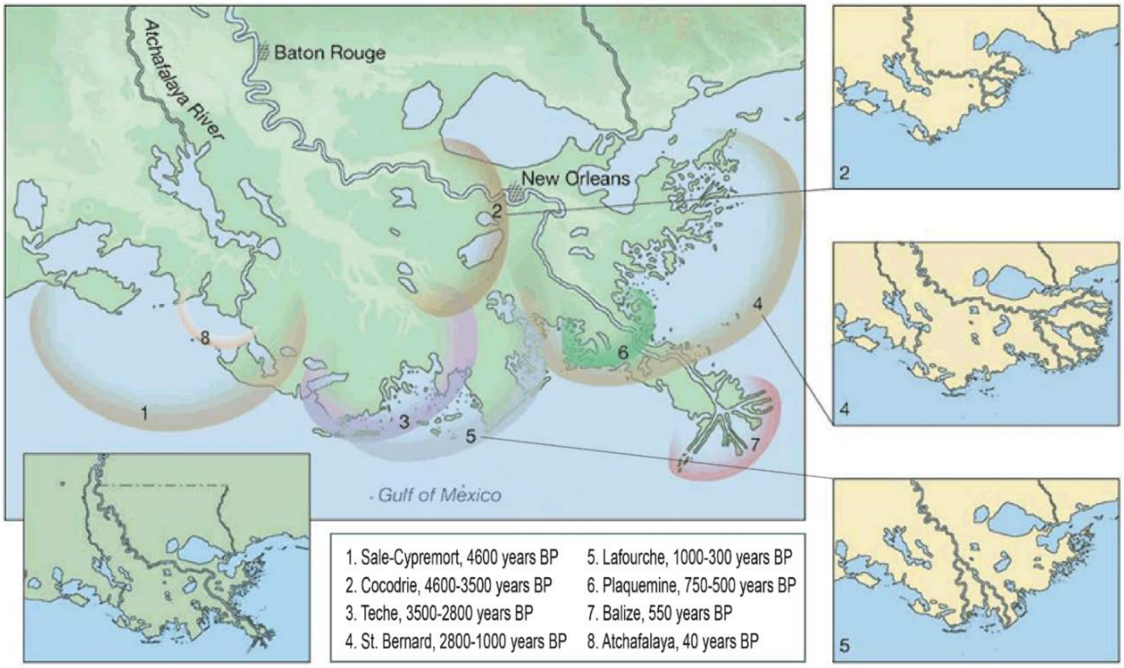

Figure 1.2. Chronology of Mississippi River delta formation. Modified from Tasa Graphic Arts, Inc. (2002).

In addition to delta formation, bars and beaches along the Louisiana western coast have been formed by the longshore transport and marine reworking of terrigenous material (Figure 1.3) (Selley 1996; McBride *et al*. 2007). These thin sand- and shell-rich “cheniers” make up a complex geomorphic area comprised of transgressive (traditional cheniers), regressive (beach ridges), and laterally accreting (spits) ridges (McBride *et al*. 2007). Though the Chenier Plain is largely formed by Mississippi River sediment, the evolution and response of ridges are also driven by the local reworking of sediment, and the trapping and stabilization of sediment at tidal entrances (McBride *et al*. 2007). The morphology of the Chenier (based on the French word for “oak tree”) Plain is the primary result of longshore transport and from two major processes: an intensive interaction between high-energy forces related to waves, tides, wind, and currents; and the constantly changing sea level over geologic time (Kennett 1981).

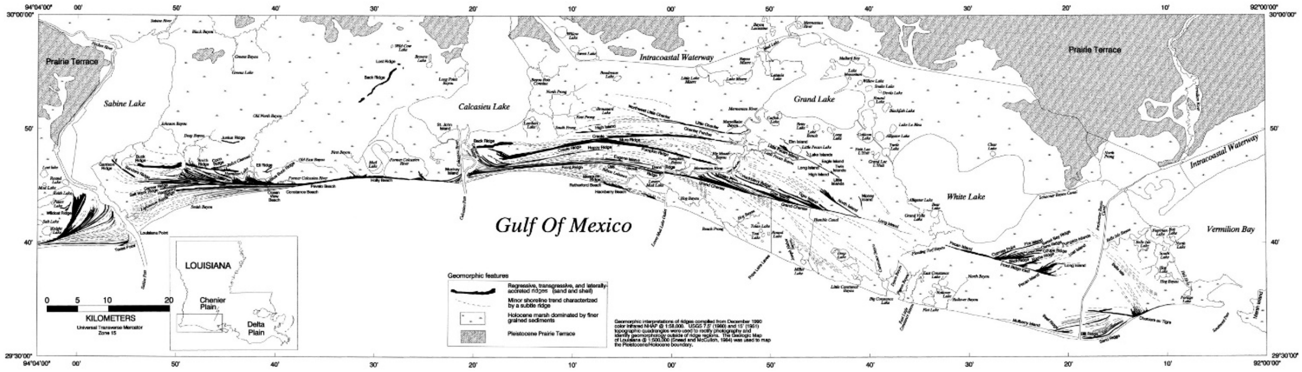

Figure 1.3. Geomorphology of the Louisiana Chenier Plain. Courtesy of McBride *et al*. (2007).

Wetland Formation and Function

Coastal wetland development and function is typically regulated by a variety of biotic and abiotic factors. Wetlands in the Mississippi River Delta were established primarily via Mississippi River sediment deposition over the past 6,000 years (Nyman *et al*. 1990). Across their lifespan, these wetlands undergo three phases of growth and abandonment, (1) rapid growth with increasing or stable river discharge, (2) moderate stability with waning discharge, and (3) abandonment and subsequent subsidence-dominated vertical development (Roberts 1997). Initial and rapid sediment infilling increases substrate elevation, promoting vegetation and wetland formation, which further increases wetland elevation through increasing mineral sediment deposition (via sediment trapping) and plant organic matter accumulation (Cahoon *et al*. 2011).

Wetlands contribute to a number of significant goods and services, including flood control, water filtering and storage, food supply, shoreline and storm protection, cultural value, recreation, critical habitat, and regulation of major chemical, physical, and biological processes (Pratolongo *et al*. 2009). Figure 1.4 depicts the relative contributions of the ecosystem goods and services provided by wetland sediments along a land-sea gradient (Wall *et al*. 2004). Specifically, sediments in coastal ecosystems contribute to the regulation of major biogeochemical cycles; the bioremediation of wastes and pollutants; erosion control; the retention and delivery of nutrients to plants and algae; the generation and renewal of soil structure and fertility; the modification of the hydrological cycle; the regulation of gases; the modification of anthropogenically driven global change; plant production; and landscape heterogeneity and stability (Wall *et al*. 2004).

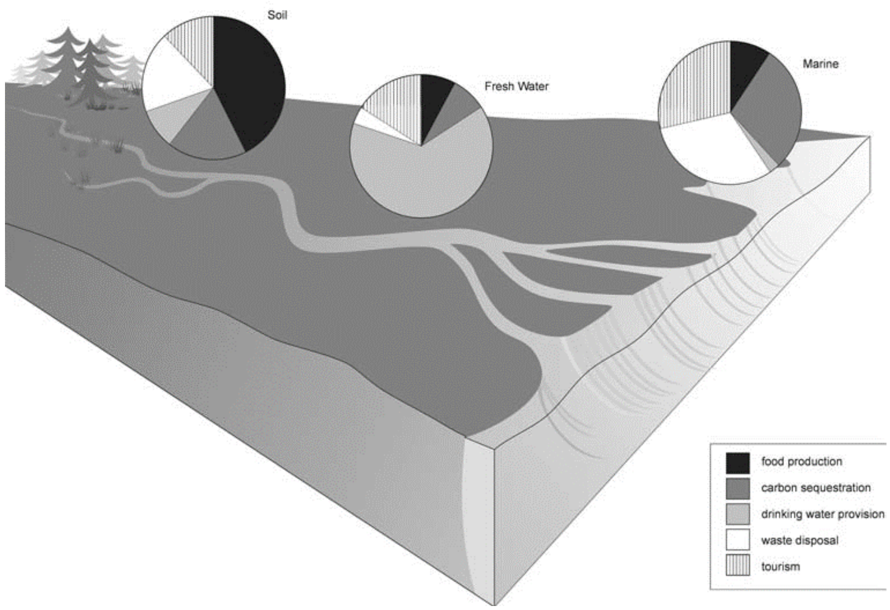

Figure 1.4. Importance of terrestrial, freshwater, and marine sediments for providing goods and services. Courtesy of Wall *et al*. (2004).

Wetland Loss

Major episodic flooding of colonized and developed lands along the Mississippi River system has resulted in the construction of levees and other flood protection measures. The purpose of these measures are to reduce flooding events by training the rivers flow onto and over the continental shelf. Though these measures have successfully reduced the number of flooding events, the levees have also significantly reduced riverine connectivity to coastal marshes, thereby depriving many of these wetland landscapes of vital freshwater and sediment inputs (Mississippi River Delta Science and Engineering Special Team [MRDSEST] 2012). These flood protection measures have also resulted in the rapid decline in sediment load during the last century (Figure 1.5). These sediment reductions were primarily driven by the flushing of previously stored sediments (due to channelization), increased sediment trapping by river training structures, and reduced erosion due to bank protection structures and soil conservation measures in the upper Mississippi River system (Meade *et al*. 2010). During this period the Mississippi River basin shifted from a transport-limited to a supply-limited system (Meade *et al*. 2010). These declining suspended loads (80 percent decrease since the middle of the nineteenth century), in addition to the flood protection measures, have resulted in significant reductions in the unconfined and overbank distributions of Mississippi River sediment (Kesel 1988, 1989).

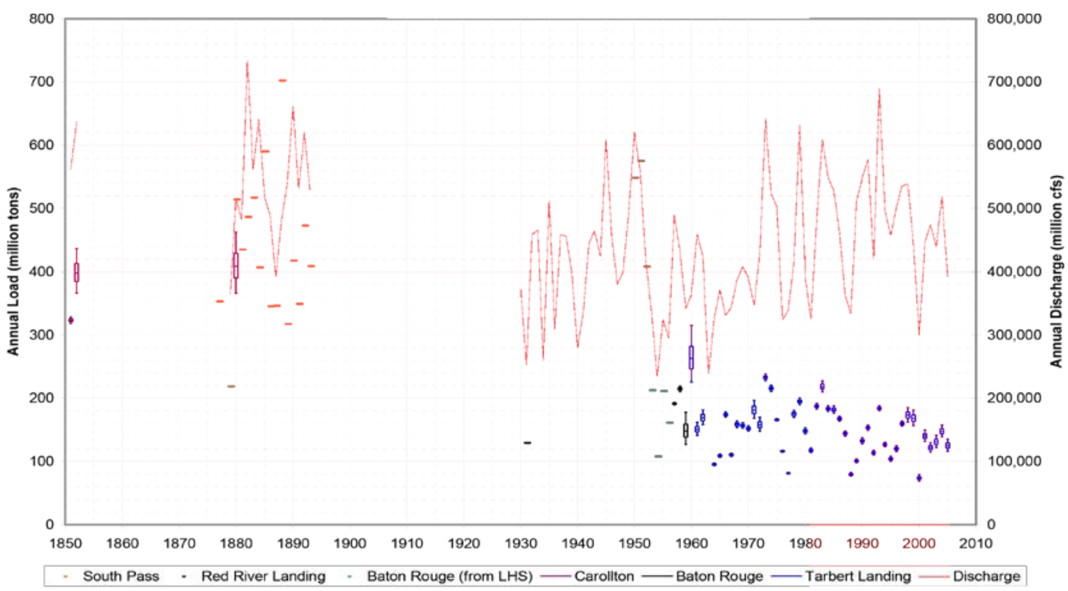

Figure 1.5. Long-term changes in annual load and discharge in the Mississippi River from multiple stations. Modified from Thorne *et al*. (2008).

The stability and productivity of wetland landscapes are largely driven by salinity, subsidence, climate change, episodic events, and nutrient and sediment availability (Mulholland *et al*. 1997; DeLaune *et al*. 2005; Steyer *et al*. 2008; Barras 2009). Sediments are critical inputs that are necessary for maintaining wetland platform elevation and countering subsidence and other erosional processes. Therefore, eliminating or reducing these key riverine inputs (i.e., reduced sediment load and delivery) from wetland landscapes can have significant long-term effects on marsh productivity and stability. The recent reduction in Mississippi River sediment load and distribution is a major contributor to the approximately 4,880 km^2 of wetlands that have been lost since the Mississippi River system levees were reinforced after the major flood of 1927 (Figure 1.6) (Couvillion *et al*. 2011).

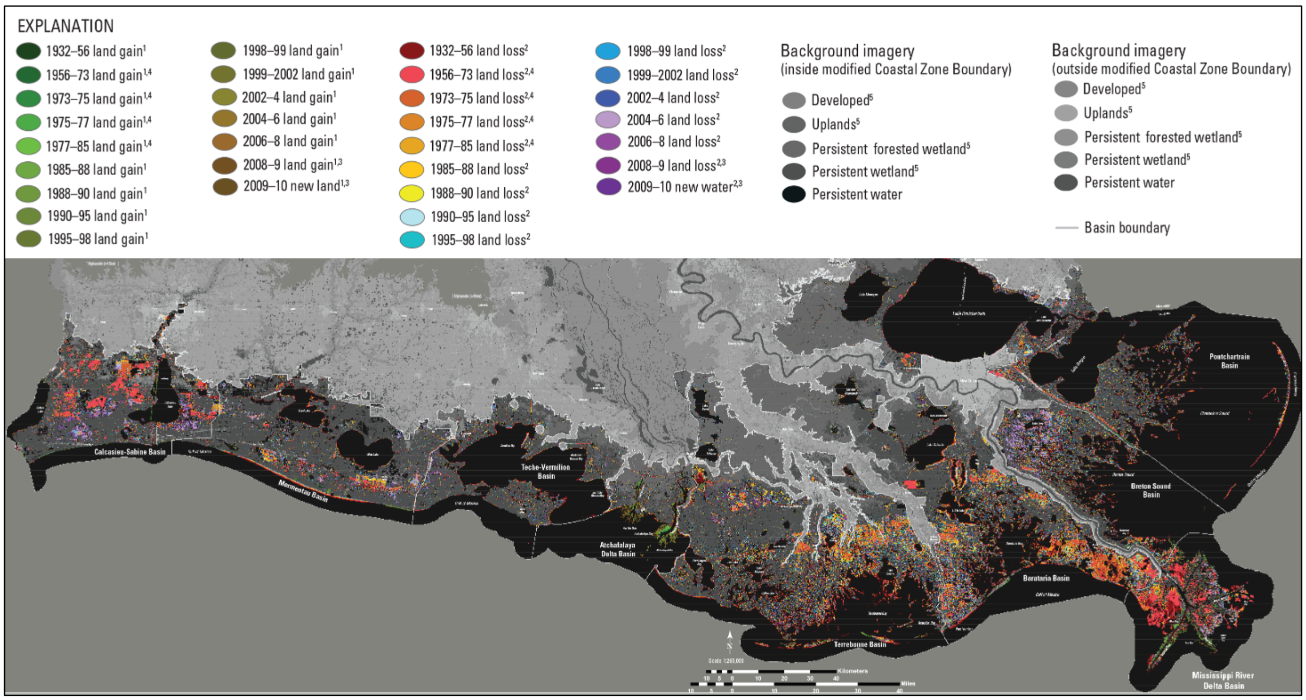

Figure 1.6. Land area change in coastal Louisiana from 1932 to 2010. Modified from Couvillion *et al*. (2008).

Wetland Restoration

In the 1970s, small- and basin-scale wetland management and restoration plans were initiated to counteract Louisiana's wetland losses (Gagliano *et al*. 1973). However, not until the authorization of the Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA) in 1990 were the large efforts to restore Louisiana's wetlands initiated (Williams *et al*. 1997). Two of the primary restoration techniques promoted and utilized in coastal Louisiana have been marsh creation using dredged material and reestablishing river connectivity to wetland landscapes though freshwater and sediment diversions. Though the ecological benefits of these restoration techniques have long been assumed (and are often the target of restoration activities), the short- and long-term effects of those techniques, and the ability to create wetlands that mimic natural processes, are uncertain (MRDSEST 2012).

The primary purpose of wetland restoration through placed or diverted sediments is to increase spatial integrity by converting open-water features to sub-aerial habitat or to increase ecosystem stability by nourishing deteriorating wetlands. These restoration applications remain highly promoted and implemented, and have resulted in more than 2,000 man-made islands, 100 marsh creation projects, and nearly 1,000 habitat development projects in the United States (Brandon and Price 2007). However, necessary monitoring and assessments of restoration benefits and performance are often hindered by budget and resource constraints. As a result, knowledge gaps persist about the short- and long-term ecological evolution of restored and nourished sites, potentially resulting in ad-hoc and inefficient management of sediment and ecosystem resources. Therefore, a systematic approach to establishing ecosystem and project targets, quantifying environmental benefits, and evaluating project performance and resilience is necessary for maximizing the application and adaptive management of sediment for wetland creation and nourishment.

RESEARCH

Study Area – Louisiana Coastal Wetlands

This study explores the impacts of sediment introduction on plant community composition, species distributions, productivity, and wetland structure and resilience in degrading coastal Louisiana landscapes. The study was conducted in multiple parts and evaluations, across multiple sites ranging from project to landscape-scale (Figure 1.7). The largest assessment unit, the Louisiana coastal zone, encompasses approximately 14,000 km^2 and consists of Louisiana wetlands that are influenced by coastal processes. This study also assessed the impacts of sediment additions to wetlands within the northern Chandeleur Islands and the Sabine Refuge (Figure 1.7). The northern Chandeleur barrier island chain is located approximately 96 kilometers east of New Orleans, Louisiana. The northern island arc has experienced a long-term reduction in sand volume, which has resulted in rapid erosion of island features (due primarily to hurricane impacts) and the inability to maintain many of its subaerial features (US Army Corps of Engineers 2012). The Sabine Refuge Marsh Creation project and reference areas, consist primarily of intermediate and brackish wetlands that are located west of the Calcasieu Ship Channel near Hackberry, Louisiana (Figure 1.7). This area, which was severely impacted by hurricanes and canal building, experienced significant conversion from wetlands to open water between 1956 and 1978 (Miller 2014).

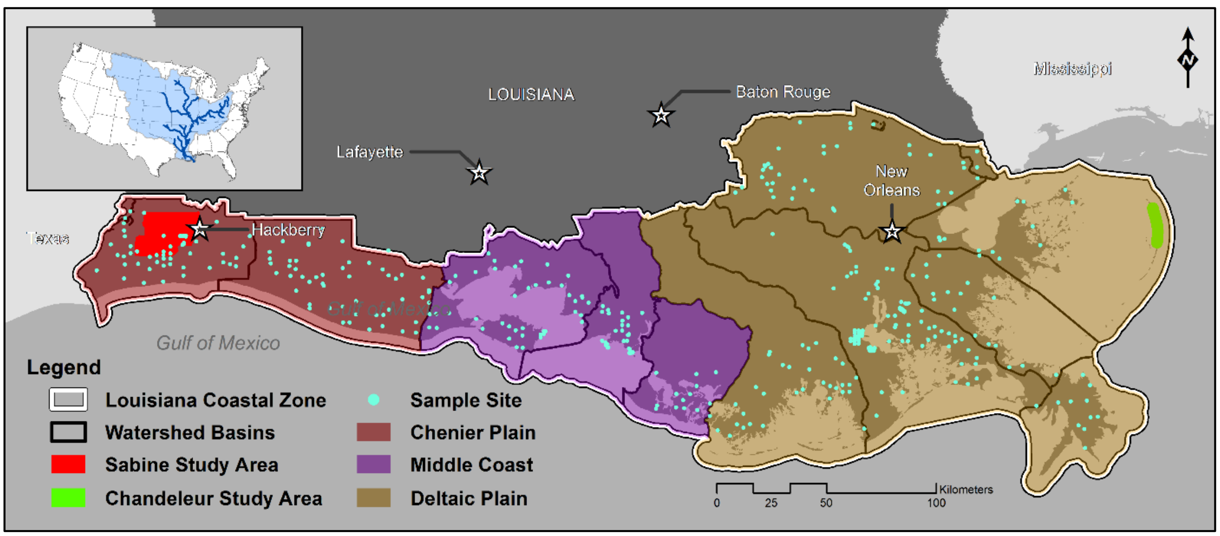

Figure 1.7. Location Map of study sites and components.

Research Objectives

In recent decades recognition by coastal resource managers and scientists about the ecological value of placed or diverted river sediments has elevated stakeholder dialogue, planning, and actions on identifying and pursuing a wider range of sediment applications. Placed or diverted sediments, which provide abatement to ongoing subsidence, erosion, and sediment and nutrient deprivation, have been the basis of many ecosystem restoration strategies in coastal wetlands. Though the ecosystem benefits associated with these applications are generally assumed, the short- and long-term monitoring necessary to quantify and validate those benefits have focused primarily on landscape form (areal extent and elevation), and less on productivity, function, and resilience. Understanding the geomorphic factors and ecological processes that govern wetland creation and nourishment applications will increase predictive and adaptive management capabilities. For ecosystem restoration through sediment diversion or placement to remain a viable long-term solution, the monitoring of existing sites is essential for assessing the ability of restoration measures to provide expected benefits and satisfy the needs defined by coastal wetland stewards and stakeholders. The overarching goal of this study was to gain an improved understanding of the effects of sediment introduction into wetland systems and the factors influencing the establishment, function, evolution, and resilience of restored wetland vegetation communities.

Alterations to Louisiana's river systems and local hydrology have resulted in reduced freshwater, sediment, and nutrient inputs to wetland landscapes, causing significant negative impacts on marsh productivity and stability. Therefore, Chapter 2 considered regional- and basin-scale impacts of river connectivity and sediment availability on wetland productivity. Satellite data were used in conjunction with river discharge, river sediment concentration, and wetland accretion data to evaluate correlations between river connectivity and wetland productivity. Chapter 2 also linked wetland area, configuration, and productivity with river connectivity to provide an enhanced understanding of river and sediment importance for wetland stability and restoration.

Louisiana's chronic wetland deterioration has also resulted in massive soil organic matter loss and subsequent carbon release through oxidation (DeLaune and White 2011; Williams 1995). To combat these losses, and reestablish ecosystem function, goods, and services, many restoration projects have been constructed or planned throughout coastal Louisiana. Chapter 3 used an exceptionally large data set to derive carbon accumulation rates from key soil characteristics and processes, and assessed and compared bulk density, organic matter, total carbon, vertical accretion (short- and longer-term), and carbon accumulation rates across time (chronosequence) and space (i.e., coastwide, watershed basins, and vegetation zones).

As part of the emergency response plan, sand berms were constructed along sensitive barrier islands to reduce impacts from the Deepwater Horizon oil spill. Chapter 4 evaluated the effects of redistributed berm sediments on island elevation, habitat, productivity, and floristic quality. A Geographic Information System (GIS) and remote sensing techniques were used to evaluate the evolution and redistribution of berm sediment within the Chandeleur Island system and assess the impacts of redistributed berm sediment on existing and new island features.

Restoration efforts in the United States have created or benefitted large expanses of wetlands. Typical goals of wetland restoration efforts are to conserve, create, or enhance wetland form, and to achieve wetland function that approaches natural conditions. Failure to adequately maintain restored wetland elevation, hydrology, sedimentation, or vegetation can ultimately lead to degrading wetland conditions. Measures of wetland condition have been used to monitor and assess project performance, resilience, and adaptive management needs. Chapter 5 assessed the use of landscape metrics (i.e., wetland area, edge density, and aggregation index) and vegetative indices (i.e., vegetation cover, normalized difference vegetative index, and floristic quality index) to evaluate change and trends in restored wetland condition, function, and resilience, and compared those to reference wetlands. The last chapter (6) is a summary and synthesis of these studies, providing a perspective on the relevance of sediment distribution for wetland restoration.

Integration with Other Research Projects

These studies are largely funded by the United States Army Corps of Engineers Ecosystem Management and Restoration Research Program (EMRRP), Dredging Operations Technical Support

Program (DOTS), and Engineering With Nature (EWN) Program, and were developed to provide clearer understandings of integrity, resilience, reliability, and sustainability of coastal ecosystem restoration projects. These studies utilized data made available through active agreements with the National Geospatial-Intelligence Agency, current monitoring systems (Coastwide Reference Monitoring System and Coastal Wetlands Planning, Protection and Restoration Act), and previous data collection efforts (USACE Beneficial Use Monitoring Program and BP Coastal Wetland Vegetation Assessment Data). These studies were also leveraged to acquire resources and in-kind support from the Louisiana State University's Department of Oceanography and Coastal Sciences, U.S. Geological Survey's Wetland and Aquatic Research Center, the U.S. Fish and Wildlife Service, and the Louisiana Department of Wildlife and Fisheries.

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CHAPTER 2 – USE OF THE NORMALIZED DIFFERENCE VEGETATION INDEX AND LANDSCAPE METRICS TO ASSESS EFFECTS OF RIVERINE INPUTS ON WETLAND PRODUCTIVITY AND STABILITY

INTRODUCTION

Over the last century, flood risk reduction measures constructed in south Louisiana have significantly reduced connectivity between the Mississippi River (MR) system and coastal marshes (Kesel 1988; Mississippi River Delta Science and Engineering Special Team [MRDSEST] 2012). In the rapidly subsiding Mississippi River Delta (active and inactive deltas), this disconnect has resulted in sediment and nutrient deficits which have contributed to Louisiana's 4,877 square kilometers (km^2) of wetland loss (a net wetland change of -25%) that occurred between 1932 and 2010 (Craig *et al*. 1979; Turner 1997; Kennish 2001; Couvillion *et al*. 2011). Considering projected rates of relative sea-level rise, it is expected that river connectivity and sediment delivery to wetland landscapes will become increasingly vital for maintaining and restoring wetland ecosystem structure and functions (Jankowski *et al*. 2017).

The ecological benefits of a connected ecosystem have long been assumed and are often the target of restoration activities. Many of Louisiana's large-scale wetland restoration plans include river connectivity measures (i.e., sediment and nutrient delivery) to promote wetland nourishment and land building processes. Numerous small-scale studies have shown an increase in wetland extent, biomass, and vigor with increasing freshwater, sediment, and nutrients inputs (Martin *et al*. 2002; DeLaune *et al*. 2005; McFalls *et al*. 2010; Roberts *et al*. 2015; DeLaune *et al*. 2016). However, the degree to which river connectivity (including freshwater and sediment diversion restoration measures) influences large-scale wetland productivity and stability in the current Louisiana landscape, is still debated (Kearney *et al*. 2011; MRDSEST 2012; Suir *et al*. 2014).

Hydrologic Connectivity

Hydrologic connectivity, which is defined as the water-mediated transfer of matter, energy, and organisms within or between elements of the hydrologic cycle, is a fundamental component of ecological integrity in wetland landscapes (Amoros and Roux 1988; Heiler *et al*. 1995; Pringle 2003; Freeman *et al*. 2007). Historically, Louisiana has had an abundance of riverine connectivity, consisting of a complex network of rivers and distributaries (green lines in Figure 2.1) that traversed the Middle Coast and Deltaic Plain (Fisk 1944). However, flood risk reduction features have disconnected or restricted those nourishing rivers and bayous from large expanses of wetland landscapes. South of the Old River Control Structure near Simmesport, Louisiana, approximately 2,350 km of federally maintained primary levees have been constructed along the Atchafalaya River (AR) and MR. These levees, in addition to declining suspended load (80 percent decrease since the middle of the nineteenth century), have resulted in significant reductions in unconfined and overbank distribution of sediment (Kesel 1988, 1989). Because only the lower reaches of the AR (lower 48 km) and MR (lower 32 km) are un-leveed, the unconfined or overbank distributions of river waters are typically discharged onto or over the continental shelf, or into relatively isolated and emaciated wetlands (Walker and Rouse 1993; Suir *et al*. 2014). Some AR waters are transported through smaller crevasses and pathways and into nearby Middle Coast and active delta wetlands (Swarzenski 2003) (Figure 2.1, yellow lines within the Middle Coast region). The Gulf Intracoastal Waterway (GIWW) is a primary AR distributary, conveying river water approximately 50 km east and 80 km west of the river (Swarzenski 2003).

Objectives

The purpose of this study was to assess the impacts of riverine inputs on wetlands by using remote sensing data to compare hydrologically connected landscapes (Middle Coast) to areas that are either more disconnected (Deltaic Plain) or connect to low volume rivers (Chenier Plain). Ecosystems with higher hydrologic connectivity are assumed to contain more stable and productive wetlands due to increased nutrient and sediment delivery. The specific objectives of this study were to: (1) evaluate sediment delivery potential and river connectivity; (2) assess correlations between wetland productivity and riverine influence; (3) evaluate trends in wetland stability and correlations to productivity and river connectivity; and (4) consider the implications of this research on restoration activities and adaptive management.

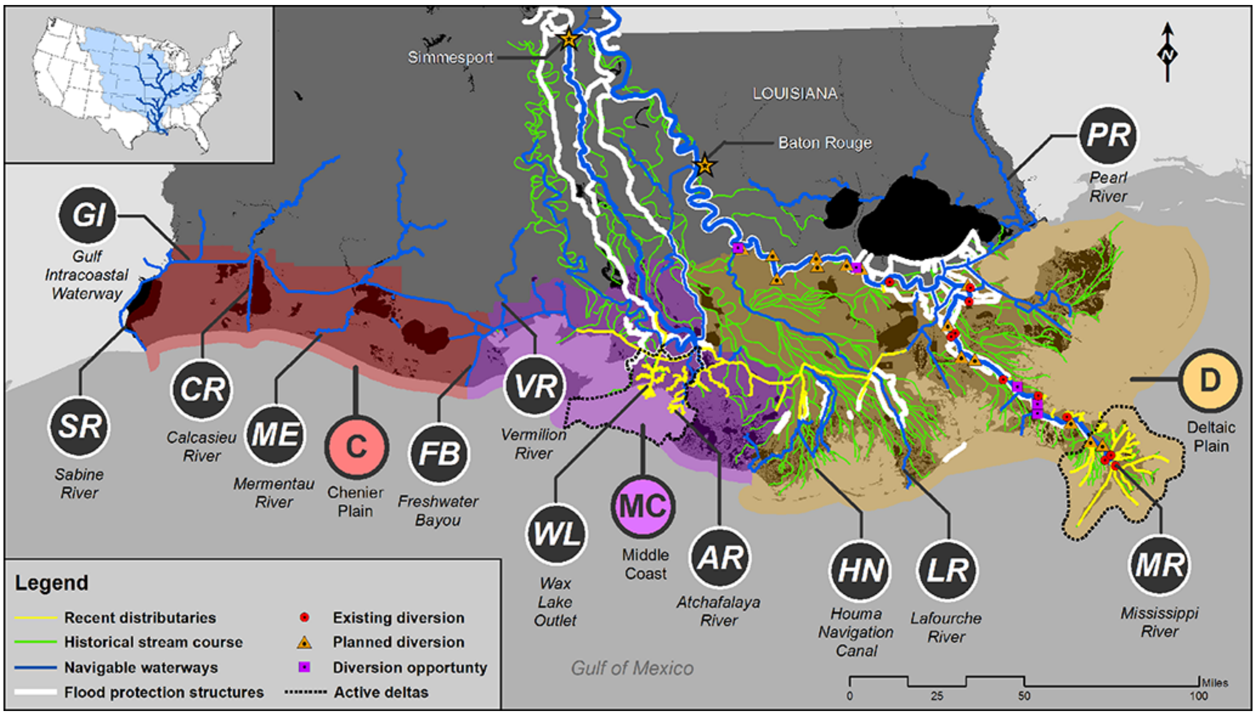

Figure 2.1. Map depicting the Mississippi River drainage basin (inset), major navigable waterways, flood risk reduction levees, historical and recent tributaries, diversions, and active deltas in coastal Louisiana (Fisk 1944; Huh *et al*. 2001; Khalil 2012; Shi and Wang 2009; USACE 2006).

METHODS

Study Area and Assessment Units

The study area, encompassing approximately 14,000 km^2, consisted of Louisiana wetlands that are influenced by coastal processes (Figure 2.2). To assess potential correlations between wetland productivity and hydro connectivity; while considering seasonal trends, geomorphic settings, and episodic impacts; multi-scale assessment units were established. These include (1) Geomorphologic Zones (GZ), (2) River Buffers (RB), (3) Watershed Basins (WB), and (4) Vegetation by Basin units (VB). The GZ consist of three distinct geomorphologic areas within coastal Louisiana. These zones, Chenier Plain, Middle Coast, and Deltaic Plain, have and continue to develop under different coastal processes (Figure 2.2). Likewise, the RB units allow for assessments of condition and influence, but specifically as a function of distance from primary rivers. The RB consist of buffers that radiate at 5 km increments (based on Visser *et al*. 2003) from each river to a total distance of 40 km or to distances of overlapping coverages from neighboring RB.

The WB units consist of Louisiana drainage basins and subwatersheds (Louisiana Department of Environmental Quality [LDEQ] 2004). Since LDEQ basins were delineate based on catchment areas of a river (up to its confluence), they serve as suitable units for assessing hydrologic connectivity. Since some watershed basins are large and encompass distinct subwatersheds of interest, and some are small and adjacent to basins of similar hydrology, several modifications were made to the original boundaries. These modifications include the Sabine and Calcasieu basins, which are moderately small with similar hydrology, so they were combined to form the Calcasieu/Sabine WB unit. Also, since the AR has been shown to substantially influence the western portion of the Terrebonne drainage basin (Visser *et al*. 2003), the basin was divided into the Penchant Marsh unit to the west (area receiving AR influence) and the more river-disconnected Terrebonne proper unit to the east (Wang *et al*. 1993). Similarly, since the Pontchartrain drainage basin consisted of hydrologically unique subwatersheds, it was divided into the Pontchartrain proper, Breton Sound, and Biloxi Marsh units. However, since the Pontchartrain proper subunit consists primarily of forested wetlands and swamp, it was excluded from this study.

Louisiana's marshes have traditionally been characterized by their salt tolerance, and grouped into Fresh (0 to 0.5 practical salinity [S_P]), Intermediate (0.5 to 5 S_P), Brackish (5 to 18 S_P), and Saline (18 to 30 S_P) classes (Sasser *et al*. 2014). The impacts of sediment and nutrient loading on the plants in each of these vegetation zones can vary significantly (Visser *et al*. 2003), therefore, the Vegetation by Basin (VB) zones were used to compare productivity for each unique vegetation zone by drainage basin combination (Figure 2.2).

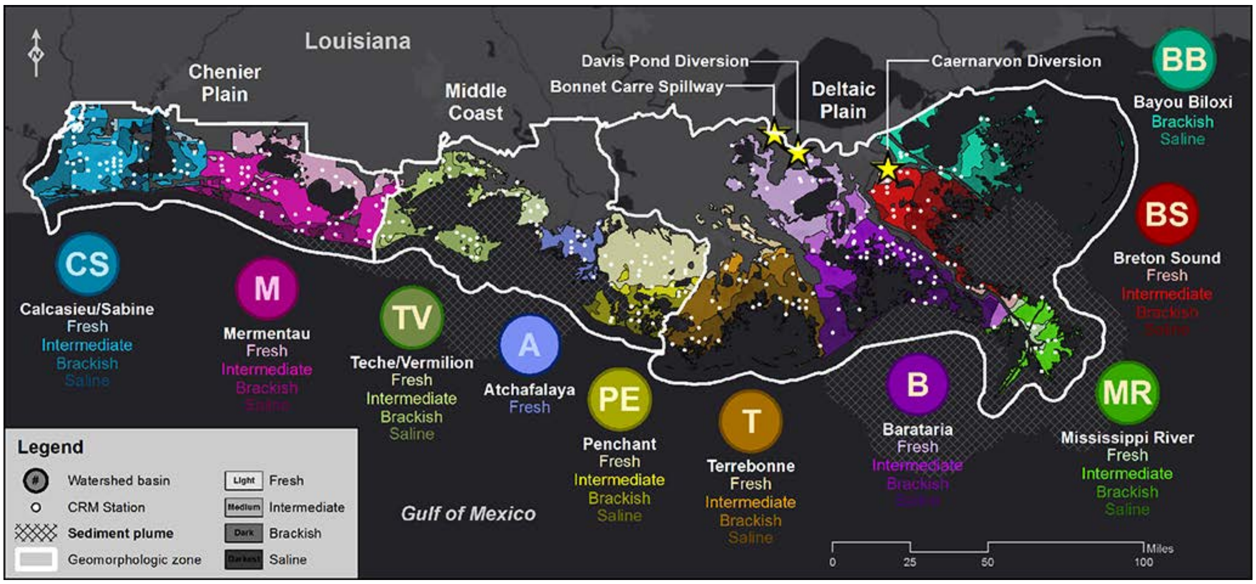

Figure 2.2. Coastal Zone, Watershed Basins, and Vegetation by Basins assessment units in coastal Louisiana. White dots represent the locations of the Coastwide Reference Monitoring System stations and hatched areas represent the typical sediment plume for the Atchafalaya and Mississippi Rivers.

Sediment Availability, Accretion, and Wetland Change

Assessing correlations between plant productivity and riverine inputs require the establishment of sediment delivery potential (Bianchi *et al*. 2002; Falcini *et al*. 2012; Roberts *et al*. 2015; DeLaune *et al*. 2016). This includes measurements of instream sediment concentration and evaluations of river connectivity to assessment unit wetlands. Mean daily discharge and total suspended sediment data for major rivers in Louisiana were extracted from literature or computed using United States Geological Survey (USGS) National Water Information System data (2016). Additionally, similar methods to Khan *et al*. (2013) and Turnipseed *et al*. (2014) were used to assess surface elevation changes within Louisiana's coastal watershed basins. Elevation and accretion data from all available Coastwide Reference Monitoring System (CRMS) stations (n = 292; Figure 2.2) were used to evaluate surface elevation changes related to the major river flood events in 2008 and 2011 (Figure 2.3), as well as the mean elevation changes across the entire CRMS period of collection (2008-2016). CRMS surface elevation table (SET) data provide measures of recent sedimentation (long-term and flood-related) at higher spatial scale than previous data sets and assessments. The extent and density of these CRMS data provide unique opportunities for evaluating cause and effects of elevation and elevation change on wetland processes (Jankowski *et al*. 2017).

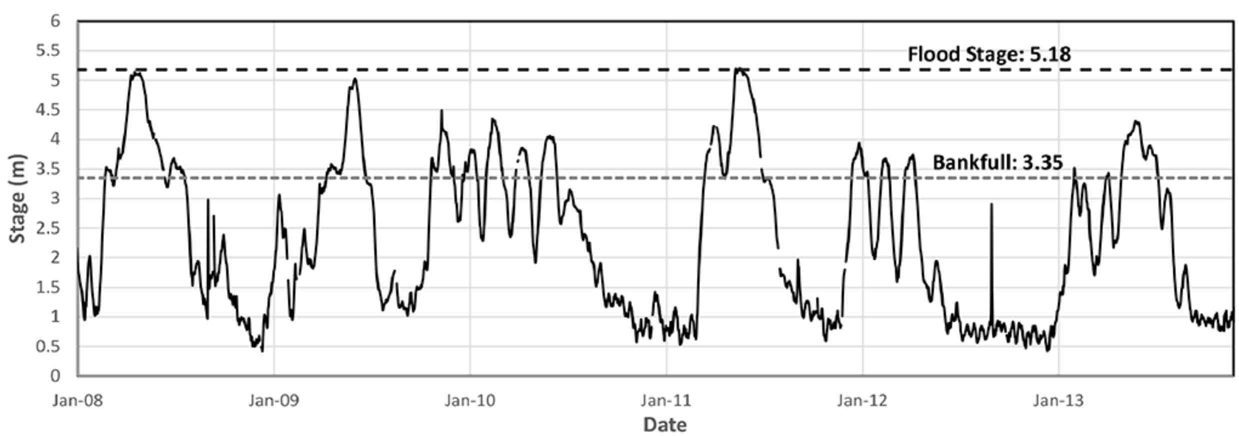

Figure 2.3. Daily stage (meters) for the Mississippi River at New Orleans, Louisiana (Rivergages.com accessed 28 Jan 2017).

Another indirect measure of long-term sediment delivery involves analyzing wetland change patterns with distance from sediment source, as described in Visser *et al*. (2003). The percentage of wetland change from 1956 to 2008 was calculated with distance from primary Louisiana navigable waterways using the RB assessment zones. The 1956 wetland and water data (Barras *et al*. 1994; 1:24,000) are based on panchromatic aerial photography-derived habitat data that were generated by the USGS Wetland and Aquatic Research Center (Suir *et al*. 2011). The 2008 data consist of wetland and water classified Landsat Thematic Mapper (TM) imagery that were previously developed for hurricane assessments (Barras 2009). The wetland and water classified data were resampled and analyzed at a spatial resolution of 28 m. Wetland change percentages were calculated for each buffer by subtracting the area of wetland in 1956 from the 2008 area, dividing by the total buffer area, and multiplying by 100.

Remote Sensing

Since the Normalized Difference Vegetation Index (NDVI) has well established correlations to plant characteristics (i.e., photosynthetic activity and biomass; Carle 2013), it was used in conjunction with Landsat (28 meter) and MODerate Resolution Imaging Spectroradiometer (MODIS, 250 m [bands 1-2] and 500 m [bands 3-7]) satellite imagery to assess wetland productivity. Both data sets were acquired using the Google Earth Engine (GEE) image service. GEE utilizes radiometrically and atmospherically corrected imagery, and aggregation functions (i.e., use of outlier values to remove cloud cover from neighboring scenes) to create image composites (Strahler *et al*. 1999; Chander *et al*. 2009). The GEE service also provides NDVI data that are derived as:

$$NDVI = \frac{NIR - Red}{NIR + Red}, \tag{1}$$

where this ratio of the near-infrared band (NIR) and red band (Red) is used to measure an ecosystem's ability to capture solar energy and convert it to organic carbon or biomass (Rouse *et al*. 1974; An *et al*. 2013). All non-marsh features (i.e., forest, developed lands, and water) were excluded from each Landsat and MODIS image. Since NDVI values less than zero (< 0) are typical of non-vegetation features (e.g., water, cloud, impervious surfaces) (Reif *et al*. 2011; Carle 2013), those were also excluded.

Landscape Configuration

Landscape ecology is based on the premise that there are strong correlations between landscape pattern (configuration) and ecosystem function (Gustafson 1998). The aggregation index (AI) has evolved as a primary metric for linking structure to ecosystem function and is defined as the frequency with which different pairs of patch types appear side-by-side (McGarigal 2015), including like adjacencies between the same patch-type. Combined with wetland area change, AI provides a measure of landscape integrity that is suited for assessing wetland stability and potential correlations to plant productivity (Suir *et al*. 2009; Sun *et al*. 2015; Couvillion *et al*. 2016). The class-level aggregation index (AI) is derived as:

$$AI = \left(\frac{g_{i,i}}{\max\_g_{i,i}}\right)(100), \tag{2}$$

where $g_{i,i}$ is the number of like adjacencies between pixels of patch type i (class), $\max\_g_{i,i}$ is the maximum number of like adjacencies between pixels of patch type (class) i (He *et al*. 2000; McGarigal 2015). The AI was computed for all assessment units using FRAGSTATS v4.2 (McGarigal *et al*. 2012) and a sequential series of 19 wetland and water data sets. Existing wetland and water data from 1988 to 2008 (Barras *et al*. 1994; Hartley *et al*. 2000; Barras 2005; Morton *et al*. 2005; Barras 2009) were supplemented by performing wetland and water classifications on newly acquired Landsat TM imagery through the GEE service (2009, 2010, 2011, and 2013). These wetland and water classified data were also used to compute the total Class Area (CA) of wetlands within each assessment unit.

Statistical Analysis

All data sets were transformed and formatted as comma separated values (CSV) files for statistical analyses. In order to attain comparability among NDVI for each assessment scale, statistical analyses were conducted using Statistical Analysis System software version 9.2 (SAS 2010). The PROC GLM procedure was used to perform a one-way analysis of variance (ANOVA) and a means separation test (Tukey's, $\alpha = 0.05$) to evaluate significance of differences between NDVI for each assessment unit. Additionally, a second order polynomial regression with coefficient of determination (r^2) was used to evaluate correlations between productivity (NDVI) and stability (aggregation index) with consideration of hydrologic connectivity.

RESULTS AND DISCUSSION

Hydrology

To assess the impacts of river-borne sediment on wetland productivity and configuration, linkages between river and wetlands must be evaluated. This was accomplished by assessing sediment source and delivery potential using river discharge, river suspended sediment concentrations, and trends in accretion rates and wetland change (with distance from source). Table 2.1 lists the river flow and suspended sediment concentrations for the primary navigable waterways (with related WB) in coastal Louisiana. The mean flows of the MR and AR (combined with Wax Lake Outlet) are approximately 16,000 m^3s^{-1} and 6,000 m^3s^{-1}, respectively (Sprague *et al*. 2009; Rego *et al*. 2010). The mean sediment concentration of the MR and AR are approximately 260 milligram per liter (mgL^{-1}) and 470 mgL^{-1}, respectively (Rosen and Xu 2013; Thorne *et al*. 2008). Since 1950, the AR has conveyed all of the suspended- and bed-sediment load of the Red River, and approximately 35%, 60%, and 30% of the MR's suspended sediment, bed sediment, and latitude flow (all river system water passing through latitudinal plane), respectively (USACE 2004; Hupp *et al*. 2008). The remaining primary waterways (excluding the GIWW) have reported mean flows that range from 33 to 219 m^3s^{-1}, and mean sediment concentrations that range from 17 to 56 mgL^{-1} (Rosen and Xu 2011; LDEQ 2016; USGS 2016) (Table 2.1). The AR and MR are generally one magnitude higher in mean total suspended sediment concentration and several magnitudes higher in discharge rates than other primary waterways. Suspended sediment concentrations of the AR's distributaries are also higher than most isolated rivers and bayous. This is evident in the GIWW, with mean sediment concentrations of 177 mgL^{-1} west of Wax Lake Outlet (east of Cypremort Point) and 137 mgL^{-1} east of the AR (west of the Houma Navigation Canal) (Swarzenski 2003).

Table 2.1. Mean daily discharge (flow) and mean total suspended solids (TSS) for primary rivers in coastal Louisiana. Modified from Benke and Cushing (2011).

River	Basin	Mean Flow (m^3s^{-1})	Mean TSS (mgL^{-1})	Source
Sabine	Calcasieu/Sabine	219	17	Rosen and Xu 2011
Calcasieu	Calcasieu/Sabine	72	18	Rosen and Xu 2011
Mermentau	Mermentau	82	26	Rosen and Xu 2011
Vermilion	Teche/Vermilion	33	56	Rosen and Xu 2011
GIWW west of WLO*	Vermilion	158	177	Swarzenski 2003
Atchafalaya/Wax Lake Outlet**	Atchafalaya	6,227	469	Rego *et al*. 2010; Rosen and Xu 2013
GIWW east of Atchafalaya***	Penchant	156	137	Swarzenski 2003
Houma Navigation Canal	Terrebonne	90	41	LDEQ 2016; USGS 2016
Lafourche	Barataria	35	27	LDEQ 2016; USGS 2016
Mississippi	Mississippi River	16,339	259†	Sprague *et al*. 2009, Thorne *et al*. 2008

* West of the Wax Lake Outlet to Cypremort Point
** Flow and TSS for Atchafalaya and Wax Lake Outlet combined
*** East of the Atchafalaya River to the Houma Navigation Canal
† Mississippi River at Tarbert Landing

Sediment Accumulation

Figure 2.4, Panel A, illustrates surface elevation change by CRMS station (green and red dots represent elevation increases and decreases, respectively) and mean elevation change by WB units (polygons) that were computed using pre- (mean surface elevation between October 2010 and April 2011) and post- (July 2011 to November 2011) 2011 flood data. The darkest green dots and darkest polygons represent areas of maximum elevation increases, while dark red dots represent areas of maximum decreases. Figure 2.4A shows the majority of the CRMS stations in close proximity to larger river outfalls or floodway systems (i.e., Bonnet Carré Spillway, which opened on 9 May 2011 to alleviate flooding stress [USACE 2016]) experienced increases in relative surface elevation, while those located at greater distances were dominated by decreases in elevation.

These findings are similar to those by Falcini *et al*. (2012), who observed sites within the Atchafalaya (1.61 ± 0.96 gram per cubic centimeter [g cm^{-2}], n = 14) and Mississippi River Deltas (1.14 ± 0.78 g cm^{-2}, n = 9) had the greatest 2011 flood related accumulation of sediment. Figure 2.4A also shows the WB units near or between large river outfalls (i.e., Terrebonne and Barataria) or floodway systems experienced the highest mean elevation increases. Though Falcini *et al*. (2012) observed more moderate relative sediment accumulation in these areas (Terrebonne 0.42 ± 0.18 g cm^{-2}, n = 14 and Barataria 0.34 ± 0.22 g cm^{-2}, n = 8), the dissimilarities are potentially due to differences in assessment area scale and sample locations. Accretion data from the 2008 MR flood correspond to the elevation data and trends that were observed with the 2011 flood, therefore, those data are not shown here for brevity. Panel B of figure 2.4 illustrates the elevation change rate for each CRMS station from 2006 to 2016. Across the period of record the elevation trends were similar to those that were observed pre- and post-floods, with higher increases in relative elevation around high flow and high sediment-concentration rivers (i.e., AR and MR). These findings corroborate those by Jankowski *et al*. (2017), who correlated accretion rates with proximity to riverine sediment inputs, connectivity to the Gulf of Mexico, and impacts of Chenier ridges and impoundments.

Wetland Change with Distance from River

A second hydrologic connectivity assessment was performed using the wetland change with distance from river method described in Visser *et al*. (2003). The wetland change percentage by distance (buffers) results are provided in Table 2.2. Similar to findings by Visser *et al*., wetland loss increased with distance from sediment source for most Chenier Plain and Middle Coast basins. This was not true for all basins, especially those neighboring the AR and MR basins (i.e., Teche/Vermilion, Terrebonne, and Barataria basins), since their outer regions receive large concentrations of sediments through coastal processes, thereby superseding the influence from nearby smaller rivers (Rego *et al*. 2010). This was also not true for the eastern Barataria basin (MR West) and MR active delta (MR East, Table 2.2), which are either disconnected (west), or consist of small catchments in high energy environments (east) where the MR transports large proportions of its sediment beyond the delta's wetlands (Winer 2011). Conversely, within the northern Breton Sound basin (MR North, Table 2.2), data corroborate those by Visser *et al*. (2003) which show the buffers receiving freshwater inputs through the Caernarvon Freshwater Diversion (25 km and 30 km, Figure 2.2) are those with the lowest wetland loss percentages.

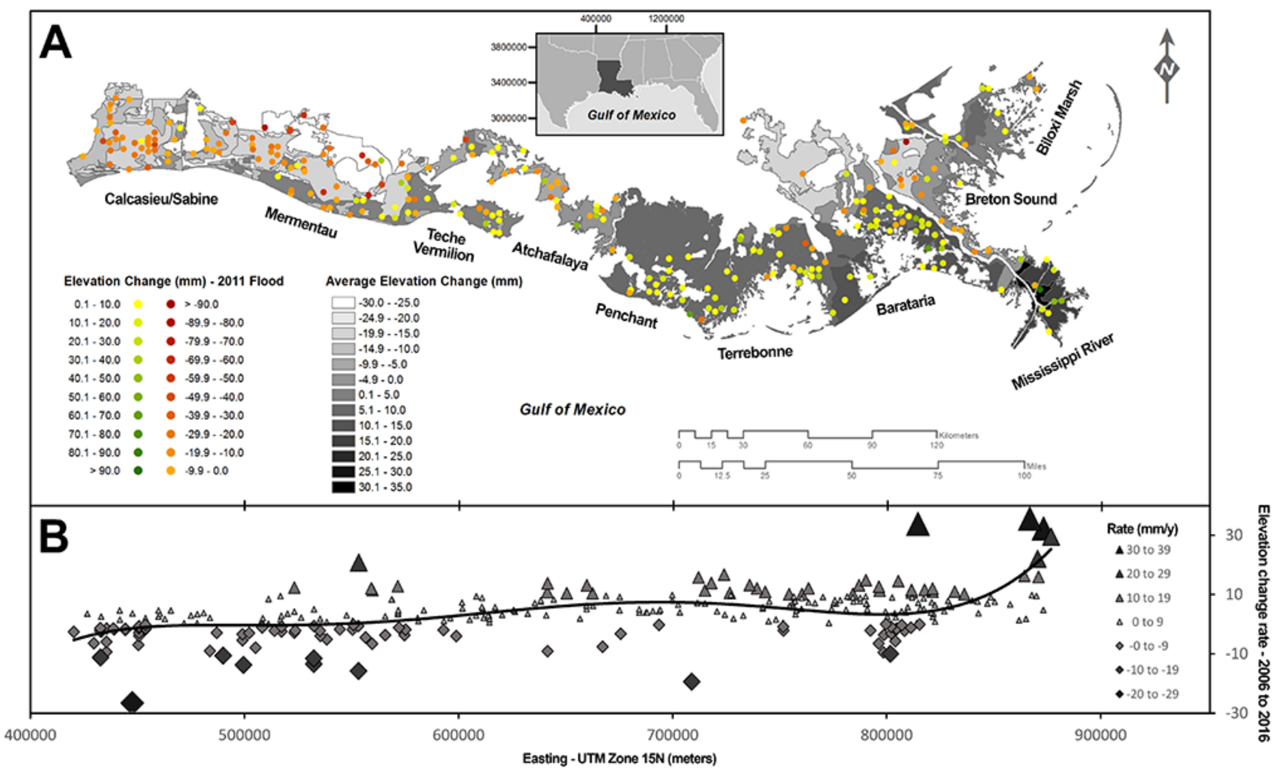

Figure 2.4. Baseline and flood-related relative elevation change across the Coastwide Reference Monitoring System (CRMS) period of record. Panel A shows the change in elevation (pre- and post-2011 Mississippi River flood) for all CRMS stations (yellows to greens represent increasing elevations and oranges to reds represent decreasing elevations) and the mean elevation change by watershed basin (polygons). Panel B represents total elevation change rates, where increases are represented by triangles and losses by diamonds. Magnitude of change is color ramped in all panels.

These assessments of hydrologic connectivity largely corroborate previous smaller-scale studies, where sedimentation and nutrient availability increase with connectivity to riverine source and hydro period (inundation and duration), and generally, wetlands in close proximity to highly connected rivers experience higher rates of accretion and lower wetland loss. Exceptions to these include wetlands receiving sediment from distant sources, areas with increased tidal exchange and more frequent salinity spikes due to hydrologic alterations (i.e., Houma Navigation Canal impacts in Terrebonne Basin) (CLEAR 2006; Steyer *et al*. 2008), and disconnected wetlands–especially those in rapidly subsiding landscapes (i.e., lower Barataria and Mississippi River basins) (Suir *et al*. 2013; Suir *et al*. 2014).

Table 2.2. Coastal wetland change (1956 to 2013) with distance from primary rivers.

		Distance from River (km)							
		5	10	15	20	25	30	35	40
Location	Basin	Wetland Change (%) from 1956 to 2013							
Sabine River	Calcasieu/Sabine	-8.9	-10.5	-20.8	-25.0	-40.4	-30.2	-	-
Calcasieu River	Calcasieu/Sabine	-7.3	-19.1	-31.8	-23.1	-24.3	-27.2	-	-
Mermentau River	Mermentau	-10.8	-14.2	-13.5	-13.5	-9.0	-12.2	-12.9	-16.7
Vermilion/Freshwater Bayou	Teche/Vermilion	-15.7	-10.8	-12.7	-8.6	-10.2	-7.8	-6.5	-5.9
Atchafalaya/Wax Lake Outlet	Atchafalaya	2.2	-3.4	-2.3	-11.0	-7.5	-7.6	-6.7	-7.3
GIWW (Atchafalaya Influence)	Multiple	-9.6	-10.6	-14.0	-15.0	-15.9	-12.6	-	-
Houma Navigation Canal	Terrebonne	-22.1	-25.8	-17.1	-14.9	-16.8	-15.6	-8.7	-5.1
Bayou LaFourche	Barataria - west	-23.8	-24.2	-16.4	-15.7	-14.3	-9.5	-12.2	-10.0
Mississippi River West	Barataria - east	-32.8	-40.3	-32.6	-8.4	-5.0	-7.1	-10.6	-16.5
Mississippi River East	Mississippi River	-22.6	-15.0	-10.5	-9.5	-5.9	-1.5	0.1	-1.1
Mississippi River North	Breton Sound	-13.1	-23.6	-21.3	-16.6	-7.7	-7.0	-7.4	-5.4

Geomorphologic Zone Productivity

Figure 2.5 illustrates the spatial variability and patterns of MODIS-derived NDVI within the Louisiana coastal zone and across the Chenier Plain, Middle Coast, and Deltaic Plain units. The mean NDVI, per pixel, ranged from 0.01 to 0.6 across the 2003 to 2013 period of analysis. The Chenier Plain had a mean NDVI of 0.48 ± 0.10 and consisted primarily of moderate vegetative productivity (yellow to orange hues). Though portions of this Plain are still influenced by the Atchafalaya River (i.e., westward transportation of sediment and reworking via marine processes), larger areas receive inputs from moderate sediment and nutrient concentration rivers (Penland and Suter 1989; Gammill *et al*. 2002; Rosen and Xu 2011). The Chenier Plain has also undergone extreme hydrologic modifications (i.e., channelization, water control structures, oil and gas access canals) that have increased flooding frequency and duration, which in turn has had significant negative impacts on wetland productivity and condition (Gammill *et al*. 2002; Rosen and Xu 2011). Figure 2.5 also shows large expanses of wetlands within the Middle Coast region were highly productive (dark green hues), with a mean NDVI of 0.55 ± 0.12. The Middle Coast is a hydrologically connected landscape with riverine, marine, atmospheric, and seasonal processes that deliver high concentrations of sediment and nutrients to wetlands and prograding deltas (Wax Lake and Atchafalaya) (Perez *et al*. 2000, 2003; Rosen and Xu 2013). The Deltaic Plain, which consisted of a mixture of high to low productivity (green and red hues, Figure 2.5), had a mean NDVI of 0.4 ± 0.08. The Deltaic Plain contains some isolated wetlands that receive high inputs from the MR, some inputs from the AR (northern Terrebonne via the ICWW), yet most are disconnected wetlands that have undergone extensive degradation (Couvillion *et al*. 2011).

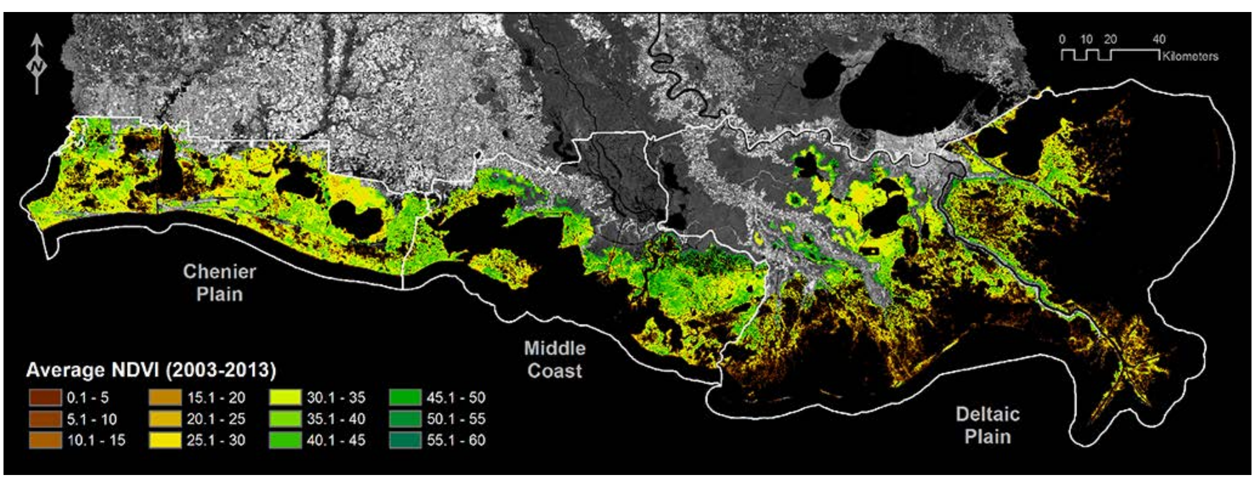

Figure 2.5. Productivity classification based on quartile distribution of MODIS-derived mean Normalized Difference Vegetation Index values (2003-2013) in coastal Louisiana.

Watershed Basin Productivity

Watershed basins are delineated by drainage area and therefore provide units that are useful for assessing hydrology-related landscape condition (Figure 2.2). Figure 2.6 shows the mean NDVI values from 2003 to 2013 for each watershed basin in coastal Louisiana, and the corresponding means for the Middle Coast, Chenier and Deltaic Plains (dashed lines). The Calcasieu/Sabine and Mermentau basins (comprising the Chenier Plain), had mean NDVI values of 0.46 ± 0.09 and 0.5 ± 0.11, respectively. This was lower than those of the Middle Coast but higher than the Deltaic Plain basins. The Middle Coast basins, Teche/Vermilion, Atchafalaya, and Penchant, had mean NDVI values of 0.53 ± 0.1, 0.59 ± 0.14, and 0.53 ± 0.1, respectively. The mean productivity in these Middle Coast units were significantly higher ($p<0.05$) than all but the Mermentau basin, which receives AR sediment and nutrients through shoreward transport and onshore deposition (Gammill *et al*. 2002; Draut *et al*. 2005). The Deltaic Plain basins, Terrebonne, Barataria, Breton Sound, Biloxi Marsh, and Mississippi River, had the lowest NDVI values, at 0.36 ± 0.06, 0.44 ± 0.08, 0.45 ± 0.12, 0.41 ± 0.06, and 0.36 ± 0.12, respectively.

Although mean NDVI values were not significantly different between some basins, small changes or differences in NDVI values have been correlated to significant differences in biomass. Tan *et al*. (2003) quantified the relationship between Landsat-derived NDVI values and wetland vegetation biomass, concluding that each 0.1 change in NDVI value correlates to 500 g/m^2 change in aboveground biomass (r^2 = 0.82). Correspondingly, low and decreasing productivity in coastal Louisiana basins have been attributed to marsh deterioration and fragmentation resulting from long-term sediment, nutrient, and freshwater deprivation (Boesch *et al*. 1994; Day *et al*. 2000; Cardoch *et al*. 2002; Couvillion *et al*. 2016). The general tendency in these data show a correlation between wetland productivity and river connectivity.

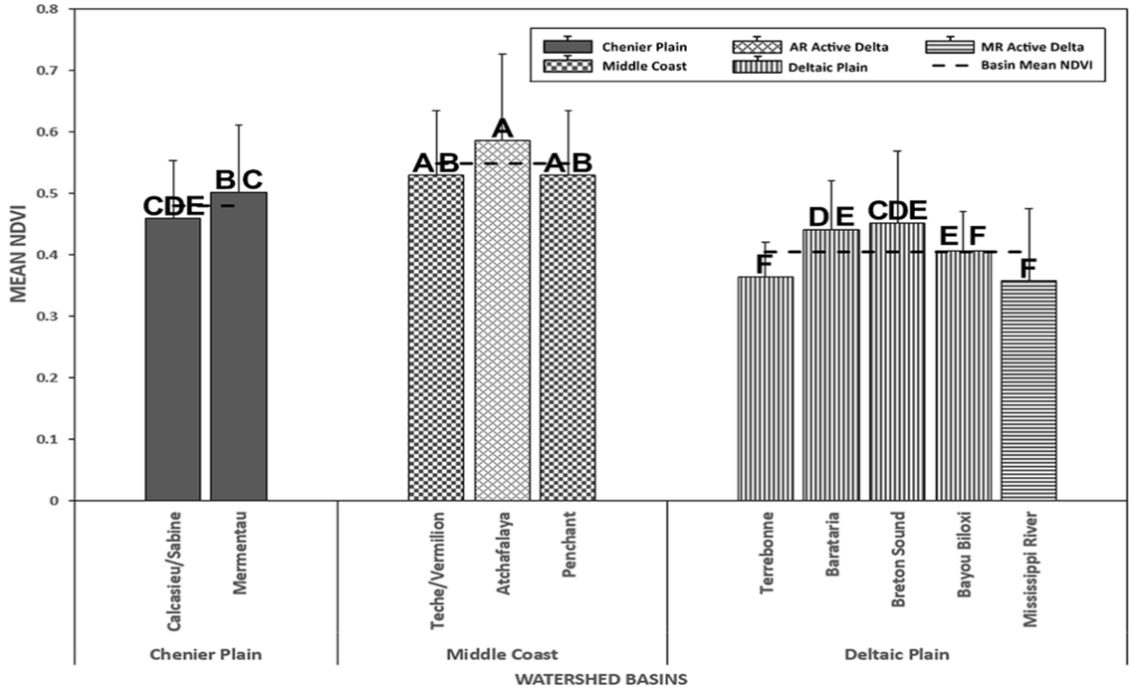

Figure 2.6. Mean Normalized Difference Vegetation Index values (2003 to 2013) for each geomorphologic zone (dashed line) and watershed basin (bars) in coastal Louisiana. Bars with the same letter are not statistically different at $p < 0.05$ (Tukey's HSD test).

Vegetation-by-Basin Productivity

Correlations were observed between NDVI (productivity) and Louisiana's vegetation zones (Figure 2.7, dashed lines). These findings corroborate those in previous research, which show that lower salinity environments typically consist of plants with higher leaf area and productivity (Gough and Grace 1998; Steyer 2008; Janousek and Mayo 2013). Figure 2.7 also provides a representation of the mean NDVI value for each VB unit. The AR influence is evident in each of the four vegetation zones. In the fresh zone the higher NDVI values occur in basins that are in closest proximity to the AR. Furthermore, even though the Terrebonne Basin is not in the Middle Coast GZ, the fresh portion of this basin frequently receives large inflow of AR water by way of the GIWW (Swarzenski 2003). Within the intermediate zone many of the basins had similar mean NDVI values, except for the Teche/Vermilion and the Mississippi River basin areas, which accounted for the maximum (0.54) and minimum (0.36), respectively.

Similar trends occurred in the brackish and saline zones, where NDVI values were highest for the Chenier, Teche-Vermilion, and northeastern Deltaic Plain basins. The high NDVI values in the brackish and saline portions of the Chenier and Teche-Vermilion basins are likely due in part to westward marine transport of AR sediment and nutrients (Gammill *et al*. 2002). The forcings contributing to higher Deltaic Plain values in the brackish and saline zones are less obvious, but could be due to impacts of Caernarvon Freshwater Diversion sediments on Breton Sound wetlands, and nutrient and suspended solids from coastal discharges (from Pearl River and Lake Pontchartrain passes) assimilating in the Bayou Biloxi system (Poirrier and Handley 2002).

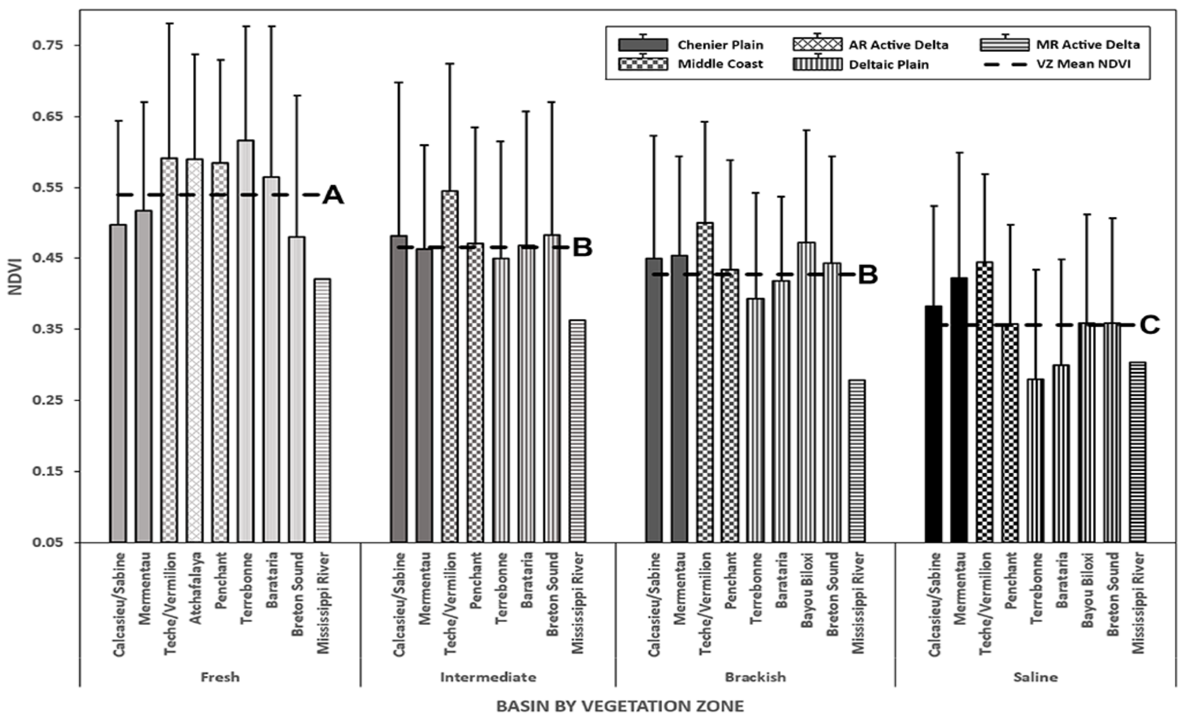

Figure 2.7. Mean Normalized Difference Vegetation Index values for each vegetation zone (dashed line) and basins (bars) within vegetation zone. Dashed lines with the same letter are not statistically different at $p < 0.05$ (Tukey's HSD test).

Flood Impacts on Productivity

A departure from average approach was used to assess flood impacts on plant productivity. This method, which compares end of growing season NDVI-derived mean productivity from non-flood years (baseline) to those from flood years, is a useful measure that links riverine connectivity to plant response. The premise behind this assessment is that wetlands with higher river connectivity will receive higher than normal sediment and nutrient inputs (due to major flood events, Day *et al*. 2016) and will therefore have a higher productivity (or positive departure from average). Figure 2.8 illustrates departure from average values comparing MODIS-derived NDVI from August 2011 (post-flood peak biomass) to average August values from non-flood years (and excluding years with major hurricane events).

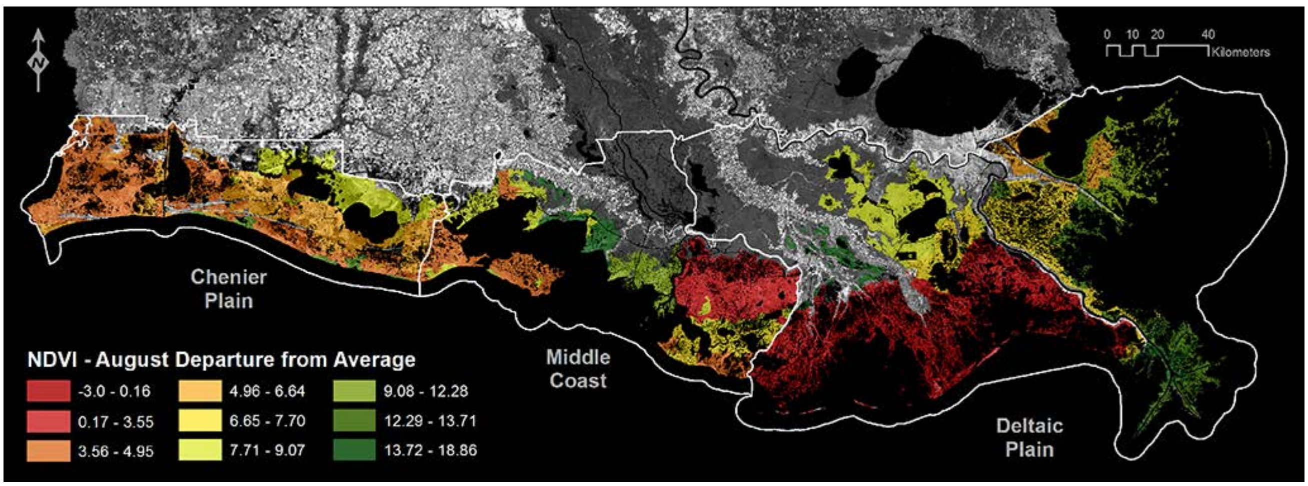

Figure 2.8. Departure from average using Normalized Difference Vegetation Index from August 2011 (post-flood peak biomass), where greens represent above-average vegetation productivity and reds represent below-average productivity.

The highest positive departure values (greens and yellows) were observed in the Atchafalaya, Mississippi River, upper Mermentau and Barataria, and parts of the Teche-Vermilion and Bayou Biloxi basins. These areas either received freshwater and sediment through natural riverine processes, marine transport of river constituents, or through diverted river waters. Moderate to low positive departures (orange) were observed in a majority of the Calcasieu-Sabine and Mermentau basin wetlands. Negative departures in NDVI values (red) were observed in the upper Penchant, and lower Terrebonne and Barataria basins. Though these departures were negative, they were relatively small, and potentially due to floating aquatics, flood duration, and salinity shifts within wetlands (Coastal Protection and Restoration Authority [CPRA] of Louisiana 2017). Overall, the departure from average results demonstrate the direct response of vegetation to floods, and further corroborate previous finding of hydrologic connectivity.

Landscape Metrics

Figure 2.9 illustrates the Aggregation Index (AI) rates of change (slope) for each VB, which were computed using 16 wetland and water classified Landsat images from 1988 to 2013. The AI rates of change provide a measure of landscape condition that correlates to wetland integrity and stability (Suir *et al*. 2013; Couvillion *et al*. 2016). Though the AI slopes ranged from -0.24 (red) to 0.01 (dark green), the majority of VB units experienced decreasing AI rates over this period. Only the Atchafalaya Fresh and Teche-Vermilion Saline units experienced positive rates of AI. Rates for these units were most closely matched by the Barataria Fresh and Mississippi River delta units, which experienced small decreasing rates of AI. Most VB units experienced moderately negative AI rates, except for the more saline Terrebonne and Barataria units, and the fresher Breton Sound units, which experienced the largest negative rates of change. The more stable AI rates in Atchafalaya, Teche-Vermilion, and Mississippi River units are anticipated results with their proximity to large river influence. However, stable AI in the upper Barataria is less expected, but potentially due to river inputs through the GIWW and Davis Pond Diversion (Figure 2.2), and its inland position, which limits the erosive pulses and presses that are active in the intertidal zone. The less stable AI rates in lower Terrebonne and Barataria basins are also anticipated since these regions are highly sediment deprived and have been subjected to salinity intrusion, oil and gas access canal impacts, and accelerated rates of subsidence (Sasser *et al*. 1986). The less stable AI in the upper Breton Sound basin is potentially due to Hurricane Katrina impacts (Barras 2005). Figure 2.9 also illustrates the range of AI and wetland stability in the GZ units. Overall, Middle Coast wetlands retained higher levels of aggregation and spatial integrity. Chenier Plain wetlands experienced moderately decreasing aggregation, while the wetlands of the Deltaic Plain experienced a wider range of aggregation change.

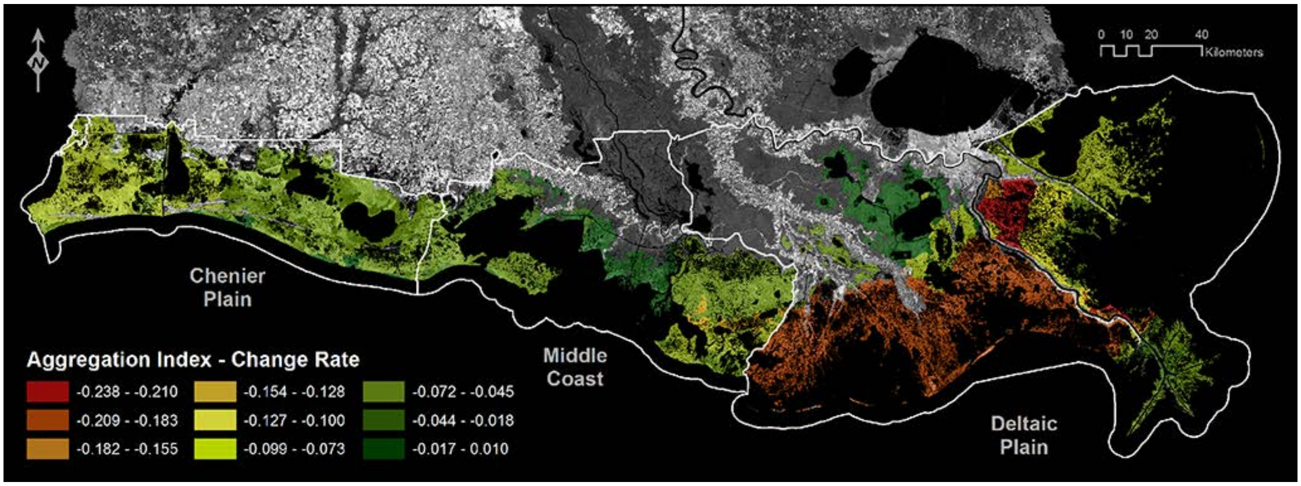

Figure 2.9. Landsat derived Aggregation Index mean change rate (1988 to 2013) by Vegetation by Basin assessment unit and assessed by geomorphological zone. Dark green areas represent wetland landscapes with highest stability and red areas with lowest stability.

Multi-Metric Evaluation

Aggregation Index provides temporal and spatial measures of wetland structure that, combined with Class Area (CA) and NDVI values, allow for assessments of additional linkages between river connectivity with wetland productivity and spatial integrity. Figure 2.10 illustrates the mean NDVI values (green bars, above and below axis represents positive and negative values, respectively), along with the CA and AI change rates (red and yellow, respectively) for each of the VB units. The majority of VB zone wetlands exhibited moderate CA, NDVI, and AI values and rates across the period of analysis, with few zones near the minima and maxima. Similar relative tends exist across these metrics, with wetland areas in close proximity to large river influence, or those that receive river sediment via marine processes, having the highest productivity and stability.

Conversely, VB zones with lesser river influence, especially those with higher susceptibility (i.e., energy, altered hydrology, and subsidence) experienced the lowest productivity and stability.

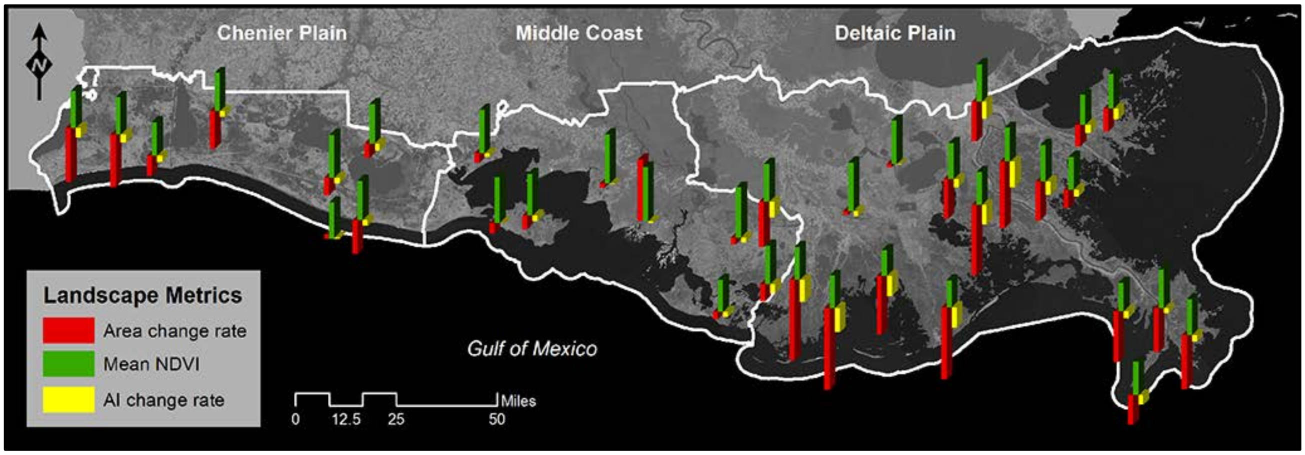

Figure 2.10. Landsat derived mean Normalized Difference Vegetation Index (NDVI) values, and Class Area (CA) and Aggregation Index (AI) change rates, for each vegetation by watershed basin assessment unit.

While each metric provides separate measures of wetland structure or function, individually, they lack the ability to link causal mechanisms to, and relationships between, wetland productivity and stability. Figure 2.11 plots the mean NDVI against the mean AI for all VB across the 1988 to 2013 period to evaluate correlations between wetland productivity and spatial integrity. The plot shows moderate correlations (r^2 = 0.558) between NDVI and AI, where higher NDVI values typically return higher AI values. Anomalies occur in areas where major hurricanes have impacted wetland productivity, which has been shown to affect NDVI values, even over long time periods (Li *et al.* 2016).

To consider correlations between river connectivity with wetland function and structure, data points within Figure 2.11 were color coded based on GZ. The mean NDVI and AI values by VB unit within the Chenier Plain, Middle Coast, and Deltaic Plain are represented by the orange, green, and red points, respectively. The two blue data points represent Deltaic Plain assessment units that receive AR inputs via the GIWW (Terrebonne Fresh) or AR and MR inputs via the GIWW and Davis Pond Diversion (Barataria Fresh). Figure 2.11 shows the higher NDVI and AI combinations are dominated by the Middle Coast VB, the moderate NDVI and AI combinations are dominated by the Chenier Plain VB, and the lower NDVI and AI combinations are dominated by the Deltaic Plain VB units. These trends show wetlands with highest river connectivity typically are the most productive and stable. One exception is the Mississippi River Delta, which accumulates more sediment than all other VB units, however, current sedimentation is insufficient in offsetting the combined effects of altered hydrology, salinity fluxes, wind- and wave-induced erosion, and the high rates of compaction and subsidence (Gagliano *et al*. 1981).

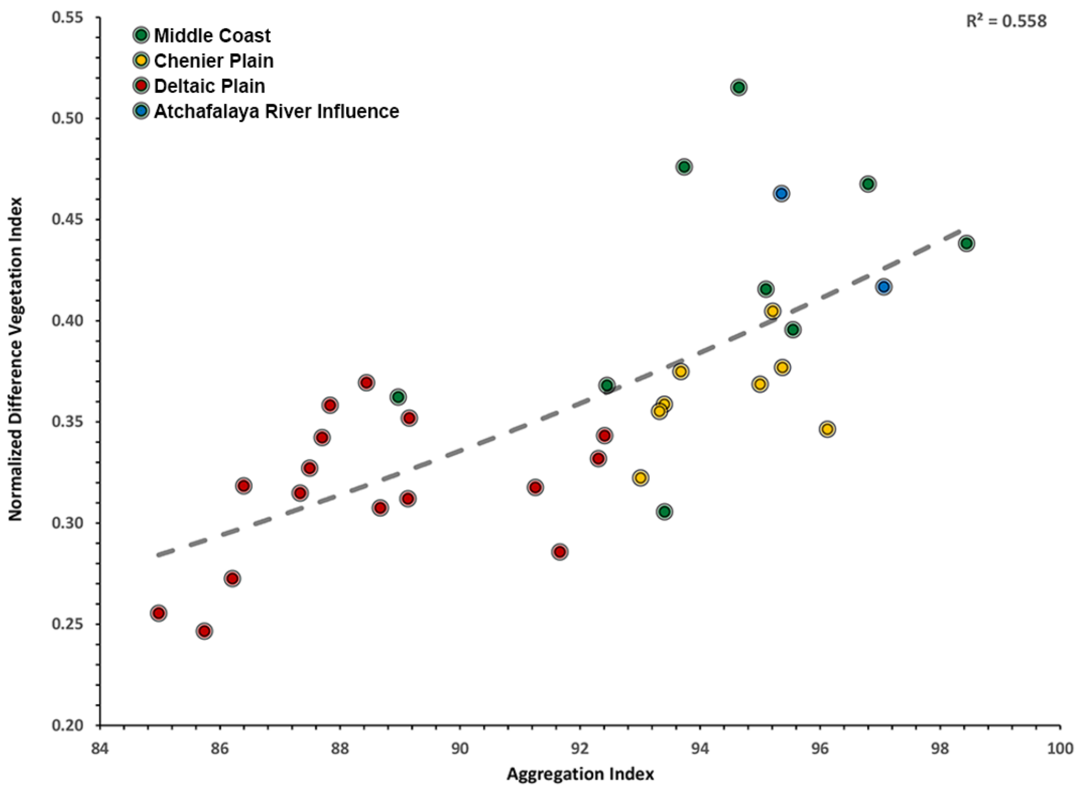

Figure 2.11. Coastwide plots of mean Normalized Difference Vegetation Index versus mean Aggregation Index for all vegetation by watershed basin assessment units. The Chenier Plain, Middle Coast, and Deltaic Plain are represented by the orange, green, and red dots, respectively. The blue dots represent assessment units that receive river inputs from distant sources.

CONCLUSIONS

Remote sensing and landscape analyses provided enhanced techniques for evaluating river influence on biomass and correlations to spatial integrity. MODIS- and Landsat-derived NDVI, CA, and AI—integrated with river discharge, river sediment concentration, and accretion data—were used to perform multi-temporal and -spatial scale assessments to differentiate wetland productivity and stability based on proximity to large river systems. Louisiana wetland productivity is highly associated with seasonality and vegetation zones, susceptible to episodic events (hurricanes and floods), and significantly correlated to river connectivity. This was observed under baseline conditions, post-major flood events, and across short and long periods of observation. Similarly, positive correlations between landscape stability and river influence were observed. Ultimately, these assessments validate assumptions that wetland productivity and stability are at least partial functions of river connectivity. Though wetland loss is often the combined effects of subsidence, energy, saltwater intrusion, and human activities, sediment deprivation has been shown to be a primary driver in the long-term degradation of wetland structure and function. Continued evaluations of wetland productivity and landscape configuration, along with other ecosystem drivers, will provide a greater understanding of river and sediment importance for wetland stability and restoration.

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CHAPTER 3 – COMPARING CARBON ACCUMULATION IN RESTORED AND NATURAL WETLAND SOILS OF COASTAL LOUISIANA

INTRODUCTION

Recent and future-projected effects from climate change has stimulated the need to reduce greenhouse gas (GHG) sources and to increase GHG sinks to help mitigate those effects. Wetlands provide numerous ecosystem goods and services ranging from protecting and improving water quality, providing critical habitat, storing floodwaters, to providing important biogeochemical processes where nutrients, organic compounds, metals, and components of organic matter are transformed and stored (Brady and Weil 1996; Osland *et al*. 2012; Reddy and DeLaune 2008). Though they only occupy approximately 5% of the Earth's surface, wetlands represent the largest component (40%) of the terrestrial biological carbon (C) pool (~ 2,500 petagram [Pg]), and are important links in the sequestration of carbon and cycling of atmospheric gases (Armentano and Menges 1986; Chmura *et al*. 2003; Hossler and Bouchard 2010; Lal and Pimentel 2008; Mitsch and Gosselink 2000; Mitsch *et al*. 2013). Carbon sequestration in wetland systems consist of the rapid accumulation and storage of soil organic matter (SOM) in wetland sediments (Bridgham *et al*. 2006; Mcleod *et al*. 2011; Sifleet *et al*. 2011). Also, these sediments provide anaerobic, acidic, and thermal conditions that result in the sequestration of carbon for much longer periods than other systems (Burkett and Kusler 2000).

North American wetlands account for 42% of the global carbon pool (Bridgham *et al*. 2006). This sheer abundance, in addition to the potential and critical nature of carbon sequestration (i.e., buffers the emissions of GHGs from soil to the atmosphere), make SOM one of the Nation's most important resource (Albrecht 1938; Lal 2004). The SOM content within wetland systems are primarily driven by processes such as biodegradation, photochemical oxidation, sedimentation, volatilization, and sorption (Kayranli 2010). Since these processes are highly dependent on wetland health and productivity, the release of stored carbon to the atmosphere is significantly increased when wetland conditions degrade (Lane 2016).

With extreme reductions in wetlands worldwide, approximately 50% of wetlands have been loss since 1900, many wetland goods and services are at risk (Davidson 2014). Wetlands, through natural function and losses, have significantly contributed to GHG emissions, accounting for approximately 15% to 40% of the annual global CH_4 flux per year (Ehhalt *et al*. 2001; Poffenbarger *et al*. 2011). In the United States, early wetland loss was dominated by the draining and conversion of wetlands to agricultural lands, which accelerated oxidation of stored carbon and its release to the atmosphere as CO_2 (Armentano and Menges 1986). In Louisiana, which accounts for approximately 40% of the Nation's wetlands, but 90% of its loss, marsh deterioration has resulted in massive organic matter loss through the exportation to estuaries and offshore areas, and subsequent carbon release through oxidation (Couvillion *et al*. 2011; DeLaune and White 2011; Williams 1995).

To remediate these losses, many ecosystem stakeholders have advanced protection and restoration strategies to reestablish critical wetland goods and services. One relatively new strategy is to utilize wetland creation and restoration measures to increase soil organic carbon (SOC) density, distribution, and stability in the soil (Lal 2004). However, in many cases carbon sequestration and storage are secondary benefits or “added value” of wetland restoration. Uncertainties persist about the long-term linkages between climate change and wetland processes, especially in restored systems (Chmura *et al*. 2003; Edwards and Proffitt 2003). Though some created wetlands have been shown to quickly achieve vegetative equivalency to naturally occurring target wetlands (specifically when sites are planted), most require decades or longer (especially with SOC accumulation), or, they never achieve equivalency (Edwards and Proffitt 2003; Hogan *et al*. 2004; Hossler and Bouchard 2010; Moreno-Mateos *et al*. 2012; Osland *et al*. 2012).

For more informed resource management decisions, considerable research is needed to evaluate and compare the rates of carbon sequestration in restored ecosystems to naturally occurring reference wetlands (Loomis and Craft 2010). Therefore, the purpose of this study was to evaluate the influence that wetland ecosystem management and restoration have on carbon sequestration potential and chronosequence. This was accomplished by comparing organic material, bulk density, carbon content, and rates of accumulation between various ages and types of restored and reference wetlands. The specific objectives of this study were to (1) compile a comprehensive set of all relevant restoration project and soils data, (2) map the spatial distribution of relevant wetland soils characteristics, (3) compute carbon sequestration rates for restored and natural wetland sites, (4) compare soil function across type and age of restoration measure, and (5) evaluate implications for future restoration and climate change.

METHODS

Study Area and Assessment Units

The Louisiana coastal zone is dominated by histosol wetlands that occupy an ecological niche across unique ranges of condition and function, ranging from riverine-influenced fresh and brackish areas to “Blue Carbon” marshes nearest the coast where salinities above 17 part per thousand (ppt) reduce the production of methane and other GHGs to negligible amounts (Chambers *et al*. 2013; DeLaune *et al*. 2013). To evaluate key wetland functions, this study utilized soil samples, data from scientific literature, and those collected as part of multiple restoration and monitoring programs and projects (Figure 3.1). Qualifying samples and sites were selected from wetland restoration areas (i.e., wetland creation, terracing, hydrologic alteration, freshwater diversion, sediment diversion) and target reference areas where soil nutrient analyses have been performed or where soil cores were available for analyses. The Program sites consisted primarily of Coastwide Reference Monitoring System (CRMS) and Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA) monitoring stations. These stations are part of large-scale data collection and monitoring systems that were developed to characterize and compare wetland hydrology, ecology, soil, and geomorphology conditions across project and non-project areas throughout coastal Louisiana (Steyer *et al*. 2003; Wang *et al*. 2017). The size and density of these data sets offer unprecedented opportunities for studying coastal wetland dynamics and subtle soil processes and interactions (Jankowski *et al*. 2017). The Program sites were supplemented with Project sites, where samples were collected (by the authors and others) within the following sites: (1) Sabine Refuge Marsh Creation project, (2) Wax Lake Delta (DeLaune *et al*. 2016), (3) Atchafalaya Big Island Mining creation project, (4) Davis Pond Freshwater Diversion (DeLaune *et al*. 2013), (5) Bayou Labranche Wetland Creation project (Richardi 2014), and (6) Little Lake Marsh Creation project (Figure 3.1).

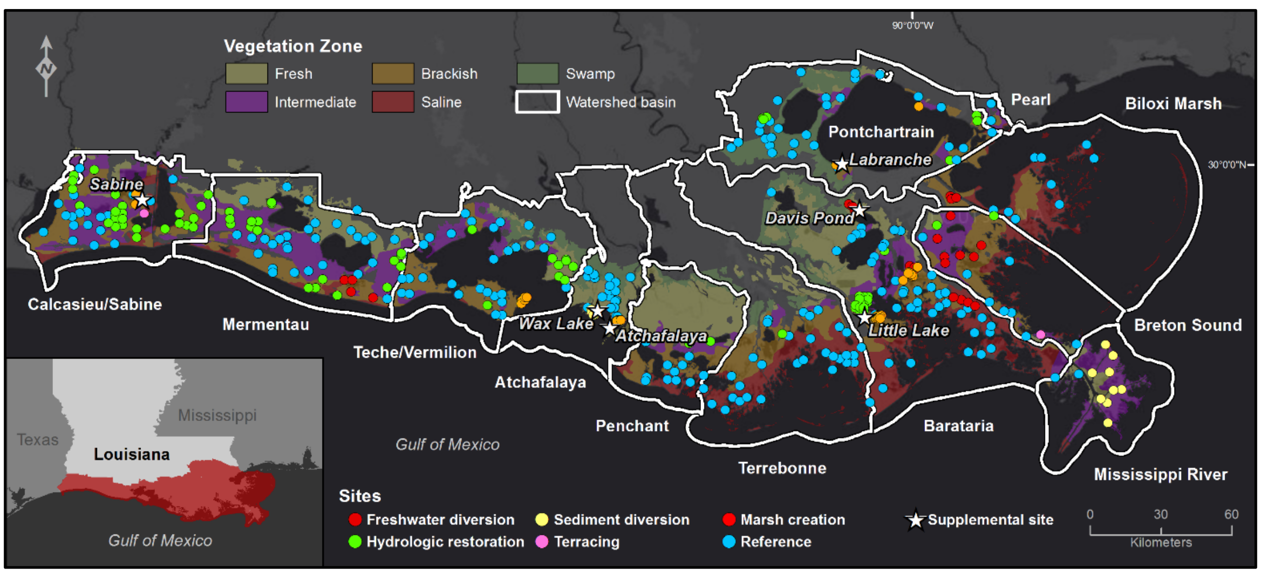

Figure 3.1. Location map of the Assessment Units (Coastal Zone, Watershed Basins, and Vegetation Zones), Program sites (dots), and supplemental Project sites (stars) in coastal Louisiana.

The coastal zone was divided into multi-scale assessment units to evaluate potential correlations between soil function and restoration type, whilst considering geomorphic and hydrologic settings. These assessment units include (1) Coastal Zone (CZ), (2) Watershed Basins

(WB), (3) Vegetation Zones (VZ), and (4) Vegetation by Basin units (VB) (Figure 3.1). This allowed for spatial and chronosequence approaches (i.e., space-for-time substitution) to evaluate impacts and age of restoration on soil function. Mean relative short-term (feldspar) and longer-term (decadal from cesium data) vertical accretion, bulk density, organic matter, carbon content, and short-term carbon accumulation rates were calculated and evaluated for all assessment units.

Soil Acquisition, Sampling, and Analysis

Soils data utilized in this study consisted of those from the scientific literature, from previously collected Program soils (Coastal Protection and Restoration Authority [CPRA] of Louisiana, 2017), or were sampled from select Project sites. Project soil cores were collected from restoration and reference stations at the Sabine (Oct 2015), Wax Lake (Jun 2013), Atchafalaya (Sep 2015), Davis Pond (2011), Bayou Labranche (multi-year), and Little Lake (Nov 2014 and Oct 2015) study sites. Subsurface soils at the Project sites were sampled with a 5-cm diameter thin walled aluminum corer (with a sharpened edge) to a depth of 15-cm. This depth typically contains the highest soil carbon content and is a reasonable proxy for use in standard carbon estimation (Jenkins *et al*. 2010). The Program sites are typically sampled with 10.2-cm diameter corers to a depth of 30-cm, and sliced into 4-cm increments. The standard for both Project and Program samples were to place soils into labelled sealable storage bags and transport to LSU or contracting laboratories for processing.

Key soil characteristics and processes in the marsh soils were determined using techniques previously reported by DeLaune *et al*. (2013). Short- and longer-term vertical accretion rates were also extracted from Program repositories and scientific literature, respectively. The short- and longer-term (decadal) vertical accretion data were calculated using the Feldspar marker and ^{137}Cs methods described in Folse *et al*. (2014) and DeLaune *et al*. (1978), respectively. For bulk density determinations, subsamples were oven-dried to a constant weight at 60º Celsius. For soil organic matter (SOM) percentage, the Walkley-Black acid-dichromate oxidation method was used for the

Project soils (Nelson and Sommer 1982), and the Loss On Ignition (LOI) method was used for the Program soils (Andrejko *et al*. 1983). The LOI method is a quicker and less expensive alternative to other methods, and is a reliable and suitable method for soil C analysis (Wright 2008). For each Program site, data from the top segments (0 to 16-cm) were averaged for congruity with Project samples. The SOM measurements were transformed to total carbon content (percentage) by dividing by a factor of 1.724 and multiplying by 100 (Allen 1974; Craft *et al*. 1991). The carbon sequestration rates (gC m^{-2} y^{-1}) for each site were calculated by multiplying the short-term sediment accretion rate (cm^3/y) by the soil bulk density (g/cm^3) and then by the carbon content (percentage) (Bernal and Mitsch 2013). With recent efforts to standardize carbon sequestration and GHG emission units, the gC m^{-2} y^{-1} rates were also converted to CO_2 equivalents (CO_2e). Every 12g of carbon (atomic mass is 12g/mol) is equal to 44g of CO_2 (atomic mass is 44g/mol), therefore the sequestration rates were converted to CO_2e by multiplying the gC m^{-2} y^{-1} rate by 44gCO_2e and dividing by 12gC (Sifleet *et al*. 2011). ESRI ArcGIS was used to manage, analyze, and map the spatial distribution of these wetland soil attributes, and their differences, over space and time. These data were used as general measures of restoration impacts on carbon fluxes, primarily through sequestration. Though wetlands also emit GHG, which can be a major component of the carbon cycle and influence or counteract sequestration rates, GHG flux assessments were beyond of the scope of this study.

Statistical Analyses

In order to attain comparability among soil attributes and rates for each assessment scale, statistical analyses were conducted using Statistical Analysis System software version 9.2. The PROC GLM procedure was used to perform a one-way analysis of variance (ANOVA) and a means separation test (Tukey's, $\alpha = 0.05$) to evaluate significance of differences between soil attributes for each assessment units. Additionally, a second order polynomial regression with coefficient of determination (r^2) was used to evaluate correlations between soil attributes and age of restoration.

RESULTS AND DISCUSSION

Coastal Zone

Soil measurements were calculated using 1,224 data points from across the coastal zone of Louisiana. The collective means and standard deviations (independent of project type, geomorphology, hydrology, and age) for select soil characteristics (i.e., bulk density, organic matter, total carbon, short-term accretion [feldspar], longer-term accretion [^{137}Cs], and short-term carbon accumulation) are provided in Table 3.1.

Table 3.1. Average bulk density, organic matter, total carbon, accretion, and carbon accumulation from all sites within the Louisiana coastal zone.

		Bulk Density	Organic Matter	Total Carbon	Short Term Accretion (Feldspar)	Longer Term Accretion (^{137}Cs)†	Short Term Carbon Accumulation	
Coastal Zone	Count	g/cm^3	percent	percent	cm y^{-1}	cm y^{-1}	gC m^{-2} y^{-1}	gCO_2e
Mean	1224	0.299	33.919	19.67	1.03	0.79	371.9	1363.6
Std	1224	0.245	21.11	12.24	0.80	0.36	294.7	1080.6

† Longer term accretion data from Byrant and Chabreck 1998; Day *et al*. 2012; DeLaune *et al*. 1989; DeLaune *et al*. 1992; Foret 1997; Foret 2001; Nyman *et al*. 1993; Rybczyk and Cahoon 2002; Sasser *et al*. 2002; Swenson and Turner 1994.

Sites within the coastal zone had significantly higher short-term accretion rates (mean 1.03 ± 0.8 cm y^{-1}) than longer-term (decadal) rates (mean 0.79 ± 0.36 cm y^{-1}). Soil data with adequate core depths were not available for the computation of longer-term carbon accumulation rates. However, since shorter-term (feldspar) accretion rates are largely similar to longer-term accretion rates (Tables 3.2-3.4), they provide a good estimate of longer-term carbon accumulation. Average short-term carbon accumulation rates for each sample site are provided in Figure 3.2. This figure illustrates the range and distribution of carbon accumulation rates across the coastal zone, where lower rates were observed in the west (Chenier Plain), higher rates occurred in the "Middle Coast", and a wider range of rates occurred in the eastern portion of the Deltaic Plain. Overall, the average carbon accumulation rate for the coastal zone was 371.88 ± 294.7 gC m^{-2} y^{-1} (1363.56 ± 1080.57 gCO_2e) (Table 3.1). This is above the average rate (118 gC m^{-2} y^{-1}; 432.67 gCO_2e) of carbon sequestration for wetlands throughout the world (Mitsch *et al*. 2013), and is indicative of the highly productive and functioning wetlands in coastal Louisiana.

Watershed Basin

Previous small-scale studies have reported increased carbon sequestration with increasing river connectivity due to decreasing mineralization of soil organic matter (Wang and Dodla 2013). Watershed basins, delineated primarily on hydrologic connectivity, were used as large-scale assessment units for evaluating general hydrologic influence on carbon accumulation (Suir *et al*. in review). The mean values for key soil characteristics are provided in Table 3.2. Mean bulk density ranged from 0.19 g cm^{-3} to 0.79 g cm^{-3} for Mermentau and Mississippi River basins, respectively. For total carbon, the means ranged from 4.1% to 26.2% for Mississippi River and Mermentau basins, respectively. Comparisons of short- and long-term accretion rates agree with previous studies which show that over time accretion slows (Sadler 1981; Smith 2009). The sediment in basins that receive larger river inputs (i.e., Atchafalaya, Mississippi River, Penchant, Vermilion-Teche) (Suir *et al*. in review) had significantly ($p<0.05$) higher carbon accumulation rates than those with lower inputs (i.e., Biloxi Marsh, Calcasieu/Sabine, Mermentau). The Mississippi River basin had the highest mean carbon accumulation rate (585 ± 476 gC m^{-2} y^{-1}; 2145 ± 1745.3 gCO_2e), and Calcasieu/Sabine basin had the lowest (132 ± 111 gC m^{-2} y^{-1}; 484 ± 407 gCO_2e). The general tendency in these data show a correlation between carbon accumulation and river hydrogeomorphology, where higher carbon accumulation rates are generally found in hydrogeomorphic zones or watershed basins with the highest hydrologic connection to high flow and high sediment rivers.

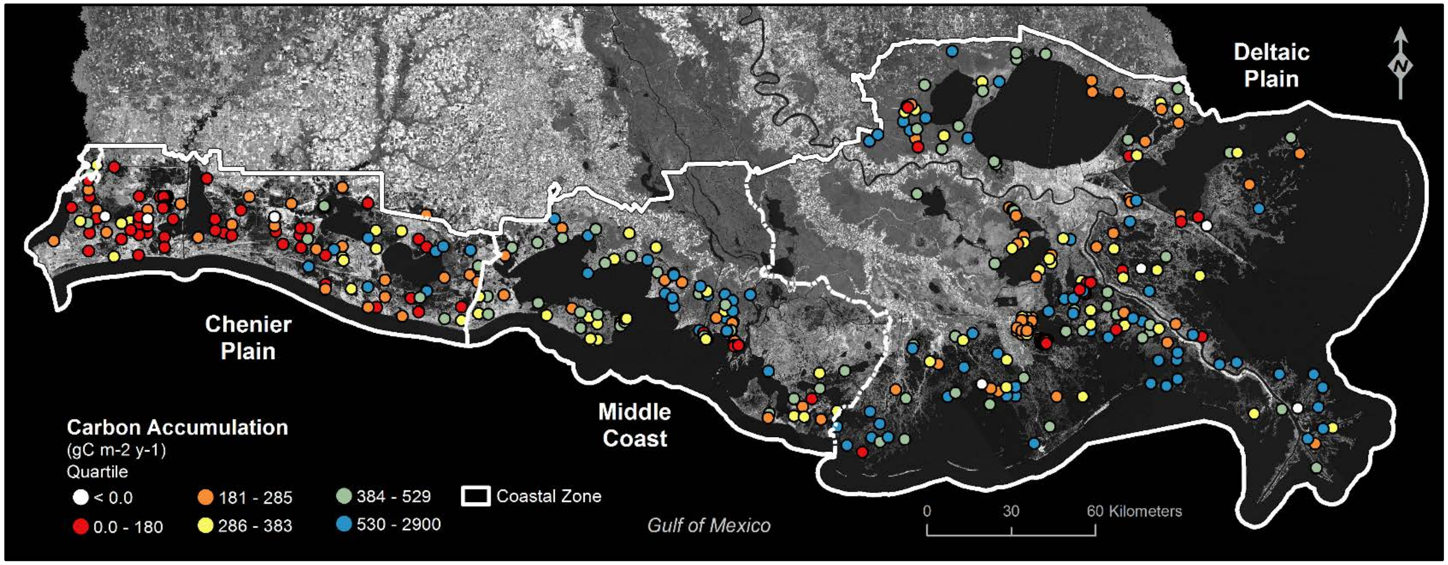

Figure 3.2. Average short-term (feldspar) carbon accumulation rate for all sites within the Louisiana Coastal Zone assessment unit.

Table 3.2. Average bulk density, organic matter, total carbon, accretion, and carbon accumulation from sites within each basin unit.

Basin	Count	Bulk Density	Organic Matter	Total Carbon	Short Term Accretion (Feldspar)	Longer Term Accretion (^{137}Cs)†	Short Term Carbon Accumulation	
		mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std
		g/cm^3	percent	percent	cm y-1	cm y^{-1}	gC m^{-2} y^{-1}	gCO_2e
Atchafalaya	90	0.38 ± 0.3	19.5 ± 13.1	11.34 ± 7.6	1.57 ± 1.04	1.43 ± 0.29	436 ± 293	1599 ± 1074
Barataria	233	0.3 ± 0.31	37.8 ± 25	21.9 ± 14.5	1.31 ± 0.89	0.76 ± 0.19	451 ± 389	1654 ± 1426
Biloxi	36	0.43 ± 0.27	21.5 ± 12.9	12.48 ± 7.5	0.72 ± 0.85	0.65 ± 0.09	256 ± 343	939 ± 1258
Breton Sound	54	0.33 ± 0.17	26.4 ± 12.9	15.34 ± 7.5	1.02 ± 0.83	0.81 ± 0.35	414 ± 390	1518 ± 1430
Calcasieu-Sabine	148	0.22 ± 0.2	41.5 ± 19.7	24.04 ± 11.4	0.4 ± 0.3	0.41 ± 0.13	132 ± 111	484 ± 407
Mermentau	156	0.19 ± 0.13	45.2 ± 21.6	26.22 ± 12.6	0.72 ± 0.39	0.69 ± 0.18	272 ± 180	997 ± 660
Mississippi River	30	0.79 ± 0.28	7 ± 3	4.06 ± 1.7	2.18 ± 1.94	1.9 ± 0.11	585 ± 476	2145 ± 1745
Pearl	9	0.29 ± 0.07	24.6 ± 5	14.29 ± 2.9	0.95 ± 0.18	0.78 ± 0	374 ± 77	1371 ± 282
Penchant	39	0.33 ± 0.13	23.2 ± 9.7	13.46 ± 5.6	0.85 ± 0.33	0.85 ± 0.36	321 ± 112	1177 ± 411
Pontchartrain	179	0.27 ± 0.2	39.3 ± 19.7	22.78 ± 11.5	0.92 ± 0.36	0.73 ± 0.28	406 ± 234	1489 ± 858
Terrebonne	99	0.27 ± 0.12	30.9 ± 15.9	17.92 ± 9.2	1.34 ± 0.9	0.83 ± 0.2	501 ± 302	1837 ± 1107
Vermilion-Teche	150	0.36 ± 0.22	27.4 ± 17.1	15.9 ± 9.9	1.05 ± 0.29	0.8 ± 0.18	408 ± 136	1496 ± 499

† Longer term accretion data from Byrant and Chabreck 1998; Day *et al*. 2012; DeLaune *et al.* 1989; DeLaune *et al*. 1992; Foret 1997; Foret 2001; Nyman *et al*. 1993; Rybczyk and Cahoon 2002; Sasser *et al*. 2002; Swenson and Turner 1994.

Vegetation Zone

Short-term carbon accumulation rates in the surface layer (~15 cm) of wetlands are largely driven by net primary productivity (above and below-ground biomass) and microbial decomposition (Baustian *et al*. 2017; Bernal *et al*. 2008; Kayranli *et al*. 2010; Powlson 2011). Since primary productivity is significantly correlated to salinity (i.e., vegetation zone) (Steyer 2008; Suir *et al*. in review), carbon accumulation rates were evaluated across the vegetation zones in coastal Louisiana. The means and standard deviations of key soils characteristics by vegetation zone are provided in Table 3.3. The carbon accumulation rates ranged from a low of 300 ± 254 gC m^{-2} y^{-1} (1100 ± 931.3 gCO_2e) for the Intermediate zone, to a high of 468 ± 247 gC m^{-2} y^{-1} (1716 ± 905.67 gCO_2e) for the Swamp zone. The means of carbon accumulation rates in the Saline zone were significantly different ($p<0.05$) than those in the Brackish and Intermediate zones, while those in the Swamp zone were significantly different than the Brackish, Fresh, and Intermediate zones. Though the carbon accumulation rates in the Saline and Swamp zones were significantly higher than other zones, no definitive relationship was observed between carbon accumulation rate and changes in salinity (vegetation zone). These findings corroborate those from previous small-scale studies, which demonstrated carbon accumulation rates were similar in various marsh types in coastal Louisiana (DeLaune and White 2011; Hatton *et al*. 1982; Nyman *et al*. 2006).

Vegetation Zone by Watershed Basin

To assess the potential combined influence of salinity and hydrogeomorphology on carbon accumulation, evaluations were performed using vegetation zone by watershed basin (VB) units. Figure 3.3 provides a schematic of mean carbon accumulation rates for reference sites by VB, represented as polygons (white represents lowest rates, black represent the highest rates, and hatched areas contained no reference sites), and the rates for restoration projects are represented by dots (graduated dots correlate to range of rate). It should be noted that though these change rates are represented using different symbologies (dots for restoration and polygons for reference sites) in Figure 3.3, they were both derived using discretely collected soils data. It should also be noted that the dots representing the rates of change for the restoration sites are centroids (within each VB zone) and do not represent exact sample locations. This is an important consideration since wetland loss and function are variable and nuances of impacts across smaller areas, especially within the inactive portions of the Deltaic Plain.

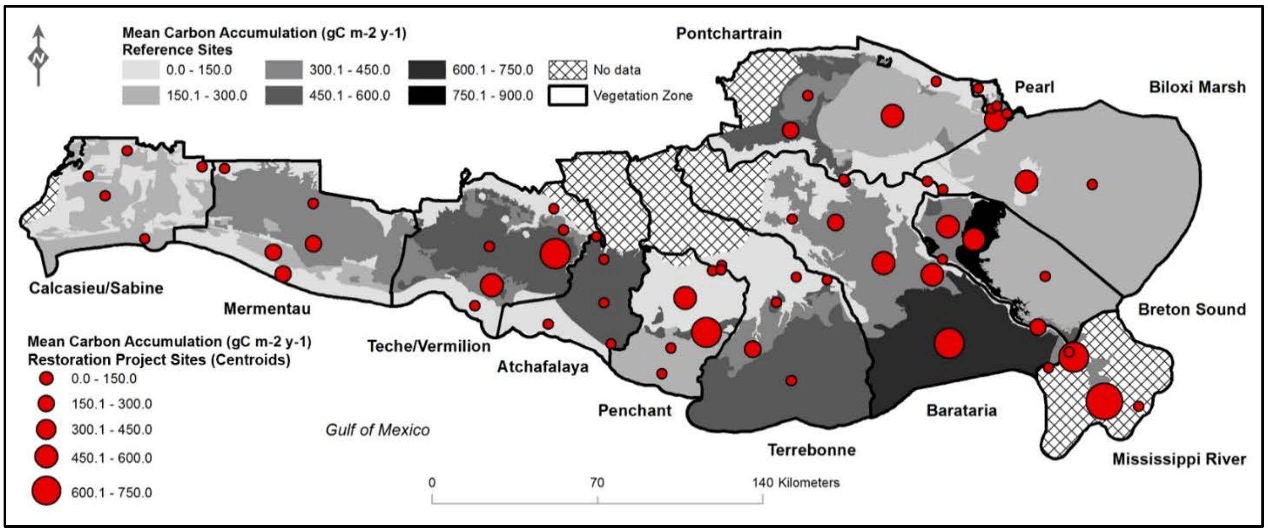

Figure 3.3. Average short-term (feldspar) carbon accumulation within Vegetation Zone by Basin assessment units for reference sites (polygons, lighter gray represents lower rates and darker gray represents higher rates) and restoration sites (red dots, smaller dots represent lower rates and larger dotes represent higher rates).

The average carbon accumulation by VB for reference sites ranged from 141 ± 95 gC m^{-2} y^{-1} (517 ± 348.3 gCO_2e) for the Calcasieu-Sabine brackish zone to 804 ± 612 gC m^{-2} y^{-1} (2948 ± 2244 gCO_2e) for the Breton brackish zone. The reference sites with the highest mean carbon accumulation (darkest polygons) were those that are either in zones of high river connectivity or consist of swamp or higher salinity tolerant marsh (Suir *et al*. in review). The average carbon accumulation by VB for project sites ranged from 31 ± 3 gC m^{-2} y^{-1} (113.67 ± 11 gCO_2e) for the Calcasieu-Sabine fresh zone to 646 ± 424 gC m^{-2} y^{-1} (2368.67 ± 1554.67 gCO_2e) for the Mississippi River intermediate zone. The

VB zones with the highest project rates of carbon accumulation (large points in Figure 3.3) were generally correlated to VB zones with highest reference site rates. This demonstrates the influence of local geomorphic, hydrologic, and coastal processes on wetland function. The variability in carbon accumulation rates across VB zones are largely driven by salinity, riverine inputs (i.e., nutrients and sediments), and whether a project site receives extended benefits (i.e., diversions) or benefits from multiple restoration measures (e.g., marsh creation site receiving added benefits from sediment or freshwater diversions).

Restoration Project

Soils of newly constructed or restored wetlands initially retain properties more typical of the terrestrial (or source) soils from which they were created, and they generally take decades to achieve functional equivalency to naturally occurring wetlands (Edwards and Proffitt 2003; Hogan *et al*. 2004). Comparisons of carbon accumulation rates between restoration and reference sites, and between restoration measures, were conducted. The mean values of bulk density, organic matter, total carbon, accretion, and carbon accumulation are provided in Table 3.4. When compared collectively, restoration sites had higher bulk densities and accretion rates, and lower SOM and TC, than reference sites, though none were significantly different (Table 3.4). The reference sites did have significantly higher short-term carbon accumulation rates, averaging 415.17 ± 300.1 gC m^{-2} y^{-1} (1522.29 ± 1100.37 gCO_2e) compared to the restoration sites, which averaged 302.29 ± 271.75 gC m^{-2} y^{-1} (1108.4 ± 996.42 gCO_2e).

The average carbon accumulation rates by restoration type ranged from a low of 240 ± 151 gC m^{-2} y^{-1} (880 ± 553.37 gCO_2e) for hydrologic restoration to a high of 520 ± 490 gC m^{-2} y^{-1} (1906.67 ± 1796.67 gCO_2e) for sediment diversions. The hydrologic restoration measure had significantly lower carbon accumulation rates than the reference, sediment diversion, and freshwater diversion sites, and the sediment diversion sites had significantly higher rates than the marsh creation sites ($p<0.05$). Though the average total carbon (g kg^{-1}) at the sediment diversion sites was significantly lower than all other restoration types, the higher bulk density and accretion rates for the sediment diversion measure resulted in higher carbon accumulation rates.

Carbon Accumulation by Age

Previous studies have reported that wetlands can be both a source and sink of carbon depending on ecosystem condition and age (DeLaune *et al*. 2016; Kayranli *et al*. 2010). The relationships between carbon accumulation and project maturity were evaluated for all restoration sites and for each restoration type. The general trends observed were slight increases in carbon accumulation with age for the freshwater diversion ($y = 18.9x + 69$) and hydrologic restoration ($y = 7.5x + 147.2$) measures, slight decrease for marsh creation ($y = -2.3x + 344.1$) sites, and considerable decreases for the terracing ($y = -50.9x + 903.6$) and sediment diversions ($y = -99x + 1792.4$) measures. Except for the terracing sites, which consisted of only two temporal data points ($r^2 = -0.99$), carbon sequestration rates were not significantly correlated ($r^2 < 0.08$) with age. Overall, the trend in carbon accumulation for all restoration sites showed a slight increase ($y = 3.8x + 262.6$) with age. These findings corroborate previous small-scale studies that have reported gradual increasing carbon accumulation in restored or created wetlands over time, and approximately several decades of maturity before restored or created wetlands reach functional equivalency to natural or reference wetlands (Moreno-Mateos *et al*. 2012).

Though the chronosequences examined may be too short (<25 y) to investigate the maturity required for wetland restoration sites to reach equilibrium with reference wetland functions, they do provide a general trajectory of carbon accumulation rate for all restoration sites, and rates for each wetland restoration type, over time.

Table 3.3. Average bulk density, organic matter, total carbon, accretion, and carbon accumulation from sites within each vegetation zone assessment unit.

		Bulk Density	Organic Matter	Total Carbon	Short Term Accretion	Longer Term Accretion	Short Term Carbon Accumulation	
Vegetation Zone	Count	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std
		g/cm^3	percent	percent	cm y^{-1}	cm y^{-1}	gC m^{-2} y^{-1}	gCO_2e
Fresh	171	0.28 ± 0.29	38.1 ± 26.2	22.1 ± 15.2	1.27 ± 1.11	1.02 ± 0.46	376 ± 268	1379 ± 983
Intermediate	293	0.24 ± 0.21	39.6 ± 21.3	23.0 ± 12.3	0.86 ± 0.65	0.76 ± 0.43	300 ± 254	1100 ± 931
Brackish	353	0.32 ± 0.28	32.1 ± 20.1	18.7 ± 11.7	0.96 ± 0.54	0.7 ± 0.22	345 ± 261	1265 ± 957
Saline	215	0.35 ± 0.19	23.2 ± 11.2	13.4 ± 6.5	1.16 ± 1.13	0.72 ± 0.2	435 ± 413	1595 ± 1514
Swamp	153	0.28 ± 0.21	39.5 ± 20.4	22.9 ± 11.8	1.03 ± 0.43	0.9 ± 0.37	468 ± 247	1716 ± 906
Other	38	0.39 ± 0.24	26.0 ± 22.0	15.1 ± 12.8	1.23 ± 0.76	0.88 ± 0.35	408 ± 195	1496 ± 715

† Longer term accretion data from Byrant and Chabreck 1998; Day *et al*. 2012; DeLaune *et al.* 1989; DeLaune *et al*. 1992; Foret 1997; Foret 2001; Nyman *et al*. 1993; Rybczyk and Cahoon 2002; Sasser *et al*. 2002; Swenson and Turner 1994.

Table 3.4. Average bulk density, organic matter, total carbon, accretion, and carbon accumulation for reference and restoration sites.

		Bulk Density	Organic Matter	Total Carbon	Short Term Accretion	Longer Term Accretion	Short Term Carbon Accumulation	
Project	Count	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std	mean ± std
		g/cm^3	percent	percent	cm y^{-1}	cm y^{-1}	gC m^{-2} y^{-1}	gCO_2e
Reference	754	0.28 ± 0.18	33.4 ± 18.7	19.37 ± 10.9	1.07 ± 0.8	0.81 ± 0.33	415 ± 300	1522 ± 1100
Restoration	469	0.33 ± 0.32	34.8 ± 24.5	20.17 ± 14.2	0.97 ± 0.8	0.78 ± 0.4	302 ± 272	1107 ± 997
Fresh Diversion	73	0.37 ± 0.4	32.7 ± 20	18.99 ± 11.6	1.05 ± 0.6	0.64 ± 0.14	377 ± 295	1382 ± 1082
Hydro Restoration	243	0.14 ± 0.09	50.5 ± 19.4	29.31 ± 11.2	0.69 ± 0.38	0.68 ± 0.22	240 ± 151	880 ± 554
Marsh Creation	114	0.52 ± 0.31	11.8 ± 9.5	6.83 ± 5.5	1.19 ± 0.62	0.75 ± 0.42	324 ± 324	1188 ± 1188
Sed. Diversion	33	0.84 ± 0.31	6.3 ± 3.4	3.67 ± 2	2.04 ± 1.99	1.82 ± 0.22	520 ± 490	1907 ± 1797
Terracing	6	0.46 ± 0.07	14.5 ± 3.7	8.41 ± 2.2	0.92 ± 0.77	0.86 ± 0.59	318 ± 252	1166 ± 924

† Longer term accretion data from Byrant and Chabreck 1998; Day *et al*. 2012; DeLaune *et al.* 1989; DeLaune *et al*. 1992; Foret 1997; Foret 2001; Nyman *et al*. 1993; Rybczyk and Cahoon 2002; Sasser *et al*. 2002; Swenson and Turner 1994.

CONCLUSIONS

The net balance of carbon in wetland systems is largely driven by hydrology (flooding regime), plants species, climate, soil OM decomposition (mineralization), and salinity (Bernal *et al*. 2008; Kayranli *et al*. 2010; Mitsch 2013). However, there are data gaps and conflicting results regarding key wetland and climate change interactions. This study set forth to compile and map the spatial distribution of relevant wetland soil characteristics, evaluate key presses, functions, and chronosequence of restored and naturally occurring wetland soils, and assess implications for future ecosystem restoration and climate change. This was accomplished by utilizing an exceptionally large data set to perform the first coastwide assessment of carbon accumulation in Louisiana wetlands. Carbon accumulation rates in the Louisiana coastal zone were generally correlated to hydrogeomorphology and distinctive trends were observed within the Chenier Plain, Middle Coast, and Deltaic Plains. Comparisons of carbon accumulation within smaller-scale assessment units revealed higher rates generally occurred in zones of high river connectivity or in swamp or higher salinity tolerant marsh. Naturally occurring wetlands had significantly higher carbon accumulation rates than the average of all restoration sites, though the sediment diversion sites had significantly higher accumulation rates than all other sites.

Putting these results in the context of other studies, the high rates of accumulation in the high-salinity marsh was likely influenced by reduced methanogenesis in this traditional Blue Carbon ecosystem, whereas the lower salinity zones of high river connectivity and swamps probably emitted GHGs, but the rates were outstripped by the high levels of biological productivity in these systems (Gough and Grace 1998; Steyer 2008; Janousek and Mayo 2013). Future research considering a broader suite of GHG fluxes could further elucidate these patterns. A more thorough understanding of carbon fluxes in existing and restorable coastal wetlands is important because of the symbiotic relationship that wetland processes have with climate change. The fate of carbon in natural and restored wetlands will be increasingly constrained by sea-level rise, salinity, and temperature, which in turn will be increasingly regulated by carbon cycling in wetlands. For instance, increasing temperatures will result in increased GHG emissions (wetlands become a major source of GHG), which in turn contribute to global warming (Kayranli *et al*. 2010). Many aspects of these processes are unknown, especially in the long-term function of restored wetlands. Few existing wetland restoration projects have the required age for adequate evaluation of function equivalency, therefore, future research should consider the use of analogs over longer periods of analyses for chronosequence assessments. Also, since the amount of carbon sequestration and release via numerous GHGs can be shifted by moderate changes to wetland systems, future studies should incorporate emissions measurements of at least CO_2 and CH_4 to provide a more complete assessment of carbon processes and balance within wetland restoration landscapes. Wetland restoration provides many opportunities to incorporate ecosystem structural and functional services. Though carbon sequestration is a relatively new focus of wetland restoration missions, it may prove to be one of the most critical for climate change management.

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CHAPTER 4 – REDISTRIBUTION AND IMPACTS OF NEARSHORE BERM SEDIMENTS ON THE CHANDELEUR BARRIER ISLANDS, LOUISIANA

INTRODUCTION

Few areas in coastal Louisiana have suffered more drastic changes to shoreline position, geometry, and configuration than the barrier islands (Martinez *et al*. 2005a). Long-term interactions of sea level rise, subsidence, wave and storm damage, and oil and gas activities have resulted in extensive erosion and narrowing, which have led to a decrease in island habitat and reductions in storm-related surge and wave protection (Martin 2010; CPRA 2017). Most of these sensitive barrier systems are on the verge of collapse and are highly susceptible to additional pressures, including anthropogenic activities.

On 20 April 2010, the United States encountered one of the largest marine oil spills in the Nation's history. This accidental spill, caused by the failure of the Macondo-252 well and explosion of the Deepwater Horizon oil rig, resulted in the estimated release of 185 million gallons of crude oil into the Gulf of Mexico (Wilde and Skrobialowski 2011). As part of the emergency response plan, the Louisiana Office of Coastal Protection and Restoration (LOCPR) proposed sand berms to reduce the amount of oil reaching barrier islands, thereby protecting these highly sensitive ecosystems. Approximately 25.75 kilometers of sand berms were constructed at two locations, the "Western Barrier Berm" (WBB seaward of Shell, Pelican, and Scofield Islands) and the "Eastern Barrier Berm" (EBB offshore, nearshore, and onshore of the northern Chandeleur Islands) (Figure 4.1). Construction of the berms, which began and ended in June 2010 and March 2011, respectively, required approximately 4.7 million cubic meters (m^3) of sand, and cost an estimated $250 million (Louisiana Legislative Auditor 2011, LOCPR 2015, CPE 2013a, 2013b, 2013c, 2013d). The berms were constructed with a +1.83 m North American Vertical Datum (NAVD88) crest elevation, 6.1 m crest width, and slopes of 1V:25H above and 1V:50H below the 121.9 m base at the -0.61 m elevation.

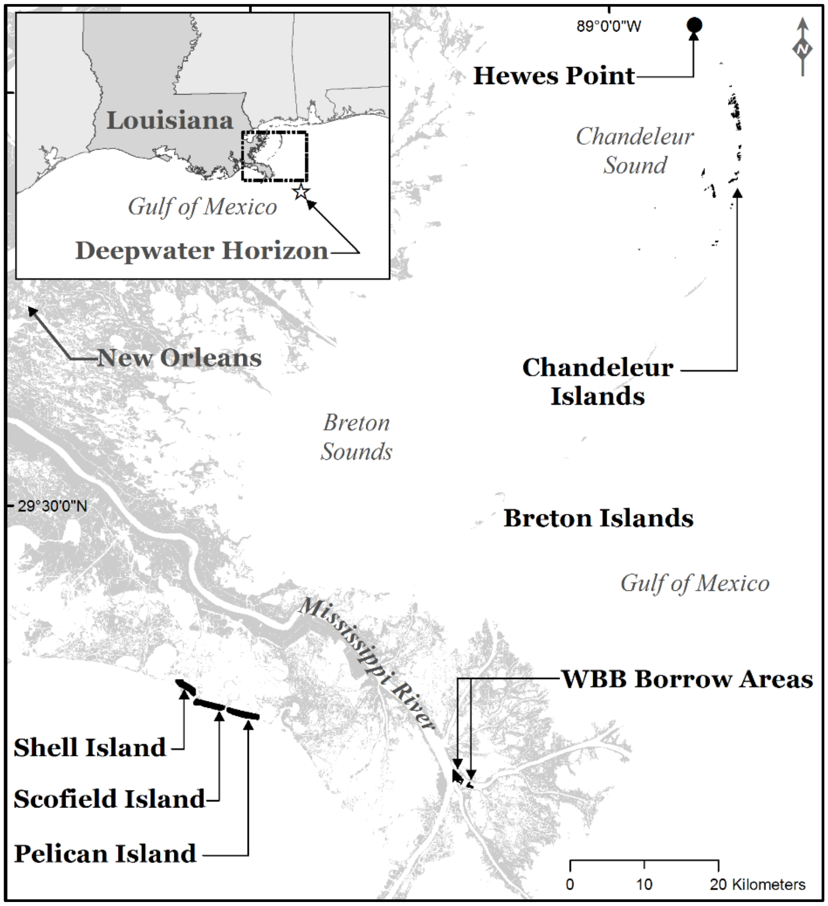

Figure 4.1. Location map depicting the Deepwater Horizon explosion site (inset), the Western Barrier Berm (Shell, Scofield, and Pelican Islands), Eastern Barrier Berm (Chandeleur Island), and sand borrow locations.

Though the sand berms were constructed with the sole purpose of retarding oil from reaching sensitive ecosystem resources, they mimic nearshore beneficial use of dredged material (BUDM) applications that are utilized for barrier island nourishment and restoration. Nearshore berm placements are becoming an increasingly preferred option for dredged material placement since they typically have lower cost, restrictions, and design and construction complexity (Wang *et al*. 2016). However, since there are various site-specific issues associated with nearshore BUDM, and since there have not been many extensive studies on the performance of various types of nearshore placements, key knowledge gaps exist related to the evolution and impacts of nearshore berms (Wang *et al*. 2016). The oil-spill-mitigation berms provide multiple unique BUDM scenarios, including offshore, nearshore, and onshore placements of sediment in high energy environments. These scenarios provide useful conditions for evaluating the stability and resilience of constructed sand berms to marine and coastal processes, and the impacts berm sediments have on wetland extent, productivity, and quality—all within the context of historical and recent conditions of Louisiana's barrier islands. Recent studies have quantified and evaluated the stability of the oil-spill-mitigation sand berms, but the redistribution and impacts of those sediments on island habitat and vegetation are still uncertain. Therefore, the objectives of this study were to (1) evaluate the redistribution of EBB sediment within the northern Chandeleur Island system; (2) assess EBB impacts by evaluating the quantity and quality of existing and new emergent vegetation as a function of redistributed sediment; and (3) consider the implications of this research on future island restoration and nourishment.

METHODS

Study Area

Though some preliminary comparisons were made between the EBB and WBBs, the primary study area consisted of the northern Chandeleur barrier island chain, approximately 96 kilometers east of New Orleans, Louisiana, and approximately 137 kilometers north of the Deepwater Horizon spill (Figure 4.1). The northern island arc has historically been dominated by wide beaches and large washover fans that were primarily vegetated by upland shrubs and grasses, and high salinity (20 to 40 parts per thousand) marshes consisting largely of *Spartina alterniflora* (smooth cordgrass), *Spartina patens* (wiregrass), *Juncus roemerianus* (needlegrass rush), *Disticlis spicata* (saltgrass), and *Avicennia germinans* (black mangrove) (Fearnley *et al*. 2009; Hymel 2007; Kahn and Roberts 1982). The islands have experienced a long-term reduction in sand volume, a trend that has been accelerated by hurricanes, resulting in rapid erosion of island features (beach, dune, and marsh platforms) and the inability to maintain many subaerial features (US Army Corps of Engineers 2012).

Evaluations of berm and vegetation condition were performed using various assessment units, including the berm-specific footprint, and island-wide discrete vegetation sample locations and bounding assessment area (Figure 4.2). The berm footprint, which was delineated using the approximately 122 m wide slope break portion of the fill area (above -0.61 m elevation NAVD88), was used to assess berm performance (Suir *et al*. 2016). For island-wide assessments, multiple data sets consisting of discretely collected vegetation survey data were used to assess vegetation quantity, quality, and change over time (Figure 4.2). The island bounding assessment area consisted of approximately 457 m seaward and inland from the island centerline, and was used for raster-based analyses of sediment distribution and vegetation extent and productivity (Figure 4.2).

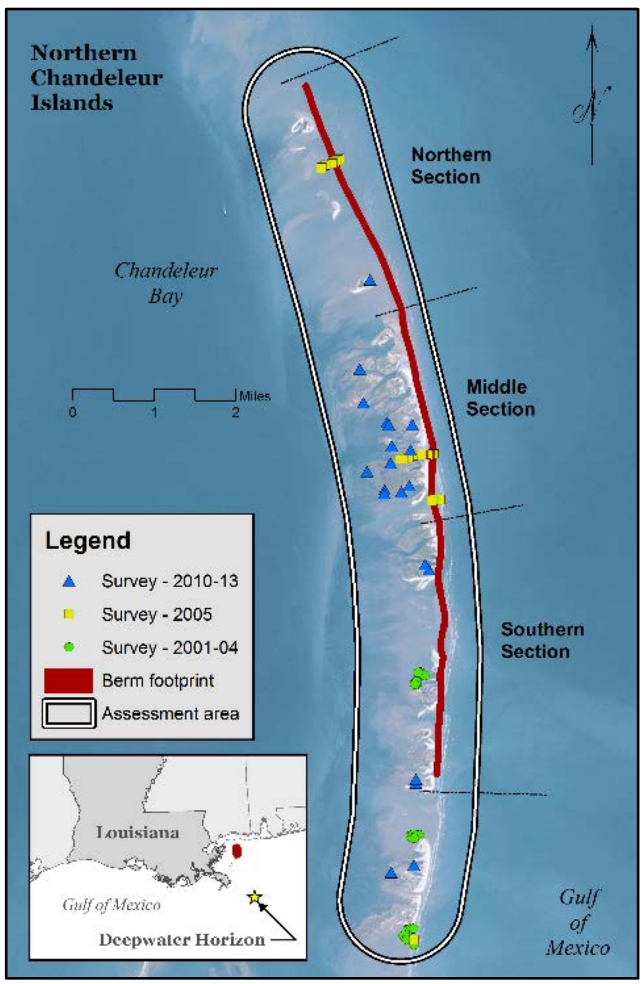

Figure 4.2. Study area assessment units, consisting of discrete sample sites, the berm footprint, and the larger island assessment area.

Geographic Information Systems and Remote Sensing

Given the remote location and limited field access to the Chandeleur Islands, Geographic Information Systems (GIS) and remote sensing data and techniques were ideally suited for this study. Remote sensing provides a means for classifying landscape features to assess the distribution and change of those features over time. The spatial analyses performed in this study evaluated the evolution of the oil-mitigation-berms and the impacts of berm sediment on island features and vegetation. The assessments of berm performance and sediment redistribution consisted of a synthesis of recent studies and elevation evaluations using newly acquired aerial photography, high resolution space-borne imagery, and light detection and ranging (LiDAR) data (Table 4.1). Assessments of habitat and vegetation utilized Barrier Island Comprehensive Monitoring (BICM) program habitat data (Fearnley *et al*. 2009; Martinez *et al*. 2005, 2006), existing National Wetlands Inventory (NWI) data (USGS 1980a, 1980b; USGS 2004), newly derived habitat and land-water data, satellite imagery derived Normalized Difference Vegetation Index (NDVI) data, and vegetation survey data. In addition to these data sets, other existing geospatial data (i.e., historical maps) and reports were used as ancillary interpretive information.

Berm Performance

The ability of the sand berms to maintain their form, in both elevation and length, were critical components for satisfying the primary project goal of retarding oil from reaching barrier island resources. A synthesis of recent berm studies and published data (i.e., elevation, length, and volumetric changes) were used to evaluate berm performance and evolution (CB&I 2013d; Plant and Guy 2013a, 2013b, 2013c; Suir *et al*. 2016). These berm performance assessments utilized survey elevation data, air- and space-borne imagery for length and segmentation evaluations, and volume changes to estimate overwash and longhore transport of sediment.

Table 4.1. Catalog of the data acquired, collected, and utilized for eastern barrier berm assessments.

Data	Collection	Source	Program/System/Sensor	Resolution/Accuracy
Elevation	Oct-01	USGS/NASA 2009	Airborne Topographic Mapper Lidar	+/-15 cm
	Dec-05	Reif *et al*. 2011	Compact Hydrographic Airborne Rapid Total	+/-20 cm
	Jun-07	Sallenger *et al*. 2015	Experimental Advanced Airborne Research Lidar	+/-15 cm
	Mar-10	Nayegandhi *et al*. 2010	Experimental Advanced Airborne Research Lidar	+/-15 cm
	Jun-11	USACE 2013	Compact Hydrographic Airborne Rapid Total	+/-20 cm
	Feb-12	Guy *et al*. 2014	Leica Geosystems Airborne ALS60	+/-15 cm
	Jul-12	CPE 2103a,b,c,d	RTK GPS	Discrete
	Jul-13	Guy and Plant 2014	Leica Geosystems Airborne ALS70	+/-18 cm
	Feb-15	USGS 2016	Leica Geosystems Airborne ALS80	+/-15 cm
Habitat	1956	USGS 1980a	NWI, Aerial Photography	1:24000
	1978	USGS 1980b	NWI, Aerial Photography	1:24000
	1988	USGS 2004	NWI, Aerial Photography	1:24000
	1998	Fearnley *et al*. 2009	BICM, Aerial Photography	1:12000
	2004	Fearnley *et al*. 2009	BICM, Aerial Photography	1:24000
	2005	Fearnley *et al*. 2009	BICM, Aerial Photography	1:40000
	2008	USFWS 2016	NWI, Digital Mapping Camera	1:10000
Vegetation	1992	Handley *et al*. 2007	Color Infrared Aerial Photography	1:32500
	2010	NOAA 2012	Aerial Photography	1:24000
	2004	CPRA 2016	CWPPRA PO-27	Discrete
	2005	Martinez *et al.* 2006	BICM	Discrete
	2013	BP 2004	Gulf Science Data	Discrete
Imagery	3/22/2004	DigitalGlobe	QuickBird-2	2.4 m multispectral
	1/25/2005	DigitalGlobe	QuickBird-2	2.4 m multispectral
	10/17/2005	DigitalGlobe	QuickBird-2	2.4 m multispectral
	6/6/2007	DigitalGlobe	QuickBird-2	2.4 m multispectral
	9/11/2008	USGS 2009	Digital Mapping Camera	1:10000
	5/24/2010	DigitalGlobe	WorldView-2	1.84 m multispectral
	10/14/2010	DigitalGlobe	WorldView-2	1.84 m multispectral
	5/6/2011	DigitalGlobe	QuickBird-2	2.4 m multispectral
	5/7/2011	USGS 2011	Digital Mapping Camera	30 cm
	8/31/2011	DigitalGlobe	QuickBird-2	2.4 m multispectral
	10/27/2011	DigitalGlobe	QuickBird-2	2.4 m multispectral
	3/4/2012	DigitalGlobe	QuickBird-2	2.4 m multispectral
	12/18/2012	DigitalGlobe	GeoEYE	1.65 m multispectral
	11/12/2013	USDA/FSA 2013	Digital Mapping Camera	1:12000
	3/1/2014	DigitalGlobe	GeoEYE	1.65 m multispectral
	5/5/2014	DigitalGlobe	GeoEYE	1.65 m multispectral
	12/10/2014	DigitalGlobe	WorldView-2	1.84 m multispectral
	2/10/2015	DigitalGlobe	WorldView-3	1.24 m multispectral
	4/6/2015	DigitalGlobe	GeoEYE	1.65 m multispectral
	12/14/2015	DigitalGlobe	WorldView-2	1.84 m multispectral
	2/19/2016	DigitalGlobe	WorldView-3	1.24 m multispectral
	4/22/2016	DigitalGlobe	WorldView-3	1.24 m multispectral
	5/5/2016	DigitalGlobe	WorldView-3	1.24 m multispectral
	6/21/2016	DigitalGlobe	WorldView-2	1.84 m multispectral
	7/7/2016	DigitalGlobe	WorldView-2	1.84 m multispectral

Sediment Redistribution

Currents, tides, waves, and wind energies are forces consistently eroding and redistributing barrier island sediments. The redistribution of berm sediments within the island system were assessed using LiDAR-based elevation data, existing NWI data, and habitat data that were generated using multispectral air- and space-borne imagery (Figure 4.1).

Elevation

Table 4.1 provides the source, program, and accuracy specifications of the eight bare earth LiDAR data sets (2001, 2005, 2007, 2010, 2011, 2012, 2013, and 2015) that were used to evaluate elevation change across the northern Chandeleur Islands. LiDAR was also used to bracket the construction of the berm to assess spatial distribution of elevation changes on existing and newly formed island features. Digital Elevation Models (DEMs), with vertical resolution of approximately ± 15 to 20 cm, were produced from the remotely sensed and geographically referenced LiDAR elevation measurements. These data provide accurate and highly detailed measurements of subaerial (bare earth) and in some instances, shallow subaqueous (shoal) Chandeleur Island features. To compute elevation differences between paired LiDAR data, the Spatial Analyst Minus tool was used in ArcGIS version 10.5. This tool was used to subtract values of the end-date raster from values of the begin-date raster on a cell-by-cell basis, producing an output raster containing difference or change values (ESRI 2015). These LiDAR based assessments were used to evaluate berm evolution (i.e., breaching and erosion) and the redistribution of berm sediments within the island system.

Habitat

A modified National Wetlands Inventory (NWI; Cowardin *et al*. 1979) classification scheme was used to evaluate the redistribution of berm sediments and develop a map of existing emergent vegetation features, while separately classifying unconsolidated features within various tidal regimes. This modified classification scheme, which consists of land, water, and unconsolidated shore classes, provides increased site- or condition-specific interpretations of target habitats. Within this classification scheme the “land” class consists of all uplands, dunes, vegetated dunes, emergent vegetation, and scrub-shrub features. The “water” class consists of open water and pond features. The “unconsolidated shore” class consists of substrate that fall in three traditional NWI subclasses, (1) irregularly flooded, (2) regularly flooded, and (3) irregularly exposed bottoms.

Comparing trends in historical and recently formed land and unconsolidated shore habitats provide measures of berm sand redistribution. The pre-construction assessments consisted of decadal or greater time-periods and the post-construction analyses generally consisted of shorter time-steps. However, the periods of analysis were ultimately based on data availability. The pre-construction data consisted of multiple dates of NWI sets (1956, 1978, and 1988) and multiple sets of BICM habitat data (1998, 2004, and 2005) (Table 4.1). All pre-construction habitat data were modified to conform to the land, water, and unconsolidated class scheme. The post-construction habitat data were derived using high resolution 2008 (USGS 2009), 2011 (USGS 2001), 2013 (USDA/FSA 2013), and 2015 (DigitalGlobe GeoEYE) imagery (Table 4.1).

The ratio and interface of land and water are some of the more important features and metrics of wetland landscapes. Therefore, this study also utilized land-water classification methodologies to evaluate wetland area change and trends during the post-construction period and compared those to the pre-construction (derived from habitat data) period. The satellite-based methodology is a variant of the standard procedures used for Coastwide Reference Monitoring System (CRMS) based land-water classifications (Folse *et al*. 2014). This classification process utilized the Normalized Difference Water Index (NDWI) and the NDVI (method details provided in the vegetation section below) to identify water and wetland features, respectively. The traditional NDWI (McFeeters 1996), which normalizes a green band against a near infrared (NIR) band, is described by the following equation:

$$\mathrm{NDWI} = \frac{Green - \mathrm{NIR}}{Green + \mathrm{NIR}}. \quad (1)$$

Recoding of the NDWI and NDVI (with and without edge enhancement) thematic files were performed through an overlay process. Clump and eliminate functions were then performed on each recoded file to reduce noise (Braud and Feng 1998; Suir *et al*. 2011). A final overlay was performed in which the NDWI and NDVI images were aggregated and recoded to single files with wetland and water class. To evaluate the accuracy of land-water classifications, confusion matrices and Kappa values were computed for all paired habitat-based and satellite-derived wetland and water data.

Vegetation

Plant species composition, cover, density, and biomass are structural components of coastal marshes that are commonly used to quantify vegetative characteristics and serve as indicators of wetland condition (Chamberlain and Ingram 2012; Cretini *et al*. 2012). This study used standard vegetation assessments (i.e., distribution and composition), the Floristic Quality Index (FQI), and the Normalized Difference Vegetative Index (NDVI) as primary measures of condition and function. The FQI is a metric traditionally used to identify and monitor critical landscapes, assess impacts from disturbance events, measure wetland ecological condition, and evaluate habitat restoration (Bourdaghs *et al*. 2006; Fennessy *et al*. 2002; Gianopulos 2014). Similarly, the NDVI is commonly used to provide estimates of above-ground biomass, primary productivity and wetland species distributions, and assess impacts from anthropogenic activities and episodic events (An *et al*. 2013; Bianchette *et al*. 2009; Klemas 2013; Steyer *et al*. 2013).

A modified FQI, one which incorporates invasive species, percent cover values, and accounts for total percent cover and overlapping canopies, was used to evaluate pre- and post-construction trends in Chandeleur Island wetlands. The FQI provides an estimate of habitat quality based on a measure of vulnerability, called the Coefficient of Conservatism (CC), and the richness or cover of a plant community (Gianopoulos 2014). CC values range from zero (not conservative) to ten (conservative and highly ecologically sensitive). Plant species are typically assigned CC values (within a local flora and by a panel of experienced botanists) based on the following characteristics: invasive plant species (CC value of 0), disturbance species (CC = 1 to 3), vigorous wetland communities (CC = 4 to 6), common species (CC = 7 to 8), and dominant wetland species (CC = 9 to 10) (Bourdaghs *et al*. 2006). The FQI uses a two-pronged approach to account for sample units with vegetation cover that is less than or equal to 100%, or is greater than 100% (overlapping canopies). If the sum of species covers within a sample unit at time *t* is less than or equal to 100, the applicable formula is as follows:

$$\mathrm{FQI}_{\mathrm{mod}\ t} = \left(\frac{\sum\ (\mathrm{COVER}_{it}\ \mathrm{x}\ \mathrm{CC}_{i})}{100} \right)\ \mathrm{x}\ 10, \tag{2}$$

where $\mathrm{FQI}_{\mathrm{mod}\ t}$ is the modified floristic quality index (unitless), COVER_{it} is the percent cover (%) for species *i* at a sample unit, within a sample site, at time *t*, and CC_i is the Coefficient of Conservatism for species *i* (Cretini *et al*. 2011).

By using 100 in the denominator (instead of the actual sum of species covers), differentiation between wetlands of similar composition (e.g., vigorous wetlands) can be made using normalized biomass (estimated through cover) (Cretini *et al*. 2012). For consistency with other restoration program (i.e., CRMS and Coastal Wetlands Planning, Protection, and Restoration Act [CWPPRA]) metrics and indices, the FQI values are multiplied by 10 to scale the scores from 0 to 100 (Cretini *et al*. 2011).

If the sum of species covers within a sample unit at time *t* is greater than 100, the applicable formula is:

$$\mathrm{FQI}_{\mathrm{mod}\ t} = \left(\frac{\sum\ (\mathrm{COVER}_{it}\ \mathrm{x}\ \mathrm{CC}_{i})}{\sum\ (\mathrm{TOTAL\ COVER}_{t})} \right)\ \mathrm{x}\ 10, \tag{3}$$

where $\mathrm{TOTAL\ COVER}_t$ refers to the percent cumulative species cover (expressed as a percentage) within a sample unit (Cretini *et al*. 2012).

The FQI assessments utilized existing vegetation survey data that were collected as part of the CWPPRA program (2001 to 2004), the BICM program (2005), and the BP Gulf Science initiative (2010 to 2013), which are represented by the green dots, yellow squares, and blue triangles in Figure 4.2, respectively. The CC values and FQI equations (Eqs. 2 and 3), developed for Louisiana plant species (Cretini *et al*. 2011, 2012), were used to calculate average FQI values for each CWPPRA, BICM, and BP vegetation survey station. For species not on the Louisiana Coefficient of

Conservatism list (Cretini *et al*. 2011), established values from regional lists or neighboring states were used in conjunction with best judgement (Herman *et al*. 2006; Mortellaro *et al*. 2012; Gianopulos 2014).

NDVI assessments were performed using GeoEYE, Quickbird, and WorldView satellite imagery collected during the pre- and post-construction periods (Table 4.1). These satellite images provide high spatial (1.24 to 2.4 m multispectral) and temporal (1-2 day sensor returns) resolution data that are useful for estimating short-term landscape variation linked to disturbance events and/or prevailing environmental conditions (Suir *et al*. 2011). All high-resolution satellite data were acquired using the DigitalGlobe Enhanced Viewer Web Hosting Service. ENVI version 5.3 was used to perform radiometric corrections on all satellite imagery. Optical inspections were then performed to identify and remove satellite images of poor quality. NDVI data were created using the standard equation (Rouse *et al*. 1974):

$$\mathrm{NDVI} = \frac{NIR2 - Red}{NIR2 + Red}, \tag{4}$$

which utilizes a ratio between a near-infrared (NIR) and red band to measure an ecosystem's ability to capture solar energy and convert it to organic carbon or biomass (An *et al*. 2013). The NDVI has well established correlations to photosynthetic activity, aboveground biomass, and leaf area index (Carle 2013).

NDVI values range from -1 to 1, where those between -1 and zero (0) are typical of non-vegetation features (e.g., water, clouds, and impervious surfaces), and those between 0 and 1 are typical of vegetated features. The higher the NDVI value the higher, generally, the biomass, productivity, and vigor of the vegetation. All non-marsh features within the study area were excluded from each image by removing all NDVI values less than zero (< 0) (Reif *et al*. 2011). ESRI ArcGIS 10.5 was used to calculate zonal statistics (i.e., mean, min, max, standard deviation) on values of each NDVI raster within the Chandeleur Island assessment area (ESRI 2015).

Statistical Analyses

In order to attain comparability among values by year, statistical analyses were conducted using Statistical Analysis System software version 9.2. The PROC GLM procedure was used to perform a one-way analysis of variance (ANOVA) and a means separation test (Tukey's, $\alpha = 0.05$) to evaluate significance of differences between elevation, habitat, and vegetation values by year or period. Additionally, a linear regression with coefficient of determination (r^2) was used to evaluate correlations between elevation, habitat, and wetland change over periods of analysis.

RESULTS AND DISCUSSION

Berm Evolution

Table 4.2 provides the pre-construction, as-built, and 30-, 90-, 180-, and 360-day post-construction survey elevation summary statistics for the EBB (Chandeleur Island) and WBB (Shell, Pelican, and Scofield Islands). The Chandeleur Island berm construction, which used approximately 2,400,000 m^3 of high quality quartz sand, increased the mean pre-construction height (within footprint) by 1.13 m (CPE 2013d). By the 360-day survey, approximately 24.1% of the Chandeleur berm sediments were lost beyond the project footprint (CPE 2013d). These losses were similar to those observed in the WBB, which experienced 41.6, 25.3, and 22.1% loss of sediment from the Shell, Pelican, and Scofield berms, respectively. However, not all segments of the EBB evolved equally. Previous research shows that over the course of a year the northern (offshore) section of the EBB experienced extensive thinning, breaching, and sediment loss, while the middle (nearshore) and southern (onshore) sections experienced more localized movement of sediment, resulting in widening of the sand berm (Plant *et al*. 2013a, b, c; Suir *et al*. 2016). All berm segments were significantly impacted by tropical storms. Tropical Storm Lee (26.4 meters per second [m s-1] and storm surge of approximately 1.2 m) made landfall on 3 September 2011 and Hurricane Isaac (35.7 m s-1 and storm surge of approximately 3.3 m) made landfall on 28 August 2012. Tropical Storm Lee initiated breaching and overwashing of the EBB, and Hurricane Isaac resulted in the full eradication of all subaerial features (Brown 2011; Berg 2013; CPE 2013d).

Table 4.2. Field survey-based elevation and volume change along the western and eastern berm.

				Elevation (m NAVD)				Volume Change	
Berm	**Survey**	**Date**	**Count**	**Min**	**Max**	**Mean**	**STD**	**Within Period**	**Percent of As-built**
Shell	Pre	Jul-10	1069	-1.89	-0.12	-0.99	0.46	-	-
	As-built	Dec-10	304	-1.04	1.80	0.59	0.65	751,847	
	30	Feb-11	632	-1.55	1.58	0.38	0.70	-96,631	
	90	Apr-11	571	-1.74	1.49	0.20	0.77	-87,437	-41.6
	180	Jul-11	1098	-1.55	1.25	0.48	0.56	131,533	
	360	Jan-11	724	-1.62	1.10	-0.07	0.79	-	
Pelican	Pre	Jul-10	1571	-1.92	1.10	-0.81	0.47	-	-
	As-built	Oct-10	684	-1.19	1.95	0.75	0.76	936,580	
	30	Jan-11	659	-1.28	1.71	0.49	0.73	-	
	90	Mar-11	740	-1.62	1.58	0.53	0.63	20,731	-25.3
	180	Jul-11	1190	-1.62	1.55	0.56	0.63	20,136	
	360	Jan-12	1229	-1.58	1.34	0.36	0.64	-	
Scofield	Pre	Jul-10	2477	-1.80	1.01	-0.78	0.43	-	-
	As-built	Dec-10	675	-1.80	1.74	0.19	0.91	900,647	
	30	Feb-11	1420	-1.95	1.65	0.25	0.83	52,227	
	90	May-11	1930	-1.58	1.55	-0.18	0.84	-	-22.1
	180	Aug-11	1791	-1.52	1.49	0.11	0.81	268,356	
	360	Feb-12	1697	-1.49	1.46	-0.02	0.76	-	
Chandeleur	Pre	Jul-10	1828	-3.29	0.40	-0.66	0.58	-	-
	As-built	May-11	1957	-3.35	2.38	0.47	0.82	2,306,6	
	30	Jul-11	5613	-1.98	2.26	0.61	0.75	289,567	
	90	Sep-11	3419	-2.41	2.04	0.19	0.76	-	-24.1
	180	Jan-12	3533	-2.99	1.65	0.20	0.80	15,239	
	360	Jul-12	3140	-3.02	1.58	0.20	0.79	1,178	

Sedimentation

Feature Elevation

Table 4.3 provides a summary of the LiDAR-derived bare earth elevation data for the northern Chandeleur Islands from 2001 to 2015. This table provides a general overview of island elevation and elevation changes that occurred on existing and newly created island features adjacent to the EBB. Though some of the LiDAR collections included shallow subaqueous features, all sets were standardized by excluding data with elevations below 0.0 m (NAVD). The 2001 landscape consisted of island features with a maximum elevation of 3.66 m (NAVD) and an overall average height of 0.47 m. These low-lying island features are indicative of a system that was sediment deprived and increasingly susceptible to additional disturbances (FitzGerald *et al*. 2016; Sherwood *et al*. 2014). A large disturbance event, Hurricane Katrina (77.8 m s^{-1}, storm surge 7.3 to 8.5 m; Knabb *et al*. 2005), which made landfall on 29 August 2005, significantly reduced the mean elevation of the Chandeleur Islands to 0.34 and 0.33 m in 2005 and 2007, respectively. A slight recovery in elevation was observed in 2010, prior to the construction of the EBB in 2011, which increased the maximum and average elevations of the island/berm complex to 3.8 and 0.63 m, respectively. Impacts from Tropical Storm Lee and Hurricane Isaac breeched and overwashed the berm, redistributing large volumes of sediment (within and out of the island system), thereby reducing the average island elevation to 0.39 m by 2013. Table 4.3 also shows the island average elevation remained relatively stable between 2013 and 2015, increasing slightly from 0.39 to 0.4 m.

Table 4.3. Light Detection And Ranging (LiDAR)-derived bare earth elevations for northern Chandeleur Island subaerial features.

			Elevation (m NAVD)			
Island	**Date**	**Period**	**Min**	**Max**	**Mean**	**Std**
Chandeleur	Oct-01	Pre-Construction	0.00	3.66	0.47	0.43
	Dec-05	Pre-Construction	0.00	1.35	0.34	0.22
	Jun-07	Pre-Construction	0.00	2.39	0.33	0.22
	Mar-10	Pre-Construction	0.00	2.35	0.42	0.33
	May-11	Construction	0.00	3.80	0.63	0.31
	Feb-12	Post-Construction	0.00	3.04	0.47	0.32
	Jul-13	Post-Construction	0.00	2.90	0.39	0.24
	Feb-15	Post-Construction	0.00	3.14	0.40	0.30

The location, distribution, and change of elevations across the northern Chandeleur Islands from 2007 to 2015 are shown in Figure 4.3. Panels A, C, and E represent the bare earth elevations (in meters) in 2007, 2011, and 2015, respectively. For these panels the elevations range from -0.3 (included to identify shallow subaqueous shoals) to 3.8 m (maxima in 2011). The elevation values are also color ramped, where dark browns represent lower elevations, dark blues represent higher elevation features, and the remaining represent mid-range elevations. The 2007 Chandeleur Island (Figure 4.2A) consisted of a post-Hurricane Katrina impacted landscape that was dominated by breaches and low relief features. Figure 4.2C represents 2011 elevations along the newly constructed berm and existing island features. These new berm features, which are represented by the light to dark blue regions at the eastern edge of the Island's beach, closed many of the island's existing breaches and passes. Figure 4.2E represents island feature elevations in 2015, approximately four years post-construction of the EBB. This panel illustrates the storm related overwashing of the EBB, which resulted in the breaching, thinning, and redistribution of berm sediment onto and bayside of existing island features (large brown regions).

In Figure 4.3, panels B and D represent elevation change between paired dates. The values in these panels are also color ramped, where red represents the largest reductions in elevation, and dark green represents the largest increases in elevation. Panel B illustrates the elevation changes that occurred between June 2007 and May 2011. Significant increases in elevation (dark green) were observed along the constructed berm feature. The light green regions in Panel B represent more moderate increases in elevation along existing island features, which are indicative of sediment redistribution that occurred during berm construction. Between 2007 and 2011, the maximum and mean elevations in the northern Chandeleur Island system increased by 1.41 and 0.3 m, respectively. Figure 4.3D illustrates the change in elevation between May 2011 and February 2015. Red features along the eastern edge of the island represent berm segments that experienced significant reductions in elevation. Most of those reductions were due to the overtopping and redistribution of sediment within the system. Also visible in Panel D are large areas of increased elevation, which appear as green features west of the berm footprint. These are primarily shallow shoals that are detectable with LiDAR data, however there are large areas of deeper irregularly exposed shoals that were created after the construction of the EBB. These features were identified and digitized during the habitat classification process. It is assumed that these areas (represented by the dark gray polygons in Figure 4.3B and 4.3D) are the result of redistributed berm sediments.

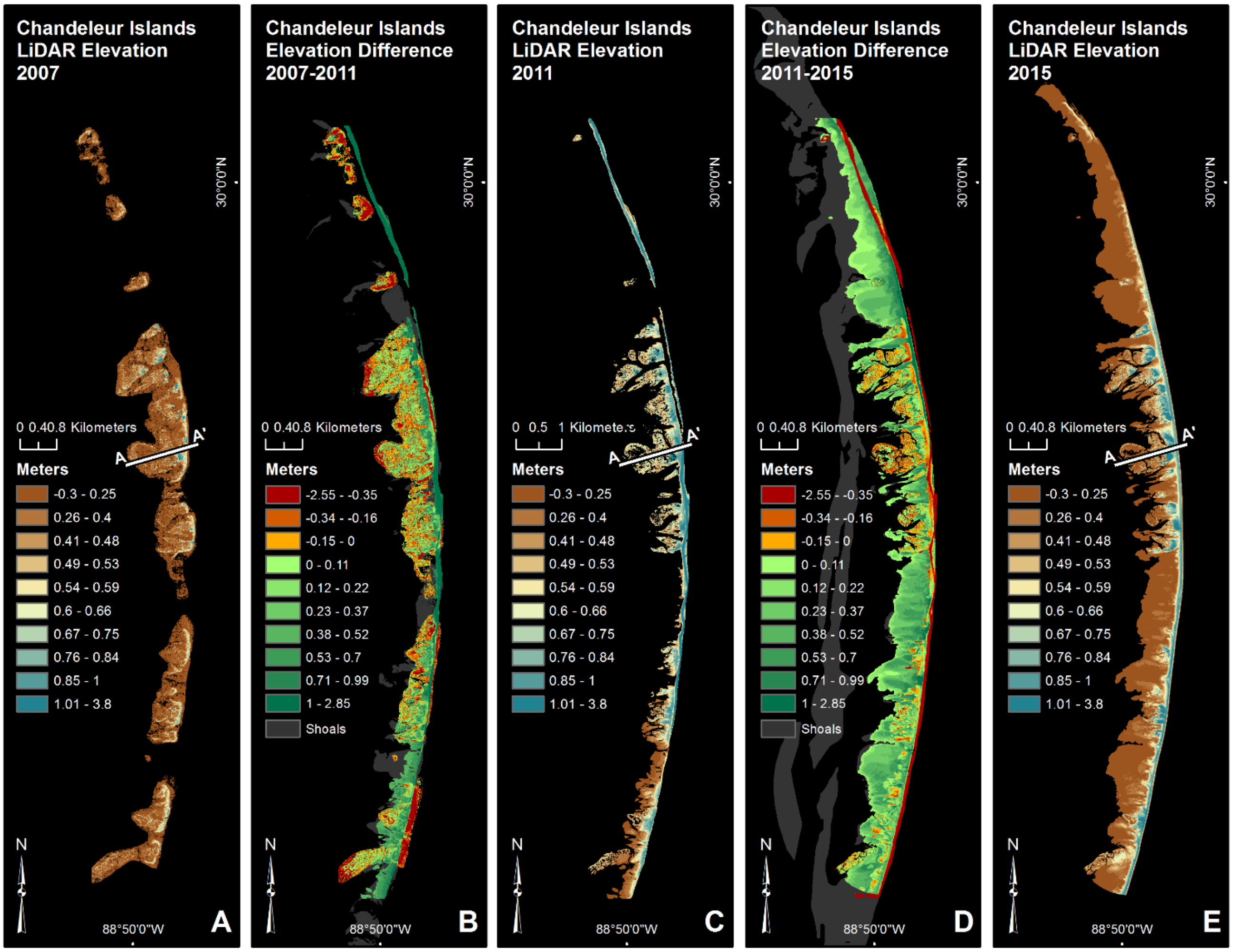

Figure 4.3. Chandeleur Islands sand berm project area Light Detection And Ranging (LiDAR) elevations and elevation change between March 2007 and February 2015. Deeper shoaled areas are represented by dark gray polygons in panels B (2011 shoals) and D (2013 shoals).

The 2007, 2011, and 2015 LiDAR data were also used to create elevation profiles along a 1,650 m transect (start and stop locations represented by A-A' in Figure 4.3) that traversed existing and newly created island features (Figure 4.4). The 2007 profile (black line in Figure 4.4) shows the island consisted of low relief subaerial beach, dune, and tidal marsh features, as well as subaqueous features within and adjacent to the tidal marsh. By 2011 (red line in Figure 4.4) the island experienced an elevation increase due to the construction of the sand berm and redistribution of dredged sediments. The largest increases in elevations in 2011 are observed within the berm footprint (near station 1,400+50) and extending to the landward side of the dune (near station 1,200+00). Many of the existing tidal marsh features also experienced increases in elevation (though some changes are potentially due to LiDAR returns through dense vegetation), except for the most westward area of marsh, which experienced wetland loss between stations 100+20 and 200+75. By 2015 (green line in Figure 4.4) large quantities of berm sediments were redistributed onto dune features and into overwash regions of the island between the dune and the back barrier wetlands (between stations 1,000+00 and 1,200+00). The 2015 tidal marsh elevations primarily ranged between the 2007 and 2011 elevations, however, some marsh elevations (nearest the overwash and dune features) were highest in the 2015 profile.

Habitat Change

Figure 4.5 provides historical and recent perspectives on the type, extent, and change of wetland (NWI derived wetland class represented by black diamonds) and unconsolidated shores (represented by gray squares) along the northern Chandeleur Islands. Air- and space-borne imagery-derived wetland and water data, from 2004 to 2015, were also used to assess the overall accuracy of the habitat derived wetland data, and to provide a more robust and higher temporal estimate of wetland change over time. Figure 4.5 shows high levels of agreement between the NWI-derived data (black diamonds) and the satellite imagery-derived wetland data (white dots). The overall classification accuracy for the paired wetland and water data was 91 percent (kappa = 0.82). Minor difference between these data sets are potentially due to varying resolution (grain size) between data sets, and varying water level at the time of data acquisition. These conditions can be influential where large areas of submerged aquatic vegetation and shallow unconsolidated shores are present.

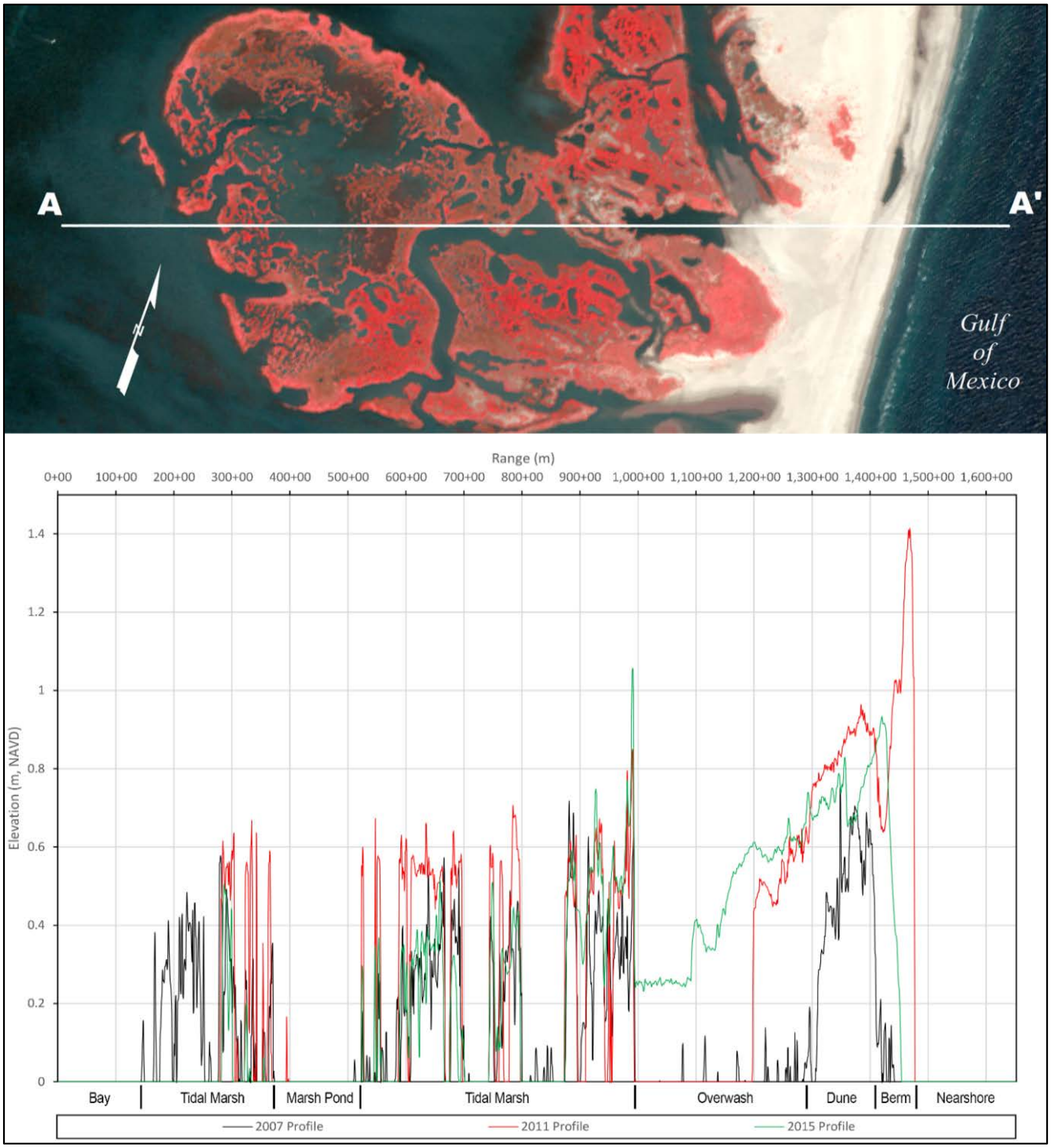

Figure 4.4. Chandeleur Islands elevation profiles from the 2007 (black line), 2011 (red line), and 2015 (green line) LiDAR data. Reference Figure 4.3 for elevation profile transect line.

In 1956, the northern Chandeleurs were a stable and productive island system, consisting of approximately 19.1 km^2 of wetland (dominated by saline marsh and scrub/shrub habitat), and approximately 10.6 km^2 of unconsolidated shores (Figure 4.5). However, during the period from 1956 to 1978, the island experienced significant erosion and habitat loss (-0.75 $km^2\ yr^{-1}$) due largely to the forces of Hurricane Camille (73.8 m s^{-1}, 6.9 m storm surge; USACE 1970), which made landfall on 17 August 1969. The rate of habitat change remained relatively constant in subsequent periods, until reaching a rate of –5.4 km^2 yr^{-1} between 2004 and 2005. The rate of habitat loss within this period was directly related to impacts from Hurricanes Katrina and Rita (79.1 m s^{-1}, approximately 2 m storm surge, 24 September 2005; Knabb *et al*. 2006). From 2005 to 2010 (pre-berm) the island's land features continued to erode, albeit, at a reduced rate of loss (-0.27 km^2 yr^{-1}), while the unconsolidated shores experienced significant post-storm gains, increasing from 0.35 km^2 in 2005 to 6.1 km^2 by 2008. The gains in unconsolidated shores during this period are probably due to storm-induced relocation of beach and dune sediments. Figure 4.5 also shows the period of berm construction (gray bar), and subsequent change in wetland and unconsolidated shore areas. By 2011, the total habitat area (land and unconsolidated shores) increased to approximately 13.8 km^2, of which, the newly constructed berm accounted for approximately 2.6 km^2 (LOCPR 2011). During the four years following the construction of the EBB, increases in wetland and unconsolidated shore areas were observed. Though these increases are moderate, +0.52 km^2 yr^{-1} and +0.38 km^2 yr^{-1} for wetland and unconsolidated shores, respectively, they reverse the long and substantial loss trends that have occurred since the 1950s.

Trends observed in Figure 4.5 largely corroborate those that were observed in Figure 4.3, with one discrepancy. Figure 4.5 shows a large increase in unconsolidated shores in 2011 that are not observed in Figure 4.3. This is potentially due to deeper shoals that were detected and classified in the habitat assessments but were not detectable using LiDAR. This assumption is further supported by the inclusion of deeper shoal areas in Figure 4.3, which are represented by the dark gray polygons in Panels B (2008 shoals) and D (2013 shoals). Based on the observed horizontal and vertical accumulation of sediment within the Chandeleur Island system, the EBB has provided geomorphological benefits to existing and newly created features.

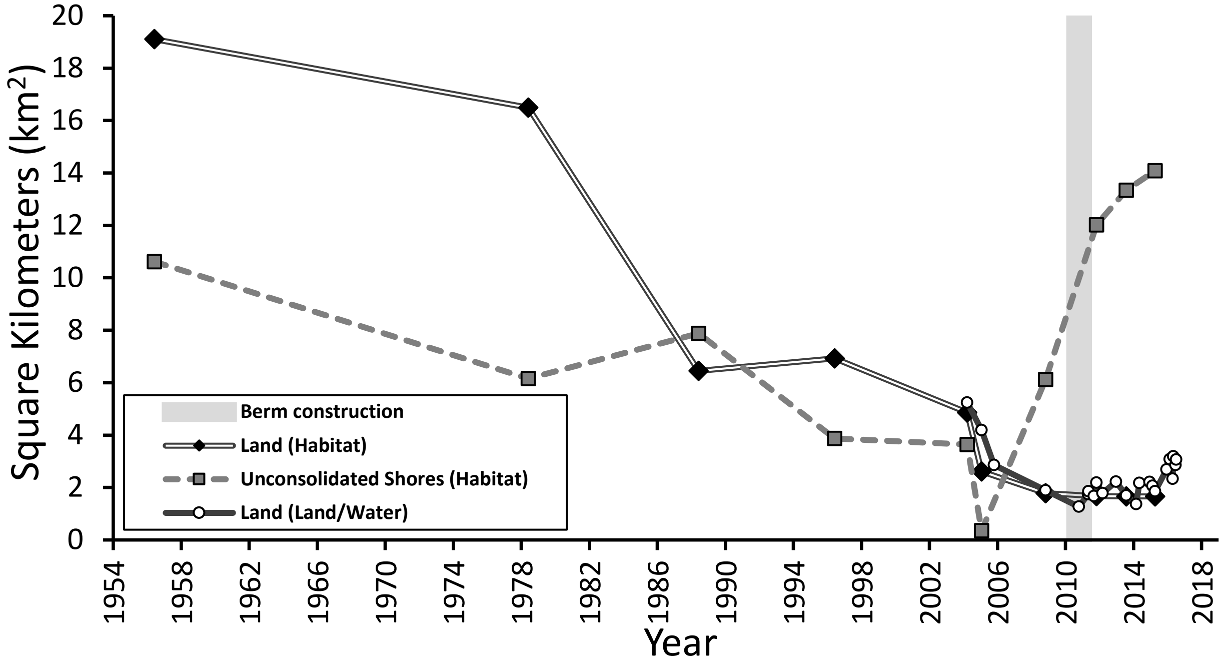

Figure 4.5. Summary of change in historical and recent habitat (modified) area along the northern Chandeleur Islands. Unconsolidated shores include beach, shore and flats (from historical data sets), and the irregularly flooded, regularly flooded, and irregularly exposed classes (derived from recent data sets).

Vegetation Impacts

Few studies have assessed the impacts of nearshore BUDM on wetland vegetation quantity, quality, and productivity. Therefore, standard vegetation assessments (i.e., distribution and composition) were used in conjunction with NDVI and FQI metrics to evaluate the condition, performance, and evolution of wetland plants during three berm-related periods of analysis (pre-construction, construction, and post-construction periods).

Normalized Difference Vegetative Index

Figure 4.6 illustrates the mean values and trajectory of NDVI data within the island assessment unit. These represent all qualifying data, regardless of season, across three distinct periods, the pre-construction (white dots), construction (gray dots), and post-construction (black dots) periods. The mean NDVI values ranged from 0.31 in 2005 (pre-construction of the EBB) to 0.75 in 2015 (post-construction). The mean NDVI values by period were 0.41 ± 0.13, 0.54 ± 0.17, and 0.56 ±

0.18, for the pre-construction, construction, and post-construction periods, respectively. The mean NDVI values for all three construction phases were significantly different ($p<0.05$).

The lower mean NDVI values during the pre-construction period, specifically those in 2005, were largely due to disturbance events (i.e., hurricanes). Regardless of impacts, these pre-construction period values (0.41) were higher than those previously computed for saline marsh (0.36) in coastal Louisiana (Suir *et al*. unpublished). By 2007, the Chandeleur Islands experienced some post-hurricane recovery of vegetation productivity (Figure 4.6). These data corroborate previous studies which report post-hurricane vegetative recovery typically occurs by the end of the next full growing season (Carle *et al*. 2015; Steyer *et al*. 2013). The dashed line in Figure 4.6 shows the general increasing trend in NDVI values across the construction and post-construction periods. Existing northern Chandeleur Island marsh experienced increased productivity that is probably due to elevational benefits (increased productivity with shallow marsh burial and reduced flooding stress) and increased nutrient levels (ammonium and phosphate sorbed to clay fractions) that occurred during the placement of EBB sediments (Walters and Kirwan 2016). The post-construction period began with tropical storm and hurricane impacts (Tropical Storm Lee and Hurricane Isaac) to existing island features, which resulted in the slight reductions in NDVI values in 2011 and 2012. However, from 2014 to 2016 the barrier island assessment area experienced vegetative recovery and increases in NDVI values. The average NDVI change rate for the post-construction period was +0.13 per year. The Chandeleur Islands experienced an overall increasing trend in NDVI (productivity), resulting in a change rate of approximately +0.02 per year, and a moderate r^2 (coefficient of determination) of 0.45 across the entire period of analysis (2004 to 2016).

The NDVI data were also used to evaluate the spatial variability and patterns of vegetative productivity, and the rate of change on a pixel-by-pixel basis. Since the influence of seasonality (temperature and sunlight) on plant productivity is well established (Ramsey and Rangoonwala 2016;

Sasser *et al*. 1995; Steyer *et al*. 2008), only NDVI data from within the growing season (May to September), and those acquired during and after construction of the EBB, were used to evaluate berm sediment impacts on aboveground biomass.

Figure 4.6. Average Normalized Difference Vegetation Index values by year for the pre-construction, construction, and post-construction periods.

Figure 4.7 shows the slope or change rate of NDVI across all qualifying data within the construction and post-construction periods of record. The NDVI rates of change are color ramped in Figure 4.7, where the yellow-to-orange-to-red colors represent decreasing rates of NDVI over time, and the green-to-blue colors represent increasing rates of NDVI over time. The majority of the northern Chandeleur Islands experienced low to moderate increasing rates of NDVI values over the period of analysis (2010 to 2016). Generally, existing island features experienced low increasing rates of NDVI (green pixels), while some of the newly created wetland features experienced the highest increasing rates (blue pixels). Few areas, namely the berm, island dune, and ephemeral shoals, experienced loss rates (oranges and red pixels). These loss features were either located in high energy positions or at elevations that are not ideal for emergent wetland persistence. Comparing Figure 4.7 to Figure 4.3 (Panels C and E) it is apparent that not only have the areal extent of shoals increased, but by 2016, many of those shoals had vegetated and experienced increasing plant productivity. Therefore, it is a reasonable conclusion that the redistributed EBB sediments have provided elevation, structural, and nutrient benefits to existing Chandeleur Island wetland features. The EBB sediment have also created new platforms and enhanced existing platforms, resulting in significant wetland plant colonization and productivity.

Floristic Quality Index

The FQI combines vegetative quantity and quality to provide measurements of vegetation condition and maturity. Low FQI values can be indicative of early successional vegetation communities, post-disturbance evolution, or other presses or pulses that are negatively impacting natural or restored wetlands. Conversely, high FQI values are more typical in mature, stable, and un-disturbed wetlands. As mentioned previously, the FQI is composed of two components, the average plant abundance (i.e., cover value percentage) by species, and the average Coefficient of Conservatism. Figure 4.8 provides both FQI components, separately, for each sampling session within the three periods of analysis, pre-construction (2001 to 2005), construction (2010 to 2011), and post-construction (2011 to 2013). The vegetation surveys conducted in 2001 show the sampled island features contained nominal amounts (1.2% to 1.6% cover) of *Spartina alterniflora* (saltmeadow cordgrass). The degraded conditions of the Chandeleur Islands at that time were a result of Hurricane Georges (30 m s^{-1}, 2.7 m storm surge, 28 September 1998; Guiney 1999), which significantly breached the island and reduced emergent features by approximately 40 percent (Hymel 2007). In 2001, as part of the CWPPRA PO-27 project, a total of 80,730 *Spartina alterniflora* plants were installed to provide stability to approximately 1.5 km^2 of unvegetated overwash fans (LCWCRTF 2004). The *Spartina alterniflora* communities expanded, increasing to average cover values of 28%, 36.3%, and 48.7% in 2002, 2003, and September 2004, respectively. In addition to increasing *Spartina alterniflora* cover, the islands also experienced a significant increase in overall cover, driven primarily by the establishment of disturbance (*Amaranthus australis* [southern amaranth]), vigorous (*Batis maritima* [turtleweed], *Iva frutescens, Salicornia bigelovii, Sesuvium portulacastrum, and Suaeda linearis*), common (*Salicornia depressa*), and dominant (*Distichlis spicata* [Coastal Salt Grass]) wetland species in 2003 and 2004. Disturbance and vigorous plants are opportunistic species and often serve as indicators of disturbed or stressed systems. The level of disturbance within the system is illustrated by the reduction in CC value between October 2002 and September 2004 (Figure 4.8). Due to *Spartina alterniflora* monocultures, the sample sites had early sampling period CC scores of 10. However, an insurgence of ruderal plants (i.e., invasive, disturbance, and vigorous species) after Hurricane Georges resulted in lower CC scores (8.6 and 8.3).

Similar trends to Hurricanes Georges were observed with Hurricane Ivan (49 m s^{-1}, 1 m storm surge; Stewart 2004), which made landfall on 16 September 2004. Hurricane Ivan significantly impacted the Chandeleur Islands, resulting in dramatic declines in vegetative cover by December 2004, and subsequent establishment of dominant and opportunistic species by March 2005. The average cover of *Spartina alterniflora* (the dominant species) were reduced during this Post-Ivan period, decreasing from 48.7% to 32.7% by December 2004, and 9.3% by March 2005. While *Spartina alterniflora* communities diminished during this period, other common and dominant species (*Avicennia germinans, Spartina patens, Juncus sp, and Distichlis spicata*) proliferated, reducing the CC score to 6.9 by March 2005. Though no *in situ* vegetation collections occurred between March 2005 and November 2010, the Island experienced significant storm impacts from Hurricanes Katrina and Rita. It is suspected the Island underwent similar, if not more severe, vegetative trends to those encountered after Hurricanes Georges and Ivan.

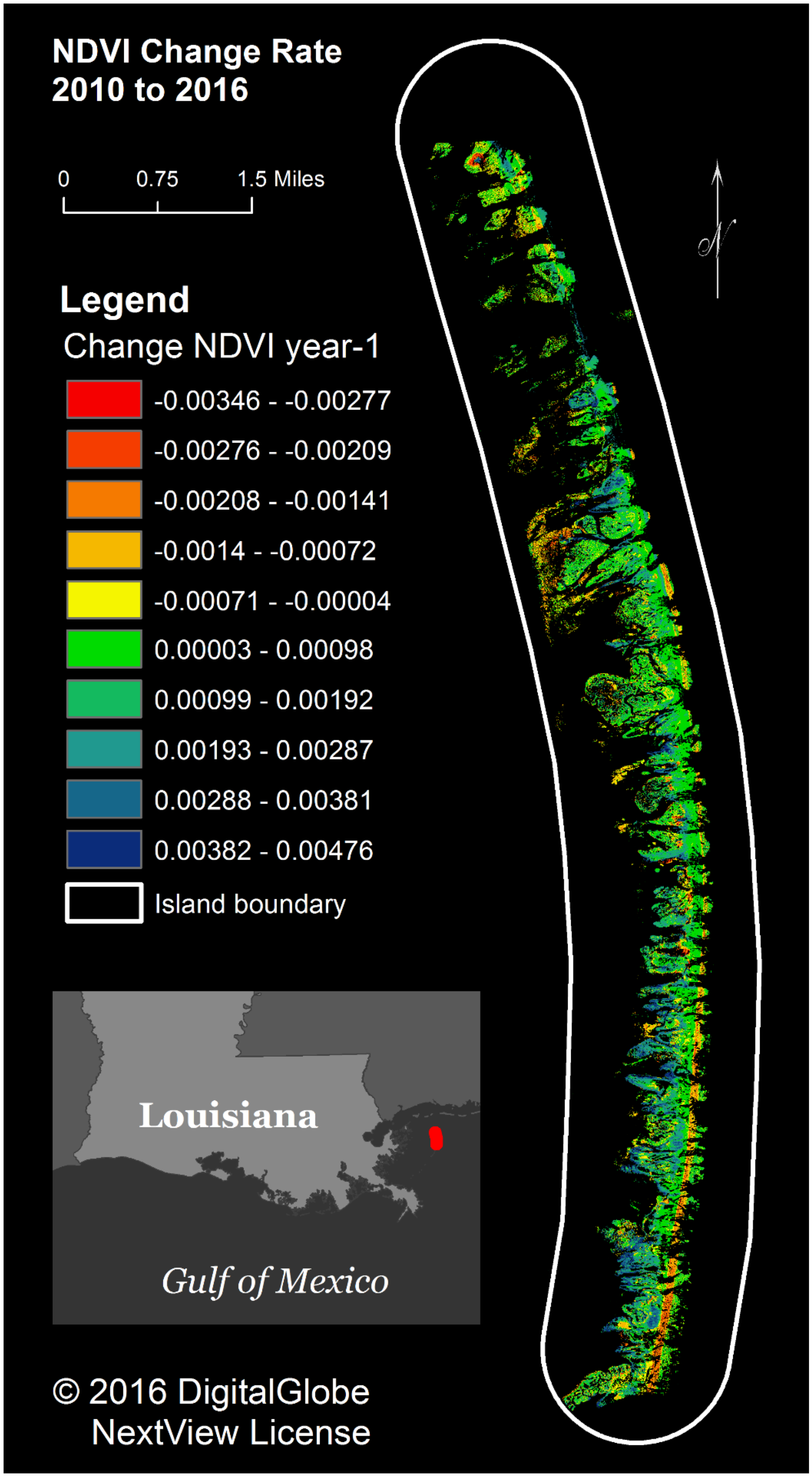

Figure 4.7. Post-construction (2010 to 2016) Normalized Difference Vegetative Index (NDVI) change rate (per-pixel) within the northern Chandeleur Islands.

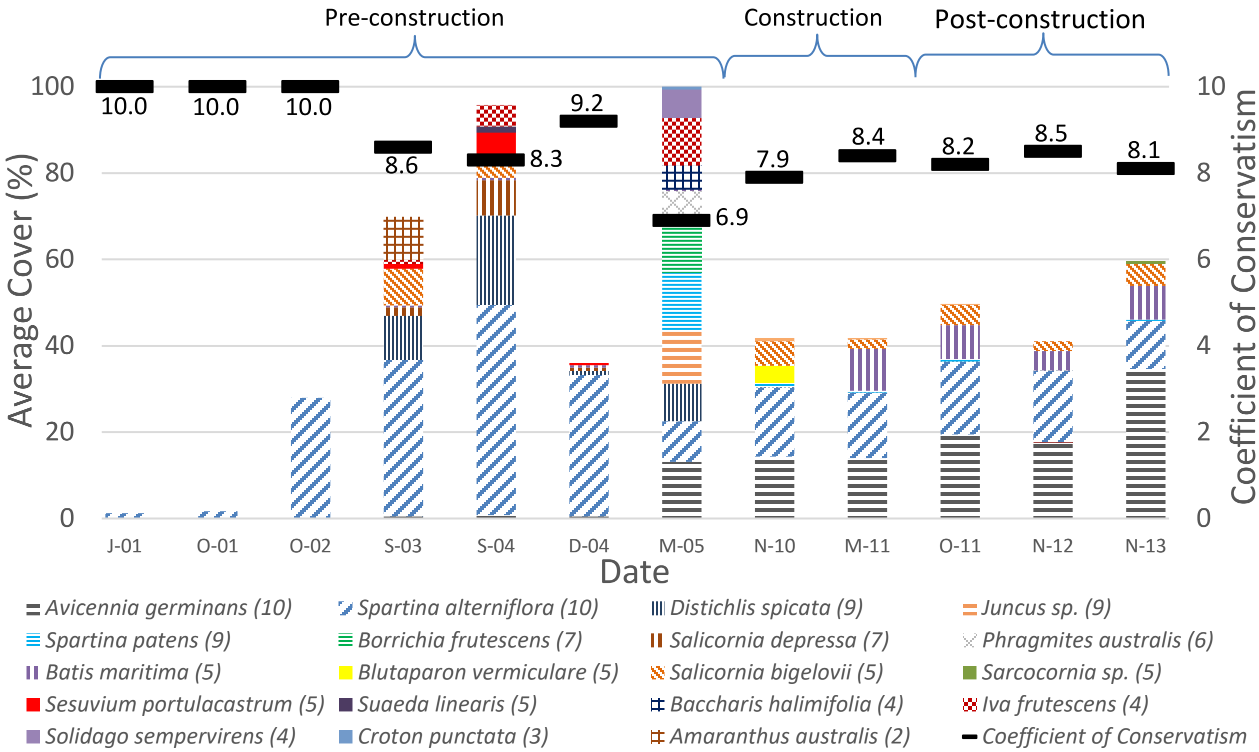

Figure 4.8. Percent cover and Coefficient of Conservatism (CC) values for species within the Northern Chandeleur Island assessment unit. CC values for each plant species are provided in parenthesis.

Five vegetation sampling sessions occurred during the construction (November 2010 and May 2011) and post-construction (October 2011, November 2012, and November 2013) periods of analysis (Figure 4.8). During both periods the sample sites consisted of stable cover values that were dominated by *Spartina alterniflora* (minimum of 11.1% in 2013 to a maximum of 16.9% in October 2011) and *Avicennia germinans* (minimum 14.0% in May 2011 to a maximum of 34.7% in 2013). The construction and post-construction periods also saw moderate cover by *Batis maritima* and *Salicornia bigelovii*. Though the construction and post-construction periods experienced moderate to severe storm events, the typical shift to disturbance vegetation type and cover were not observed. This could be a result of increased stability incurred by the redistribution of EBB sediments and the recent increase in foundation species. Foundation species, like *Spartina alterniflora* and *Avicennia germinans* provide many benefits, but few more important to the Chandeleur Islands than soil stabilization and storm protection (Diskin and Smee 2017). The vegetative stability across the Chandeleur Island sample sites during the construction and post-construction periods are evidenced in the CC scores in Figure 4.8. The CC scores across these five sample periods ranged from 7.9 to 8.5. Minimal fluctuations in CC scores are indicative of a stable system that either did not experience severe disturbance events, or was able to adequately resist disturbances.

Figure 4.9 provides the average FQI scores for each sampling site across the 2001 to 2013 period of record. The average FQI scores during the pre-construction, construction, and post-construction periods are represented by the white, gray, and black dots, respectively. Figure 4.9 also includes two trend lines, one for the pre-construction period, and one that covers the construction and post-construction periods. In the pre-construction period the average FQI by sampling session ranged from 1.24 in 2001 to 51.75 in 2004. The FQI trend during this period was one of significant increase (+0.036 per day), which was initiated primarily by post-disturbance increases in vegetation colonization and recovery. During this pre-construction period there was a moderate positive correlation (r^2 = 0.398) between FQI and time. Since there is significant variability in sample site FQI scores from 2002 to 2005, this correlation was probably driven by the lower and less variable FQI values in the early part of the period.

The average FQI scores for sampling sessions within the construction and post-construction periods ranged from 36.8 in 2010 to 51.6 in 2013. Compared to the pre-construction trend, the trajectory between 2010 and 2013 was a moderate increase in average FQI (+0.009 per year). This is in spite of climate events that significantly impacted the island during the post-construction period. The combination of moderately increasing FQI scores and cover values is indicative of a system that was stabilizing (relative to historical trends of degradation) during the construction and post-construction periods. The mean FQI scores across all periods were lower than the ideal range (>80, preliminarily established by Cretini *et al*. 2011) for saline marsh in the inactive deltaic plain.

However, Suir and Sasser (2016) surmised that lower ideal ranges of FQI (between 50 and 70) might be more reasonable, especially in degrading wetlands, recently restored wetlands, or landscapes with recent disturbance events.

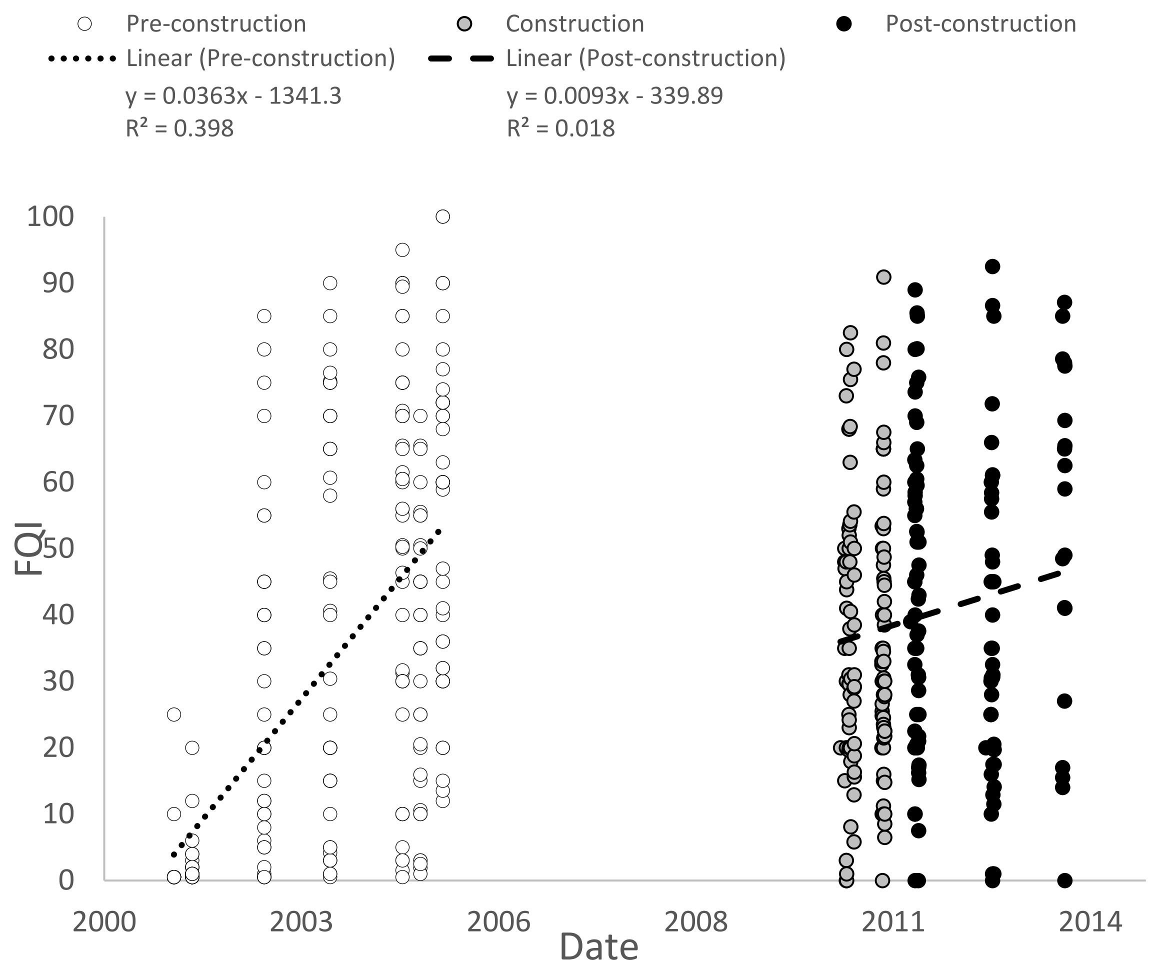

Figure 4.9. Modified Floristic Quality Index (FQI_{mod}) scores for all survey stations within the Northern Chandeleur Island assessment unit by year and construction period.

CONCLUSIONS

The purpose of this study was to evaluate EBB sediment impacts on northern Chandeleur Island features. To satisfy these objectives, GIS and remote sensing data and techniques were used to (1) evaluate the redistribution of EBB sediment within the island system; (2) assess EBB impacts by evaluating the quantity and quality of existing and new emergent vegetation as a function of berm sediment; and (3) consider the implications of this research on future island restoration and nourishment.

Though the EBB didn’t fulfill its primary mission of retarding oil (National Commission on the BP Deepwater Horizon Oil Spill and Offshore Drilling 2011; Suir and Sasser 2016), it did serve as a BUDM application that introduced sediment into a sediment- and elevation-deficient system. However, the Chandeleur beach/dune system is much lower in elevation than most mainland and barrier island beach/dune systems where BUDM would be typically practiced (most BUDM nearshore placement applications occur near robust beach/dune systems). Yet, the dissipation of the EBB is consistent with known nearshore berm processes, where sediment quickly migrates from the placement site to the resource (shoreline) it was intended to protect/nourish and beyond. EBB sediments were redistributed within and out of the Island system via marine and climate processes (i.e., storm induced scouring, breaching, overwashing).

Ultimately, the EBB sediments provided elevational lifts to existing back barrier island wetlands and shoals, and in some areas created new shoals in overwash and drift zones. The evolution and redistribution of EBB sediments were observed through elevation and habitat change assessments. Historical assessments (1956 to 2005) showed the island experienced severe storm-induced degradation and sediment deficiencies, which resulted in the thinning and lowering of all island features. Assessments using recent data (2010 to 2016) showed significant increases in the horizontal and vertical extent of existing and newly created features. These changes are largely due to the influx of high quality quartz sediments that were redistributed from the EBB. These sediments provided new shoal areas for vegetation establishment and elevational (shallow marsh burial and reduced flooding stress) and nutrient (ammonium and phosphate sorbed to clay fractions) benefits that resulted in increased vegetation productivity. The redistributed sediments also encouraged island feature stability, both in the maintenance of areal extent and the quantity and quality of vegetation present (i.e., foundational species).

Future work should investigate and monitor the long-term impacts of berm sediment and oil on existing critical (submerged aquatic vegetation) and sensitive Chandeleur Island habitat. These data would be necessary to evaluate the impacts of future disturbance events on post-construction island structure and functions. Results from this study could be used by resource managers and decision makers to identify and plan future nearshore BUDM applications, particularly in atypical environments and systems.

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CHAPTER 5 – USE OF REMOTE SENSING AND FIELD DATA TO QUANTIFY THE EFFECTS AND RESILIENCE OF DREDGED SEDIMENT USED FOR ECOSYSTEM RESTORATION

INTRODUCTION

In the last half century, research has shown that wetlands are among the most productive and beneficial ecosystems in the World (Moreno-Mateos *et al*. 2012). Wetlands provide benefits that range from regulating services (i.e., floods and drought), supporting services (i.e., soil formation and nutrient cycling), provisioning services (i.e., food and freshwater); cultural services (i.e., recreational and aesthetics), to ecosystem services (i.e., high biological productivity and critical habitat) (Millennium Ecosystem Assessment 2003; USACE 2013). With an increasing understanding of wetland importance, Federal and State governments have enacted a number of policies, regulations, and incentive programs to directly and indirectly protect, maintain, and restore the wetlands of the United States (Votteler and Muir 1996).

In the United States, restoration efforts rapidly developed into large authorities and programs such as the Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA), the United States Army Corps of Engineers (USACE) Beneficial Use of Dredged Material (BUDM) program, and many State led wetland restoration programs. In Louisiana, CWPPRA and BUDM programs have collectively created or benefited nearly 40,000 hectares of wetlands (Louisiana Coastal Wetlands Conservation and Restoration Task Force [LCWCRTF] 2015). Additionally, the Coastal Protection and Restoration Authority (CPRA) of Louisiana conservatively estimates (depending on future coastal conditions) approximately 150,000 hectares of wetlands will be created or nourished as part of its Louisiana Coastal Master Plan (Coastal Protection and Restoration Authority of Louisiana 2012).

Typical goals of wetland restoration efforts are to conserve, create, or enhance wetland form; and to achieve wetland function that approaches natural conditions. Though wetland form and function are driven by many presses and pulses, the dominant factors include elevation, hydrology, sedimentation, and vegetation (Steyer *et al*. 2008b). For a constructed wetland, failure to adequately account for or maintain one of these elements can have negative implications on other elements, and ultimately on overall wetland condition. Wetland condition has traditionally been evaluated using a system's structure and/or ability to perform a suite of functions (Cohen *et al*. 2004). Measures of wetland condition have been used to monitor and assess wetland performance, resilience, and adaptive management needs. However, monitoring wetland condition can be time-intensive, costly, and often requires repeat surveys with high precision data about the landscape, land cover, species/habitat composition, change detection, degradation, diversity, as well as system threats and pressures.

There are three basic levels of wetland monitoring and assessment: (1) *landscape assessment* – which consists of coarse inventory information that is acquired and assessed using geographic information system (GIS) and remote sensing techniques; (2) *rapid assessments* – which are site specific analyses using regionally derived and relatively simple and rapid protocols (e.g., Louisiana Wetland Rapid Assessment Method [LRAM]); and (3) *intensive site assessments* – consisting of research derived, multi-metric indices that give detailed information about wetland function (e.g., Hydrogeomorphic [HGM] Approach) (U.S. Environmental Protection Agency [USEPA] 2002b). Regardless of level, each assessment type provides metrics and indices that translate into descriptions of biological condition (Karr and Chu 1997). Landscape assessments provide useful information when evaluating wetland change trajectories or analyzing direct episodic impacts across larger spatial and temporal scales. However, they may be inadequate for analyzing complex and dynamic systems. Conversely, intensive site assessments provide detailed information that are necessary for analyzing complex systems, but these assessments are customarily labor and resource intensive; and unless high levels of detail are required, they can be unnecessary and impractical. Rapid assessments are useful when general site-specific wetland ecological conditions are required. Combining aspects from each type of wetland evaluation provides biological indices that measure or estimate wetland quantity and quality.

When measuring wetland function and condition, plants are excellent indicators due to their rapid growth rates and direct response to environmental stressors and disturbances (Cohen *et al*. 2004; Mack 2007; Smith *et al*. 1995; USEPA 2002a). Specifically, plant species composition, cover, density, and biomass are structural components of coastal marshes that are commonly used to quantify vegetative characteristics and often serve as indicators of wetland condition (Chamberlain and Ingram 2012; Cretini *et al*. 2012). Although these structural components are useful for quantifying and comparing wetland characteristics, they lack qualitative measures that are necessary for more comprehensive assessments of wetland function. Wetland plant quality can be an essential metric because it provides critical information related to habitats, effectiveness of restoration measures, resilience to disturbance events, and adaptive management needs and priorities (USEPA 2002a).

The overall goals of this study were to assess methods for determining practical indicators of wetland condition, including vegetative abundance (productivity), species composition (biodiversity), and wetland robustness (long-term resilience or resistance to structural or functional change). Specifically, the objectives of this study were to utilize remotely sensed and field collected data to: (1) evaluate and compare structural changes of restored wetlands to naturally occurring reference wetlands; (2) quantify the quality and functional changes of restored wetlands and compare to reference wetlands; and (3) assess the resilience and recovery of restored wetlands to short-term episodic events (i.e., tropical storms and salinity spikes).

METHODS

This study utilized established methods, along with field- and remotely-collected data, to assess restored wetland structure, function, and resilience, and compare those to reference wetlands.

Study Area

The Sabine Refuge Marsh Creation CWPPRA (CS-28) restoration project, and surrounding areas, were selected as the study and reference sites, respectively (Figure 5.1). These study and reference areas consist primarily of brackish wetlands that are located west of the Calcasieu Ship Channel near Hackberry, Louisiana (Figure 5.1). This area, which was severely impacted by hurricanes and canal building, experienced significant conversion from wetlands to open water between 1956 and 1978 (Miller 2014a). Efforts to restore the area wetlands comprised five separate dredging cycles and creation sites (ranging in size from 51 to 93 hectares [ha]) within an open water area of approximately 1,150 ha. The creation sites, known as Cycles 1-5, were constructed in 2002 (Cycle 1), 2007 (Cycle 3), 2010 (Cycle 2), 2014 (Cycle 4), and 2015 (Cycle 5), respectively. Using approximately 3.4 million m^3 of dredged material (sediment from the Sabine National Wildlife Refuge disposal facility, which consisted of approximately 40% sand and 60% fines), the cycles were constructed to an initial height of +0.82 to +0.94 m North American Vertical Datum 1988 (NAVD88, Geoid 99). The placement of dredged material was similar for all Cycles, expect for Cycle 2, where material was allowed to overflow the western dike, resulting in a large "overflow area" (Figure 5.1). Placed dredged material was allowed to consolidate and desiccate to a final target elevation of approximately +0.37 m NAVD88 (Sharp 2003; USACE 2005; Miller 2014a). The Cycles differ in that Cycle 1 was planted with approximately 36,000 *Spartina alterniflora* (saltmeadow cordgrass) plants around the perimeter and along its constructed and meandering hydrologic and fish access channels (trennases) (Miller 2014a; Sharp 2003). Conversely, vegetation and trenasses were allowed to occur naturally in Cycles 2, 3, 4, and 5. Additionally, emergent vegetation was allowed to colonize prior to the breaching of containment dikes (allowing for hydrologic and fisheries access) in all Cycles except for Cycle 2, which remained free of vegetation until the dikes were breached, allowing for the requisite hydrology. At the time of initial sampling, only Cycles 1 and 3 contained vegetation and previously collected vegetation survey data, so Cycles 2, 4, and 5 were only included in the remote sensing-based assessments (descriptions below).

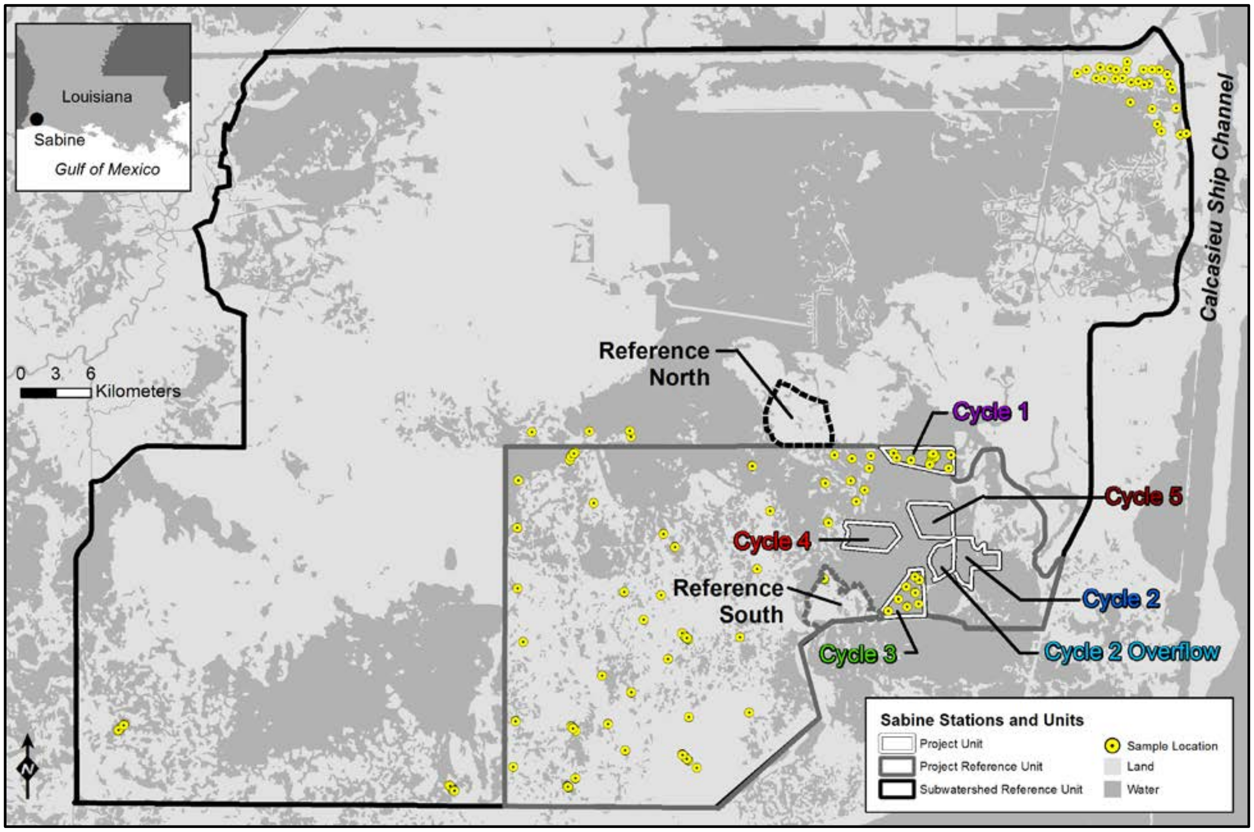

Figure 5.1. Location map of the Sabine study area assessment units (Project Cycles, Project Reference, Subwatershed Reference, Reference South, and Reference North) and data collection sites.

Assessment Units

Reference wetland sites serve as standards against which others are evaluated, and therefore can be critical components of biological assessments (USEPA 2002a). Although selection of appropriate or representative reference sites can be difficult, the use of multiple sites and scales can overcome some of the challenges of defining a reference standard for evaluating restoration performance (Matthews *et al*. 2009). The assessment units used in this study consist of varying scales: the Project, Project Reference (PR), and Subwatershed Reference (SR) units (Figure 5.1). The Project units consist of the pre-defined CWPPRA project boundaries (i.e., Sabine cycles). The PR unit, which is located west of the Project cycles, consists of nearby wetlands within the CWPPRA established reference area. The PR area, which has long been subjected to hydrologic alterations, has received hydrologic restoration measures as part of a separate CWPRRA project (CS-23) that was constructed in 2001. The SR area consists of a generalized hydrologic unit (HUC10) that was intersected with corresponding vegetation zones (intermediate and brackish; Sasser *et al*. 2014) to include wetlands that represent more natural system processes, conditions, and trajectories within a larger watershed segment (Figure 5.1).

Field Collections

Ground reference information from within each assessment unit was collected and used as primary measures of wetland function and structure. Vegetation, photographic, and global positioning data were observed in 0.25 m^2 (0.5 m x 0.5 m) plots within the study cycles (Figure 5.1). When accessible, the same plots were surveyed on repeat visits in the fall of 2014, summer 2015, and fall 2015. A Trimble GeoXH Differential Global Positioning System (DGPS) was used to determine coordinates of existing or proposed survey locations. To assist with site identification and to minimize disturbance, all plots not previously marked were done so at first arrival.

Vegetation Surveys

This study utilized existing (i.e., CWPPRA and Coastwide Reference Monitoring System [CRMS]) and newly collected vegetation data. The CWPPRA monitoring program established standardized methods for monitoring variables that are useful in determining the performance of wetland restoration projects. The vegetation monitoring component of CWPPRA collects species composition, relative abundance, and aboveground biomass data (Steyer and Stewart 1992). However, vegetation data are typically collected more frequently in a project's infancy (yearly), and less frequently (every 3-5 years) as a project nears the end of its anticipated lifespan (CWPPRA projects are typically designed and constructed for 20-year lifespans). Similarly, CRMS is a network of 392 monitoring sites in coastal Louisiana that is used to collect, process, and analyze physical, chemical, biological, and geospatial data to characterize coastal wetland landscapes inside and outside of CWPPRA projects (Cretini *et al*. 2011). Within the CRMS program, emergent vegetation are surveyed annually during the period of peak biomass (Folse *et al*. 2014). All existing vegetation data from CWPPRA and CRMS stations were acquired for the Project, PR, and SR sites (Figure 5.1). For new data collections, vegetation species composition and percent cover were collected from within 0.25 m^2 quadrats at each Project sample site during periods of peak biomass in 2014 and 2015.

Vegetation-specific sampling consisted of species identification and percent cover. Within each plot the vegetation cover of each species was visually estimated using the Braun-Blanquet (1932) scale. Since the percent cover is estimated for each strata, the total vegetation cover (sum of all layers) can exceed 100 percent. The vegetation cover values were used to calculate the Floristic Quality Index (FQI) in each Project, PR, and SR area to evaluate the ecological function of the site.

Remote Sensing

Remote sensing provides a means for classifying landscape features to assess the distribution and change of those features over time. The spatial analyses performed in this study evaluated vegetation quality and quantity as a measure of wetland structure, function, and resilience. Landscape and vegetation assessments utilized existing National Wetlands Inventory (NWI) data (USGS 1980a, 1980b, 2004), newly derived wetland-water data, satellite imagery derived Normalized Difference Vegetation Index (NDVI) data, and vegetation survey data. In addition to these data sets, other existing geospatial data were used as ancillary interpretive information. Remote sensing data and techniques were used to quantify and track landscape structure, test for differences in productivity between restored and reference wetlands, and evaluate the resilience of restored landscapes to episodic disturbance events.

Imagery Acquisition and Processing

Remote sensing assessments were performed using moderate (Landsat) and high (DigitalGlobe) spatial resolution imagery. Landsat TM images, which provide moderate spatial (resampled to 28 m) and temporal (16 day return) resolution data, were acquired using the Google Earth Engine (GEE) image service. GEE utilizes radiometrically and atmospherically corrected imagery, and aggregation functions (i.e., use of outlier values to remove cloud cover from neighboring scenes) to create image composites (Strahler *et al*. 1999; Chander *et al*. 2009). The DigitalGlobe images provide high spatial (1.24 to 2.4 m multispectral) and temporal (1-2 day sensor returns) resolution data that are useful for estimating short-term landscape variation linked to disturbance events and/or prevailing environmental conditions (Suir *et al*. 2011). All DigitalGlobe satellite data were acquired using the Enhanced Viewer Web Hosting Service. The images were geometrically and atmospherically corrected and transformed to reflectance using the Fast Line-of-Sight Atmospheric Analysis of Spectral Hypercubes (FLAASH) algorithm in ENVI 5.3 software (Mutanga *et al*. 2012). Optical inspections were performed on all scenes to identify and remove satellite images of poor quality.

Wetland and Water Classification

The quantification of landscape structure is an important precursor to understanding functional effects of wetland change (Tischendorf 2001). Therefore, this study utilized space-borne data to classify wetland and water features and evaluate their changes over time. The satellite-based methodology, which is a variant of the standard procedures used for CRMS land-water classifications (Folse *et al*. 2014), was performed on moderate- and high-spatial resolution imagery. This classification process utilized the Normalized Difference Water Index (NDWI) and the NDVI (method details provided in the vegetation section below) to identify water and wetland features, respectively. The traditional NDWI (McFeeters 1996), which normalizes a green band against a near infrared (NIR) band, is described by the following equation:

$$\text{NDWI} = \frac{Green - \text{NIR}}{Green + \text{NIR}}. \tag{1}$$

Recoding of the NDWI and NDVI thematic files (with and without edge enhancement) were performed through an overlay process. Clump and eliminate functions were then performed on each recoded file to reduce noise (Braud and Feng 1998; Suir *et al*. 2011). A final overlay was performed in which the NDWI and NDVI images were aggregated and recoded to single files with wetland and water classes.

Landscape Metrics

Landscape ecology is based on the premise that there are strong correlations between landscape pattern (configuration) and ecosystem function (Gustafson 1998). Recent studies have shown linkages between wetland loss, disturbance events, and wetland landscape configuration (Liu and Cameron 2001; Suir *et al*. 2013; Couvillion *et al*. 2016). A number of landscape metrics (i.e., percentage of landscape, edge density, and aggregation index) were selected after careful consideration of previous landscape fragmentation and configuration studies. The Aggregation Index (AI) has evolved as a primary metric for linking structure to ecosystem function and is defined as the frequency with which different pairs of patch types appear side-by-side (McGarigal 2015) and includes like adjacencies between the same patch-type. This index was used to assess landscape configuration changes over time using high resolution air- and space-borne imagery. The class-level AI is derived as:

$$AI = \left(\frac{g_{i,i}}{\max\_g_{i,i}}\right)(100), \tag{2}$$

where $g_{i,i}$ is the number of like adjacencies between pixels of patch type i (class), $\max\_g_{i,i}$ is the maximum number of like adjacencies between pixels of patch type (class) *i* (He *et al*. 2000; McGarigal 2015). The FRAGSTATS landscape pattern analysis software (version 4.2) was used to compute landscape metrics using the high-resolution imagery-derived wetland and water data sets.

Normalized Difference Vegetation Index

NDVI assessments were performed using pre- and post-construction satellite imagery collected from Landsat and DigitalGlobe satellite sensors (i.e., GeoEYE, Quickbird, and WorldView). NDVI data were created using a variant of the standard equation (Rouse *et al*. 1974):

$$\mathrm{NDVI} = \frac{NIR - Red}{NIR + Red}, \tag{3}$$

which utilizes a ratio between a near-infrared (NIR) and red band to measure an ecosystem's ability to capture solar energy and convert it to organic carbon or biomass (An *et al*. 2013). The NDVI has well established correlations to photosynthetic activity, aboveground biomass, and leaf area index (Carle 2013).

NDVI values range from -1 to 1, where values between -1 and zero (0) are typical of non-vegetation features (e.g., water, cloud, and impervious surfaces), and those between 0.2 and 1.0 are typical of green vegetation (Datt 1999; Sims and Gamon 2002). The higher the NDVI value the higher, generally, the biomass, productivity, and vigor of the vegetation. ESRI (2015) ArcGIS 10.5 was used to calculate zonal statistics (i.e., mean, min, max, and standard deviation) on values of each NDVI raster within the Sabine assessment units.

Floristic Quality Index

The traditional Floristic Quality Index (FQI), developed by Swink and Wilhelm (1979), is a weighted metric that was developed to assess the quality of native plant communities. The FQI provides an estimate of habitat quality based on a measure of vulnerability, called the Coefficient of Conservatism (CC), together with the richness of a plant community (Gianopoulos 2014). CC values range from zero (not conservative) to ten (conservative and highly ecologically sensitive), and are assigned to individual plant species within a local flora by a panel of experienced botanists, primarily based on their best professional judgment (Bourdaghs *et al*. 2006; Little 2013). Since the impact and function of plant species differ by region, CC values are specific to a State or region (Little 2013). Table 5.1 provides the criteria that is typically used to assign CC values to individual plant species. Species are also assigned to general classes based on species characteristics. These classes include invasive plant species (CC value of 0), disturbance species (CC = 1–3), vigorous wetland communities (CC = 4–6), common species (CC = 7–8), and dominant wetland species (CC = 9–10).

Table 5.1. General description and criteria for assignment of Coefficient of Conservatism (CC) scores (based on Andreas *et al*. 2004; Cohen *et al*. 2004; Cretini *et al*. 2012).

General characteristics of species	Criteria	CC
Invasive plant species	Obligate to ruderal areas	0
Plants that are opportunistic users of disturbed sites	Occurs more frequently in ruderal areas than natural areas	1
	Facultative to ruderal and natural areas	2
	Occurs less frequent in ruderal areas than natural areas	3
Plants that occur primarily in less vigorous coastal wetland communities	Occurs much more frequently in natural areas than ruderal areas	4
	Obligate to natural areas (quality of area is low)	5
	Weak affinity to high-quality natural areas	6
Plants that are common in vigorous coastal wetland communities	Moderate affinity to high-quality natural areas	7
	High affinity to high-quality natural areas	8
Plants that are dominants in vigorous coastal wetland communities	Very high affinity to high-quality natural areas	9
	Obligate to high-quality natural areas	10

In 2011, a modified FQI (FQI_{mod}) was developed for coastal Louisiana (Cretini *et al*. 2012). FQI_{mod} expanded the traditional FQI by incorporating invasive species, percent cover values, and accounting for total percent cover and overlapping canopies. The addition of invasive species was motivated by research showing strong correlations between floristic index scores and invasive species richness, human activity, and hydrologic impairments (Ervin *et al*. 2006). Additionally, incorporating percent cover and overlapping canopies provides better site characterization and coordination with current monitoring protocols (Folse *et al*. 2014). The modified FQI uses a two-pronged approach to account for sample units with vegetation cover that is less than or equal to 100%, or is greater than 100% (overlapping canopies). If the sum of species covers within a sample unit at time t is less than or equal to 100, the applicable formula is as follows:

$$\mathrm{FQI}_{\mathrm{mod}\ t} = \left(\frac{\sum\ (\mathrm{COVER}_{it} \times \mathrm{CC}_{i})}{100} \right) \times 10, \tag{4}$$

where $\mathrm{FQI}_{\mathrm{mod}\ t}$ is the modified floristic quality index (unitless), COVER_{it} is the percent cover (%) for species *i* at a sample unit, within a sample site, at time *t*, and CC_i is the Coefficient of Conservatism for species *i* (Table 5.1).

By using 100 in the denominator (instead of the actual sum of species covers), differentiation between wetlands of similar composition (e.g., vigorous wetlands) can be made using normalized biomass (estimated through cover) (Cretini *et al*. 2012). For consistency with other CRMS and CWPPRA metrics and indices, the FQI values are multiplied by 10 to scale the scores from 0 to 100 (Cretini *et al*. 2011).

If the sum of species covers within a sample unit at time *t* is greater than 100, the applicable formula is:

$$\mathrm{FQI}_{\mathrm{mod}\ t} = \left(\frac{\sum\ (\mathrm{COVER}_{it} \times \mathrm{CC}_{i})}{\sum\ (\mathrm{TOTAL\ COVER}_{t})} \right) \times 10, \tag{5}$$

where $\mathrm{TOTAL\ COVER}_t$ refers to the percent cumulative species cover (expressed as a percentage) within a sample unit (Cretini *et al*. 2012).

FQI scores provide measurements of vegetation condition and maturity. Low FQI values can be indicative of early successional vegetation communities, highly disturbed or early post-disturbance evolution, or other presses or pulses that are negatively impacting wetland function. Conversely, high FQI values are more typical in mature, stable, and un-disturbed wetlands.

For all established CWPPRA and CRMS stations within the Project, PR, and SR assessment units, the CRMS Data Download service was used to acquire station-specific FQI data from 1997 to 2015 (CPRA 2016). These data were supplemented with vegetation surveys that were performed as part of this study (surveys conducted in 2014 and 2015). The $\mathrm{FQI}_{\mathrm{mod}}$ was calculated for each vegetation station (using existing and newly surveyed data) within the Sabine Project, PR, and SR areas, using the Louisiana CC list and equations (Eqs. 4 and 5). For species not on the Louisiana Coefficient of Conservatism list, established values from regional lists or neighboring states were used in conjunction with best judgement (Herman *et al*. 2006; Mortellaro *et al*. 2012; Gianopulos 2014).

Resilience

The resilience of ecosystems, defined as the amount of disturbance a system can absorb and still return to a pre-disturbance state or domain (Leps *et al*. 1982; Holling 1996), is a critical factor underlying the sustained production of natural resources and ecosystem services, especially in highly complex and dynamic systems (Gunderson and Holling 2002). Many natural and anthropogenic-induced disturbances, such as storm energies, inundation, and salinity, can be catalysts for long-term impacts on, and degradation of, wetland ecosystems (Steyer *et al*. 2007). Hurricanes and other extreme extratropical storms have been shown to contribute to extensive erosion, breaching, scouring, and compression of coastal wetlands (Meeder 1987; Morton and Sallenger 2003; Suir *et al*. 2011). Similarly, inundation (especially with increased depth and duration) and salinity fluxes are major stressors on wetland plants that can strongly influence establishment, distribution, competition, and switching of vegetation and habitat (Steyer *et al*. 2008a).

Ecosystem resilience was evaluated and compared within restored and natural wetlands by quantifying wetland structure (i.e., wetland area and aggregation index) and function (i.e., productivity, edge density, and floristic quality index) that bracket disturbance events. Tracking these metrics over time provide assessments of wetland resilience, and comparisons of restored wetlands to naturally occurring or hydrologically altered wetlands.

Statistical analyses

Regression and correlation methods were used to assess the relationships among landscape metrics, vegetation characteristics, and wetland productivity. The Statistical Analysis System software version 9.2 PROC GLM procedure was used to perform a one-way analysis of variance (ANOVA) and a means separation test (Tukey's, $\alpha = 0.05$) to evaluate significance of differences between attributes for each assessment unit.

RESULTS AND DISCUSSION

Wetland and Water Trends

Figure 5.2 summarizes the Landsat-derived wetland area and change trends for the Project cycles, PR, and SR units from 1956 to 2016. In 1956, the Project, PR, and SR units consisted of more than 90% land, while in 2000 (prior to wetland and hydrologic restoration activities in the Sabine area) those units were reduced to approximately 5%, 67%, and 52% wetlands, respectively. Between 1956 and 1978 the Project cycle areas (pre-restoration) were largely impacted by hurricanes and canal building, which resulted in significant conversion from wetlands to open water (Miller 2014a). Wetland change in the PR and SR units were largely driven by saltwater intrusion, increased water fluctuations, and tidal scouring due to the channelization of the Calcasieu Ship Channel (Miller 2003). Since 2000, the Project cycles and other restoration measures have been constructed in the Sabine study area. Wetland restoration (CS-28 constructed in 2002, 2007, 2010, 2014, and 2015), along with hydrologic restoration (CS-23 construction in 2001) and erosion protection measures (CS-18 constructed in 1995) were devised to provide direct and indirect structural and functional benefits within the Sabine Refuge and surrounding wetlands. Approximately four years after construction, Cycles 1 and 3 regained and maintained percentages of wetlands that were greater than 90% (Figure 5.2). The Cycle 2 overflow area has experienced similar trends (89% wetlands), however, due to hydrologic restrictions to the Cycle 2 unit (containment dike gaps were not cut for hydrologic flow until 2014), the Cycle 2 unit has only achieved 76% wetlands (Figure 5.2). Although only recently constructed, the early stages of platform consolidation and vegetation within Cycles 4 and 5 mimic those that are observed in the early stages of Cycles 1, 2, and 3.

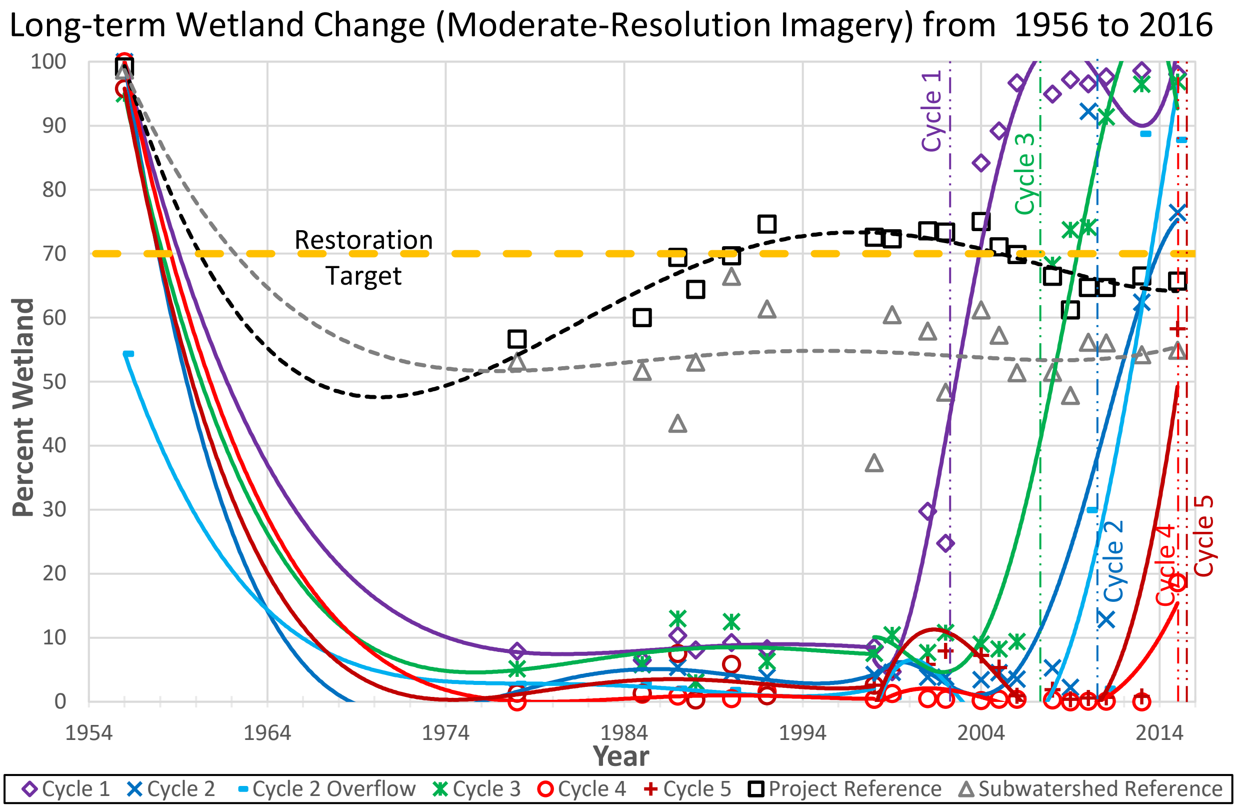

Figure 5.2. Wetland change data (derived from moderate resolution imagery) and trends (3rd order polynomial) within the Sabine assessment units from 1956 to 2016. Vertical dashed lines represent end of construction of each restoration cycle.

Figure 5.2 shows rapid increases in wetland area within the Project areas (purple, green, blue, and red trend lines), as a result of wetland restoration measures, while the hydrologic alterations and erosion control measures in the PR and SR units resulted in slight to moderate increases in wetland area (black and gray dotted trend lines). Figure 5.2 also illustrates how these restoration measures are performing with respect to the 70% wetland target (orange dashed line) that is common for many wetland restoration projects in coastal Louisiana (USACE 2004). The mature Sabine cycles (1, 2, and 3) consist of more than 70% wetlands, while the younger cycles (4 and 5) have recently undergone vegetative colonization, but are expected to follow similar trajectories and eclipse the 70% target within four or five years of construction. In recent decades the percentages of wetland area within the

PR and SR units have primarily resided below the 70% target, and have experienced decreasing trends since Hurricanes Rita (Category 3 storm, September 24, 2005) and Ike (2008).

Figure 5.3 provides a more near-term assessment (1998 to 2016) of the wetland area and change trends that were generated for the Project cycles, PR, and SR units using high-resolution air- and space-borne imagery. However, due to incomplete coverage (i.e., narrow sensor swath and clouds) of the PR and SR units by the higher restoration imagery, representative areas from each restoration unit were selected based on feature type (i.e., wetlands) and maximum coverage. These sub-units are referred to as the Reference South for the PR unit and Reference North for the SR unit (Figure 5.1).

In 1998, the Project cycles consisted of wetland percentages that ranged from a low of 1.2% to a high of 6.7% for Cycle 5 and Cycle 3, respectively. In 1998, the percentage of wetland for the Reference South and Reference North sub-units were approximately 69% and 39%, respectively (Figure 5.3). Similar to the Landsat-derived data, the high-resolution data show rapidly increasing wetland area within each of the constructed Project cycles, with each requiring approximately three to four years to reach the 70% wetland target. The Restoration South and North sub-units experienced more gradual increases in wetland percentages between 1998 and 2016, with the PR and SR sub-units increasing in wetland area by approximately 10% and 30%, respectively. These increases are largely due to restored hydrology in the area, but since they are reference sub-areas in closest proximity to the Project units, they potentially benefit from a reduction in fetch (erosional forces) as related to the constructed cycles.

Landscape Metrics

Landscape metrics were derived from high spatial resolution air- and space-borne imagery and used to measure and compare wetland structure. The utility of these measures are generally found in the ability to track their changes through space and time, and linking those structural changes to ecosystem condition and function. The Aggregation Index (AI) is a landscape metric of interest because it provides measures of landscape condition that positively correlate to wetland integrity and stability (Suir *et al*. 2013; Couvillion *et al*. 2016). Figure 5.4 illustrates AI values and trend lines for each assessment unit. These class-level values were computed using the “wetland” class from 13 classified images from 1998 to 2016. The mean AI values across the period of analysis were 98.1 ± 2.0, 92.0 ± 4.6, 93.9 ± 4.3, 97.3 ± 2.9, 91.5 ± 6.3, 84.8 ± 6.1, 97.7 ± 1.0, and 98.1 ± 0.6, for Cycle 1, Cycle 2, Cycle 2 overflow, Cycle 3, Cycle 4, Cycle 5, Reference South, and Reference North, respectively. A one-way ANOVA with a post-hoc Tukey HSD Test shows that Cycle 5 is significantly different ($p<0.05$) from all other units; Cycle 4 is significantly different ($p<0.05$) from Cycle 1, Cycle 3, Reference South, and Reference North units; Cycle 2 is significantly different ($p<0.05$) from Cycle 3, Reference South, and Reference North units; and Cycle 1 is significantly different ($p<0.05$) from Cycle 2.

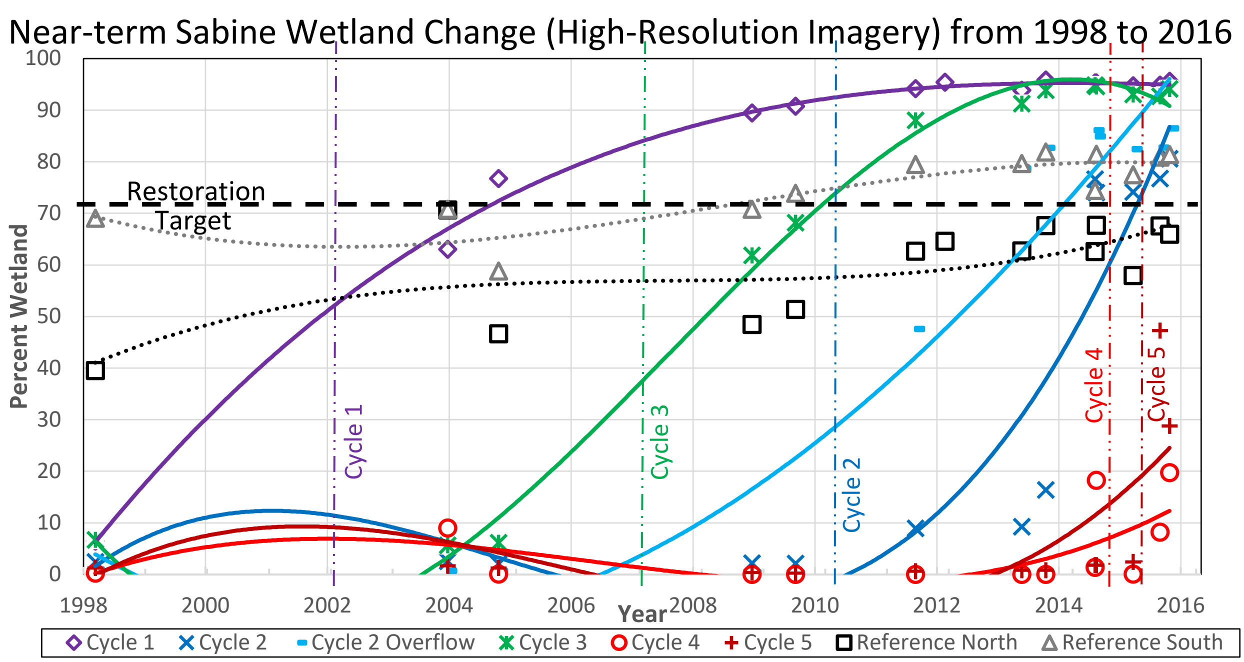

Figure 5.3. Wetland change data (derived from high resolution imagery) and trends (3rd order polynomial) within the Sabine assessment units from 1998 to 2016. Vertical dashed lines represent end of construction of each restoration cycle.

The AI values for the Reference South and North units were high in 1998, and remained relatively constant throughout the period of analysis. These values and trends are indicative of relatively stable landscapes with slight increasing near-term spatial integrity. The starting AI values and overall trends varied considerably for each of the Project cycles. This is a result of the class-level computation of the AI and the fact that some cycle units contained wetland features prior to restoration and some did not. The more mature restoration units, Cycles 1, 3, and 2, exhibited more anticipated values and trends. These included moderate starting AI values and significant increasing trends, post-construction. The increasing AI values in Cycles 1, 3, and 2 are indicative of vegetative establishment on newly constructed wetland platforms. Figure 5.4 also shows that it took approximately four to six years post-construction for the AI values in Cycles 1, 3, and 2 to exceed those of the reference units.

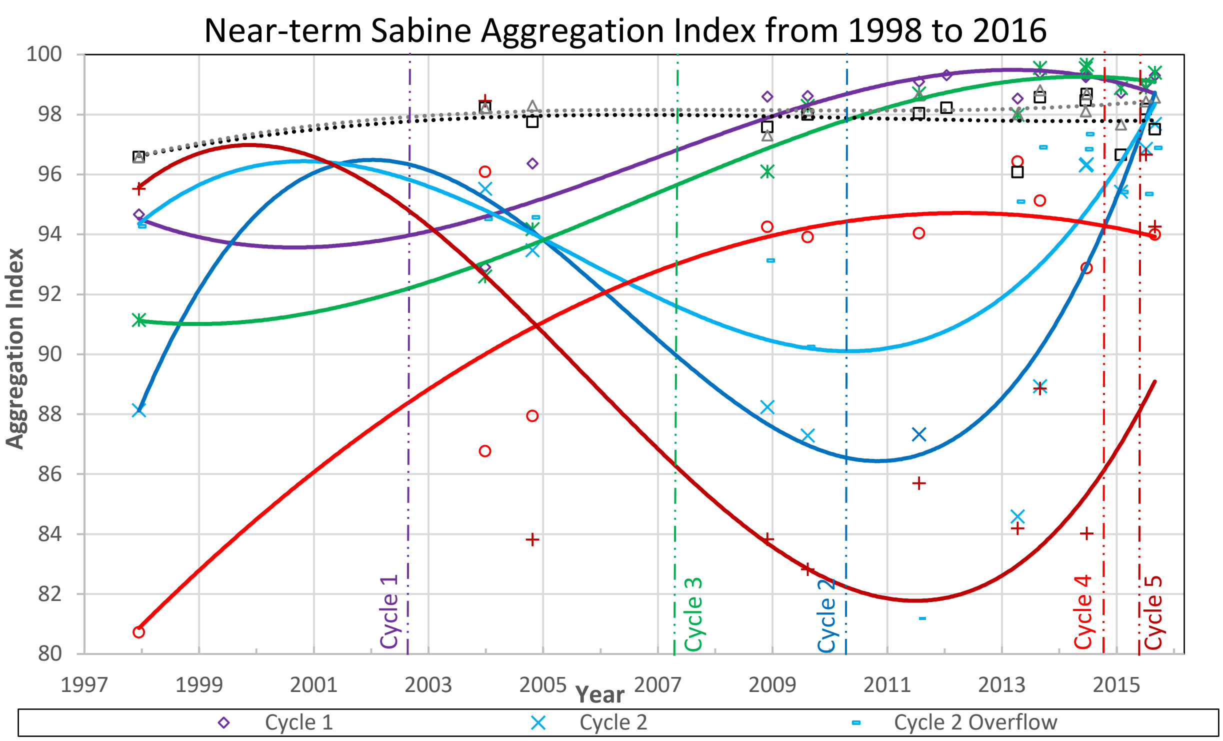

Figure 5.4. Aggregation Index data (derived from high resolution imagery) and trends (3rd order polynomial) within the Sabine assessment units from 1998 to 2016. Vertical dashed lines represent end of construction of each restoration cycle.

Overall, the AI trends corresponded to the wetland change trends that were observed in each assessment unit, which in turn corresponded to restoration measures (i.e., restored hydrology and wetland restoration) in the study area. With increasing wetland area came higher levels of aggregation or spatial integrity. This was observed in the reference units and the more mature restoration cycles, but due to site immaturity (insufficient time for material consolidation and vegetation establishment) similar trends were not discernable in recently constructed Cycles 4 and 5.

Normalized Difference Vegetation Index

As with the wetland area evaluations, the NDVI assessments were performed using moderate- and high-resolution imagery. Figure 5.5 illustrates the mean values (per image) and trajectory of Landsat-derived NDVI data within the Project cycles, PR, and SR assessment unit. These represent all qualifying data, regardless of season, across two distinct periods, the pre- (1985 to 2000) and post-restoration (2000 to 2016) periods. The mean NDVI values ranged from near zero for the Cycle 2 overflow area in October of 2008 to 0.57 for the SR unit in September of 2015. The mean NDVI values by assessment unit were 0.26 ± 0.1, 0.19 ± 0.07, 0.18 ± 0.09, 0.22 ± 0.09, 0.1 ± 0.05, 0.13 ± 0.06, 0.35 ± 0.08, and 0.35 ± 0.09, for the Cycle 1, Cycle 2, Cycle 2 overflow, Cycle 3, Cycle 4, Cycle 5, PR, and SR assessment units, respectively. The mean NDVI values for each assessment unit were significantly different ($p<0.05$), except for Cycle 2, which was not significantly different than Cycle 2 overflow; and the PR unit, which was not significantly different than the SR unit.

The lower mean NDVI values in the Project units, specifically during the pre-restoration period, are indicative of low productivity wetlands that were highly degraded by disturbance events (i.e., hurricanes) and alterations to local hydrology. The pre-restoration period values for all Project cycles were lower than those previously reported for brackish marsh (0.45) in the Calcasieu/Sabine watershed basin (Suir *et al*. unpublished). The earlier constructed cycles, 1 and 3, experienced marked increases in NDVI values during the post-restoration period. These cycles experienced increases in mean NDVI values that ranged from lows near 0.2 in 1985, to highs near 0.45 by the end of the post-restoration period of analysis. It took approximately ten years for Cycles 1 and 3 to reach productivity (NDVI) equilibrium with the PR and SR units. The vegetation productivity trends within the Cycle 2 and Cycle 2 overflow units are similar to those that were observed in the early years of Cycles 1 and 3. On the current trajectory, it is expected that the Cycle 2 units will also reach productivity equilibrium with the PR and SR units at approximately ten years after construction. Also, since Cycles 4 and 5 were constructed similarly to Cycle 3 (natural establishment of trenasses and vegetation), they are expected to evolve and function similarly to Cycle 3.

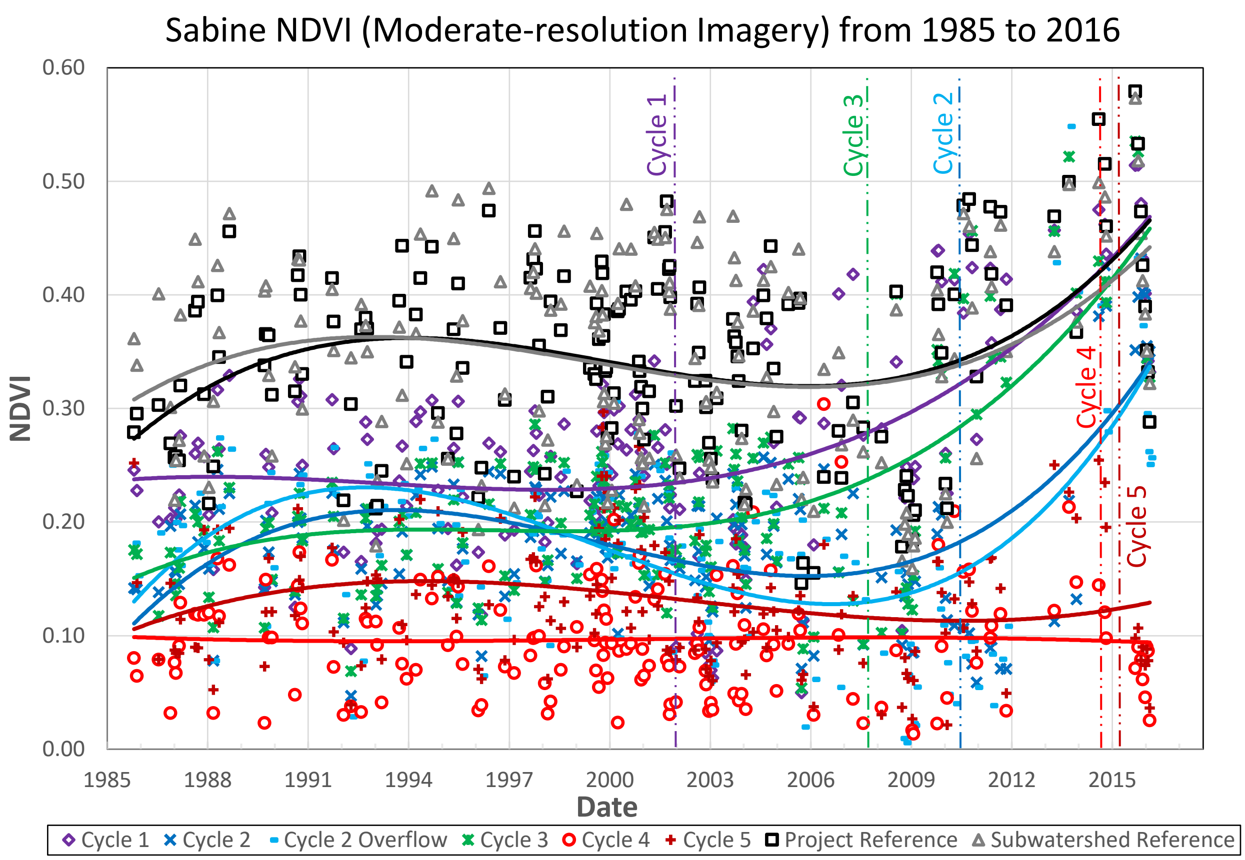

Figure 5.5. Normalized difference vegetation index data (derived from moderate resolution imagery) and trends (3rd order polynomial) within the Sabine assessment units from 1985 to 2016. Vertical dashed lines represent end of construction of each restoration cycle.

Increasing mean NDVI values were also observed in the PR and SR units. These increases are likely due to combined impacts from restored hydrology and erosion control provided by the Project cycles. Figure 5.5 also illustrates reductions in NDVI that were induced by Hurricanes Rita (2005) and Ike (2008). These downward trends in NDVI were succeeded by vegetative recovery, which typically occurs by the end of the next full growing season after a disturbance event (Carle *et al*. 2015; Steyer *et al*. 2013). In all, the PR, SR, and more mature Project cycles have recent vegetative productivity that correspond to the long-term Calcasieu basin mean NDVI (0.45).

Though the existing higher spatial resolution satellite imagery (from DigitalGlobe) lacks the temporal resolution of Landsat imagery, they do provide grain size that are more suitable for discerning smaller landscape features and reducing edge effect in image classification. Figure 5.6 illustrates the mean values (per image) and trajectory of high resolution imagery-derived NDVI data within the Project cycles, Reference North, and Reference South assessment unit from 2004 to 2016. The mean NDVI values ranged from those below zero, for the cycle units prior to their restoration (most cycle areas consisted primarily of open water with small patches of degraded wetlands), to 0.78 within the Cycle 1 unit in 2009. Similar to the Landsat-derived NDVI data, the higher resolution data experienced rapid increases in NDVI values post-restoration. The NDVI maxima for the older Cycles (1, 3, and 2) were reached at approximately five years post-construction, at which point each Cycle experienced periods of nominal reductions, followed by more stable productivity. These trends were similar to those observed in the Reference North and Reference South units, which experienced NDVI maxima during the Hurricane Rita recovery period and slight reductions and stabilization in vegetation productivity from 2009 to 2016.

Across the period of record, the mean NDVI values were not significantly different between assessment units except for Cycles 4 and 5, which were significantly different ($p<0.05$) from all other units; and Cycle 1, which was significantly different from Cycle 2. These differences are primarily due to the age of restoration. The productivity estimates from the higher resolution imagery show that the Project and Reference units had NDVI values that were slightly below the longer term trends within the Calcasieu watershed basin. These assessments show that with maturity, and when sediment deposition increases the marsh platform elevation to a range that promotes vegetation growth, the restored wetlands in this area can achieve, and potentially outperform, vegetative productivity of natural wetlands (Figure 5.6).

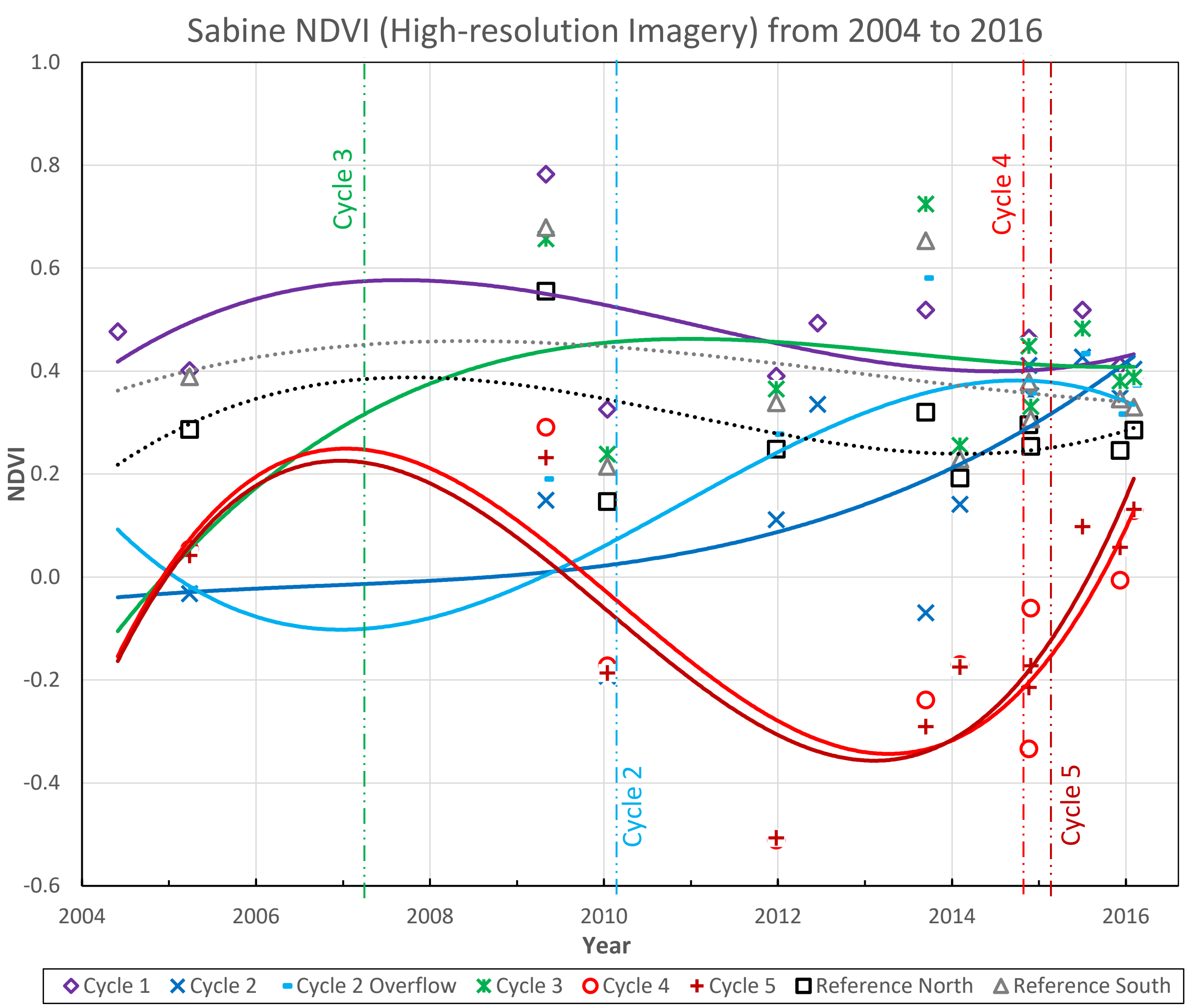

Figure 5.6. Normalized difference vegetation index data (derived from high resolution imagery) and trends (3rd order polynomial) within the Sabine assessment units from 2004 to 2016. Vertical dashed lines represent end of construction of each restoration cycle.

Vegetation Survey

Historically, vegetation survey data have been used to identify the presence of, and track changes in, vegetative species and communities over time. Miller (2014a) describes a 1968 to 1988 shift in the Sabine project area vegetation community from intermediate and fresh dominated marsh species to more brackish species including *Spartina patens* (saltmeadow cordgrass), *Schoenoplectus americanus* (chairmaker's bulrush), and *Schoenoplectus robustus* (saltmarsh bulrush).

Figure 5.7 shows the average vegetation percent cover by species for all survey stations, separated by assessment unit and year. Figure 5.7 also groups and color codes all species based on Coefficient of Conservatism values (Table 5.1). There were 85 different plant species observed across all Sabine units and stations from 1997 to 2015. Species with cover values <3% in a given year were categorized as "other." The first vegetation survey (2004) within the Project assessment unit (upper panel in Figure 5.7), shows that the edge planting as part Cycle 1 (2002) stimulated vegetation expansion, resulting in a *Spartina alterniflora* dominated landscape (57.5%) with a total cover of 59.5%. Hurricanes Lili (Category 1 storm, October 3, 2002) and Rita (2005) significantly impacted vegetation communities along the central and western portions of coastal Louisiana. Hurricane Rita reduced the percent cover within Cycle 1 to 1.8% in 2005, but the unit recovered to 90% and 81.5% cover by 2006 and 2007, respectively. *Spartina alterniflora* remained the dominant species during this recovery, accounting for 90% and 76.6% of the total cover, respectively. By 2008 the *Spartina alterniflora* monoculture within the Project sites began to shift to a vegetative assemblage of common (CC = 7-8) and dominant (CC = 9-10) species. This was due in part to the construction (2007) and natural colonization of Cycle 3. From 2011 to 2015 the typical vegetation profile for Project sites had total cover values between 75% and 87%, and consisted primarily of *Spartina alterniflora*, *Distichlis spicata* [Coastal Salt Grass], *Schoenoplectus robustus*, *Borrichia frutescens* (bushy seaside tansy), *Iva frutescens* (Jesuit's bark), and nominal percentages of "other" species.

The first vegetation surveys in the Sabine PR assessment unit occurred in 1999 and exhibited a total of 88.6% vegetation cover (Figure 5.7). The PR unit consisted primarily of *Spartina patens* (14.9%), *Distichlis spicata* (8.8%), *Schoenoplectus americanus* (7.8%), *Schoenoplectus robustus* (7.4%), and the “other” class (22 species), which accounted for 26.1% of the total cover. By 2001 and 2002 the PR sites were dominated by *Schoenoplectus americanus* and *Distichlis spicata*, with some lower percentages of *Spartina patens* and *Paspalum vaginatum* (seashore paspalum). By the 2005 surveys, the average total cover per site decreased to 44.4% and consisted of only two species, *Spartina patens* and *Paspalum vaginatum*. This change in cover was directly related to hurricane impacts. From 2006 to 2015, the PR sites exhibited a slow recovery and reestablishment of vegetation, with higher percentages of the “other” class, followed closely by increasing percentages of disturbance species (CC = 1–3) and more recently by vigorous wetland species (CC = 4–6).

In 1997, the SR stations consisted primarily of *Schoenoplectus californicus* (California bulrush, 19.1%), *Spartina patens* (15.4%), and *Paspalum vaginatum* (7.7%) (Figure 5.7). The SR stations in 1997 also consisted of 31 species that were categorized as “other”, accounting for 20.9% of the total cover. The dominant species persisted throughout the period of study, but they were occasionally equaled or surpassed in cover by *Iva frutescens* (max 14%), *Distichlis spicata* (max 11.9%), and *Leptochloa fusca* (Malabar sprangletop; max 22.1%).

The Sabine Project cycles experienced rapid vegetation establishment followed by a transition to higher diversity and colonization by common and dominant species. The PR and SR units also consisted largely by common and dominant species prior to Hurricane Rita, however, the PR unit transitioned to dominant with vigorous wetland species while the SR unit transitioned to assemblages with higher numbers of disturbance species and higher percentages of cover.

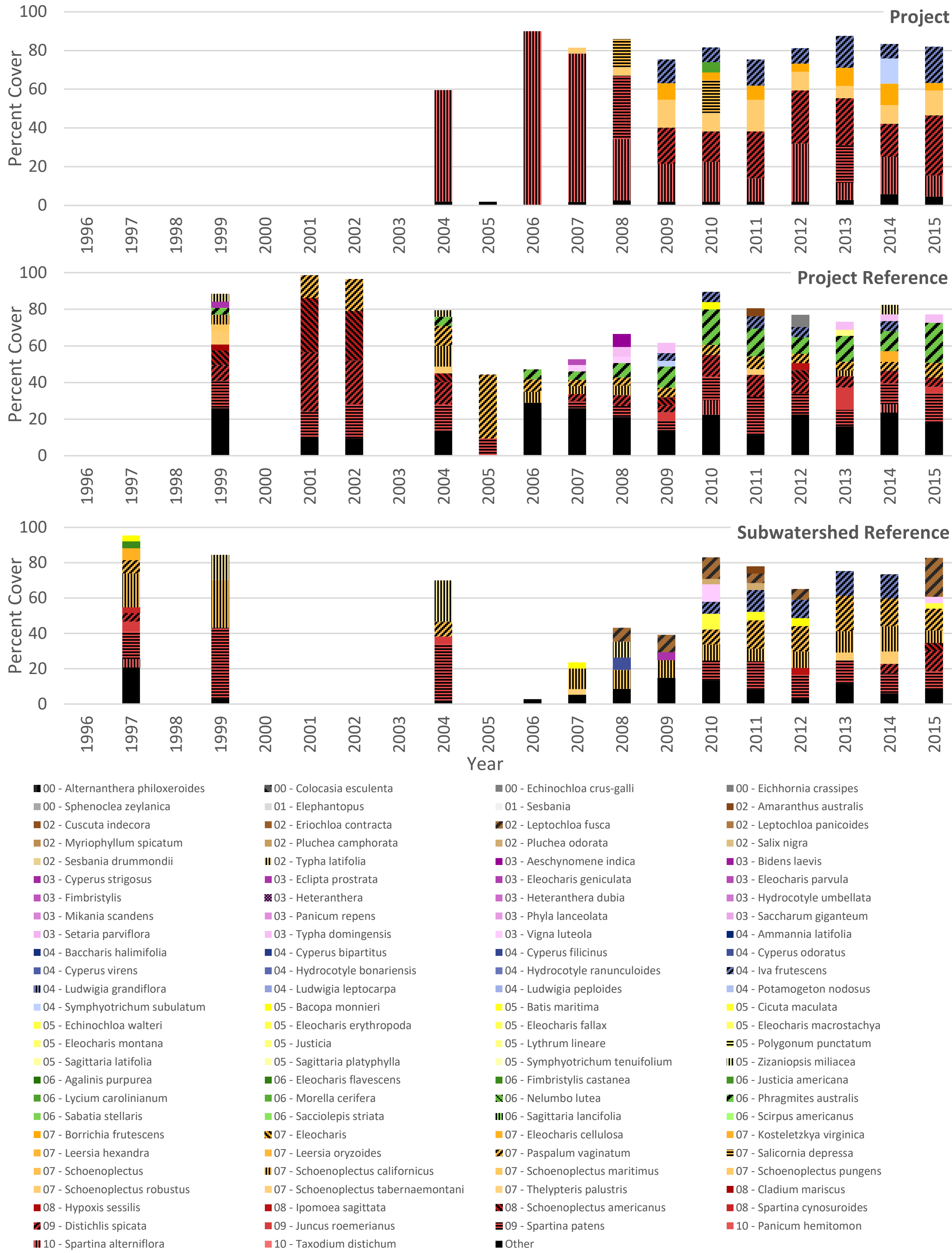

Figure 5.7. Percent cover and Coefficient of Conservatism values for species within the Sabine Project, Project Reference, and Subwatershed Reference assessment units (CRMS and CWPPRA).

Floristic Quality

FQI_{mod} scores were calculated for each survey site within each Sabine assessment unit from 1997 to 2015 (Figure 5.8). The Project sites (red dots), consisting of Cycle 1 and Cycle 3, were first surveyed in 2004 (post construction of Cycle 1) and last surveyed in 2015. The PR sites (green squares) and SR sites (yellow triangles) were both surveyed from 1999 to 2015. Trend lines (3rd order polynomial) within Figure 5.8 show the trajectories of FQI_{mod} values across each assessment unit's period of analysis. The Subwatershed Reference unit data and trend line show a landscape with rapidly declining floristic quality during the Hurricane Rita and Ike disturbance period (2005 to 2009) and equally rapid increase in FQI during the recovery period (2010 to 2015). This is indicative of a susceptible system and corroborates previous studies that have shown significant wetland area and function loss due to hurricanes, high salinity events, increased water fluctuations, and tidal scouring (Barras 2005; LCWCRTF 2002a; Miller 2014b). The Project Reference unit data and trend line show a landscape that was on a declining trajectory but stabilized in 2007 and subsequently experienced an increase in FQI_{mod} score. The long-term degeneration that has occurred in this area is evident from 1999 to 2005 (Figure 5.8), however a CWPPRA project aimed at restoring hydrologic connectivity was completed in the PR unit in 2001, and its impact can be observed in the increasing FQI_{mod} scores from 2007 to 2015. The Project unit data and trend line show a landscape with early increasing floristic quality, however the FQI_{mod} scores have stabilized since 2012, which may be attributable to upper functional limits for this system.

The Project unit FQI_{mod} data and trends are indicative of the rapid colonization and vegetative growth that are common in newly constructed wetlands. The Project unit average FQI_{mod} score from 2010 to 2015 was approximately 80. This coincides with the ideal range for Chenier Plain brackish marsh that was reported by Cretini *et al*. (2012) (Table 5.2). Since construction, the Project units have mostly maintained higher floristic quality than the reference units, indicating that they were more resistant to large disturbance events.

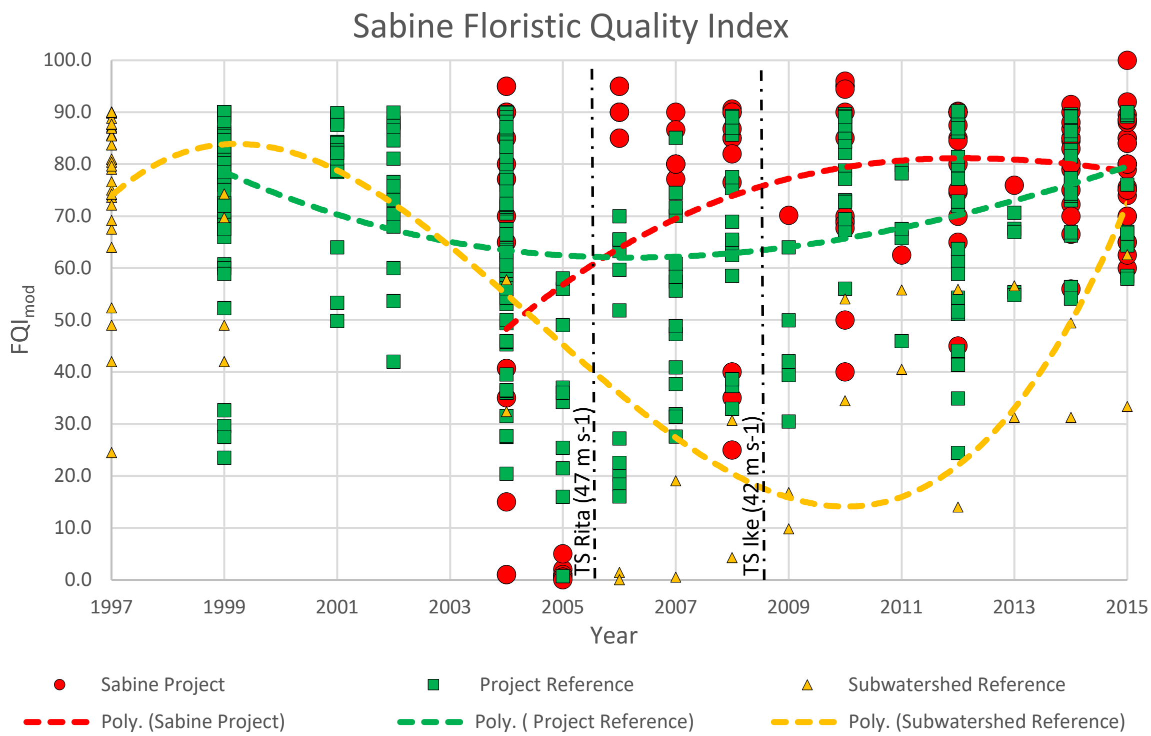

Figure 5.8. Floristic Quality Index (FQI_{mod}) scores for all survey stations within the Sabine assessment units by year. Vertical dashed lines represent tropical storm (TS) activity (wind speed m s-1) and the vertical solid line represents a salinity spike.

Table 5.2. Preliminarily established ideal range for vegetation indices in Louisiana's principal geological settings. Courtesy of Cretini *et al.* (2011).

Geological setting	**Habitat type**	**FQI_{mod}**
Inactive deltaic plain	Fresh marsh	>80
	Intermediate marsh	>80
	Brackish marsh	>80
	Saline marsh	>80
Active deltaic plain	Fresh marsh	>70
	Intermediate marsh	>70
Chenier plain	Fresh marsh	>80
	Intermediate marsh	>80
	Brackish marsh	>80
	Saline marsh	>80

Resilience

To further test the resilience of restored marsh to disturbance events, comparisons of key ecosystem metrics were made between Cycle 1 and the Reference North and South assessment units. Three principal disturbance events occurred within the Sabine study site during the post-construction period (2002 to 2016). These disturbances consisted of Hurricane Rita (landfall on 24 September 2005), with sustained winds of 47 meters per second (m s-1) and a storm surge height of approximately 1.3 m; Hurricane Ike (landfall on 13 September 2008), with sustained winds of 42 m s-1, storm surge height of approximately 3 m, and major flooding and inundation (Figure 5.9); and a high salinity event in 2009, with maintained salinity above 30 from 4 July to 18 August (38.8 salinity maxima on 5 August 2009). The elevated salinity levels were likely due, in part, to a period of moderate to severe drought that occurred between June and July, 2009 (NOAA 2009), which caused reduced inflow of freshwater into adjacent wetlands and estuary. By comparison, the average salinity across the post-construction period was 17.7.

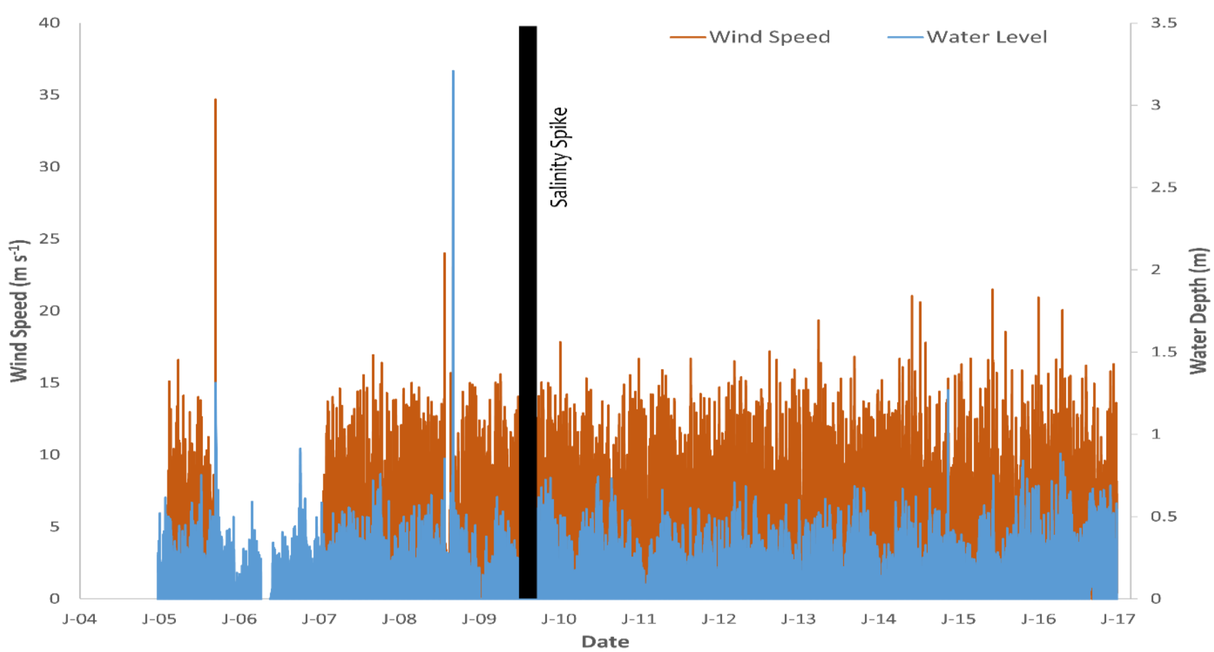

Figure 5.9. Hourly wind speed (meters per second), water depth (meters), and period of elevated salinity within the Sabine study area.

The post-construction period was further subset into a disturbance period (2002 to 2009) and a recovery period (2010 to 2016). Metrics of wetland condition (land change, edge density, aggregation index, NDVI, and FQI) during these periods provide measures of direct and indirect impacts from the disturbance events, and the recovery or resilience of Sabine wetlands (reference and restored). Table 5.3 provides change rates (slope) for each measure of resilience by assessment unit across the disturbance and recovery periods of analysis. Comparing Figure 5.3 and Table 5.3 shows the wetland gains that were achieved in Cycle 1 during the restoration period were maintained during the disturbance period, increasing in wetland area at a rate of 3.28 percent per year. The wetland change trends in the Reference South and Reference North units during the disturbance period were significantly lower than Cycle 1. The Reference South and Reference North rates of change were 0.92 and -2.94 percent per year, respectively. The reduced gain rate in the Reference South unit, and the rate of loss in the Reference North unit, are indicative of wetlands that experienced direct disturbance impacts. This is further evidenced in the wetland and water classified images within the disturbance period, which show storm-formed features (i.e., plucked marsh and amorphous ponds) in the Reference South and North units. The wetland change rates during the recovery period were 0.52, 1.12, and 2.19 for the Project, Reference South, and Reference North units, respectively. The Cycle 1 percentage of wetland and wetland change rate is indicative of a landscape that was approaching the upper wetland gain limits (greater than 90% wetland during the recovery period) and exhibited little to no impacts from the disturbance events. The Reference South unit wetland change rate is indicative of a landscape with minimal recovery after equally minimal disturbance impacts. The Reference North unit wetland change rate is indicative of a landscape with moderate recovery after equally moderate disturbance impacts. It is postulated that these differences are largely due to the erosion protection that is provided by the containment dikes around the Project cycles.

Table 5.3. Change rates (slope) of percent wetland, edge density, aggregation index, normalized difference vegetation index, and floristic quality index for the project, reference south, and reference north units across the disturbance and recovery periods of analysis.

		Project		Reference South		Reference North	
Period	Metric	Slope (per year)	SE	Slope (per year)	SE	Slope (per year)	SE
Disturbance	Percent Wetland	3.28	1.34	0.92	4.19	-2.94	4.14
Recovery	Percent Wetland	0.52	0.24	1.12	1.27	2.19	0.61
Disturbance	Edge Density	-77.94	16.91	31.76	20.74	6.65	1.21
Recovery	Edge Density	-7.72	5.30	-9.76	25.94	9.41	13.61
Disturbance	Aggregation Index	0.96	0.51	-0.21	0.05	-0.13	0.11
Recovery	Aggregation Index	0.07	0.04	0.09	0.06	-0.01	0.12
Disturbance	NDVI	0.07	0.03	0.06	0.01	0.06	0.01
Recovery	NDVI	-0.02	0.07	-0.02	0.42	-0.02	0.20
Disturbance	FQI	0.75	0.31	-0.29	0.10	-0.73	0.45
Recovery	FQI	0.03	0.07	0.08	0.09	0.33	0.22

The landscape pattern metrics and trends selected for use in this assessment (edge density and aggregation index) provide measures of wetland structure and spatial integrity. In wetland landscapes that are predominantly land, as with all post-construction period Sabine assessment units, decreasing rates of edge density and increasing rates of aggregation are indicative of water features that are converting to emergent marsh or where vegetation is establishing on newly construction wetlands platforms. The opposite is true of landscapes with increasing edge density and decreasing aggregation. For the disturbance period, Cycle 1 experienced a high negative rate of change in edge density (-77.94) and a positive rate of change in aggregation index (0.96) (Table 5.3). The rates of change were more moderate during the recovery period, -7.72 and 0.07, for edge density and aggregation index, respectively. These trends are indicative of a wetland landscape that was establishing, maturing, and resilient to episodic disturbance events. For both reference units, moderate rates of change were positive for edge density, and negative for aggregation index, during the disturbance period. These combinations are indicative of wetland landscapes with increasing edge and decreasing aggregation due to direct hurricane impacts (i.e., scouring and edge erosion).

Recovery in the reference units were dissimilar, with moderate recovery in both structural metrics for Reference South, and continued degradation in the Reference North unit.

The NDV and FQ indices provide measures of vegetative productivity and quality, both serving as indicators of wetland function and condition. Highly functioning wetlands typically have high NDVI and FQI values, while disturbed or low functioning wetlands have lower NDVI and FQI scores. The NDVI rates of change were moderately positive for all assessment units during the disturbance period, and nominally negative for all units during the recovery period (Table 5.3). Though Cycle 1 NDVI rates were higher than the reference units, they were only significantly different from the Reference North rates ($p<0.05$). These results indicate that wetland productivity in the Sabine assessment units might be more sensitive to salinity disturbances and less sensitive to storm energies. Though the FQI rates of change in the Cycle 1 unit were positive during the disturbance (0.75) and recovery (0.03) periods, the rate was significantly higher in the disturbance period. This corresponds to a landscape that was quickly vegetating during the disturbance period, and experienced an increasing FQI rate of change due to increasing vegetative establishment and cover. The lower FQI rate of change in Cycle 1 during the recovery period is likely due to functional limits (FQI above 80 during recovery period) and possibly some salinity impacts. Both reference units experienced decreasing rates of FQI during the disturbance period and increasing rates of FQI during the recovery period. These are more typical trends in floristic quality rates, where vegetative quality and quantity are quickly reduced during and soon after disturbance events, followed by recovery over subsequent growing seasons.

The measures of disturbance and recovery used in this study show that the restored wetlands largely experienced insignificant impacts from disturbance events, while the reference wetlands were significantly (for most measures) impacted. These differences are potentially related to the protection (i.e., containment dikes) and function (i.e., altered hydrology) that the large volume of placed sediments and subsequent vegetation provide by limiting storm energies and reducing the duration of increased inundation and salinity. Ultimately, the restored wetlands were more resistant to disturbance events and more resilient across all assessment periods.

CONCLUSIONS

Vegetation provides one of the best indicators for assessing the condition and performance of wetlands (Fennessy *et al.* 2002). However, using standard approaches with vegetation classification and cover data to assess wetland condition and restoration performance can be demanding, especially with long periods of analyses and large quantities of data. Though these standard measures provide assessments of vegetation species presence and abundance (percentage of cover), using these measures to compare the condition of one wetland area to another would benefit from complementary methods more aligned to assess structure and productivity across larger areas.

This study used vegetation survey data, in addition to wetland and water classified data, landscape metrics, NDVI, and FQI data to evaluate change and trends in restored wetland condition, function, and resilience, and compared those to reference wetlands. Across all measures, the restored brackish wetlands reached structural and functional equivalency to reference wetlands at approximately three to five years after construction. With adequate maturity, the restored wetlands outperformed the reference wetlands in the metrics analyzed in this study, having higher percentage of wetland, wetland aggregation (i.e., spatial integrity), aboveground vegetation productivity, and floristic quality. The restored brackish wetlands also demonstrated higher levels of stability, providing more resistance to disturbance events (i.e., hurricanes, inundation, and salinity events), and experiencing reduced levels of flux (i.e., transitional phases of invasive and disturbance species) during the recovery period.

The results of this study show that the combination of remotely collected and *in situ* data, in addition to derived metrics and indices, provided adequate measures of wetland performance (structure and function) and were able to reflect wetland resilience to and recovery from disturbance events. Ultimately, these data and methods provide helpful knowledge elements that allow for inventorying and monitoring of wetland resources, forecasting of resource condition and stability, and adaptive management strategies.

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CHAPTER 6 – SUMMARY

Louisiana consists of approximately 40% of all the coastal wetlands located in the conterminous United States, but has accounted for approximately 90% of all coastal marsh loss since the 1930s. These coastal Louisiana losses—which operate on varied time scales, and as a result of both natural and man-induced events—jeopardize the nation's most productive estuaries, largest coastal channel water-borne commerce, and most critical oil and gas infrastructure. To combat these losses many restoration projects have been constructed or planned throughout coastal Louisiana. Typical goals of wetland restoration efforts are to conserve, create, or enhance wetland form, and to achieve wetland function that approaches natural conditions. Failure to adequately maintain elevation, hydrology, sedimentation, or vegetation in restored wetlands can ultimately lead to degrading conditions. Therefore, accurate and efficient monitoring of restored wetlands is necessary to evaluate functional components and capacities, to assess the performance over time, and resistance to disturbance events (natural and anthropogenic). However, monitoring and assessing the benefits and performance of restored wetlands are often hindered by budget and resource constraints. As a result, knowledge gaps persist about the short- and long-term ecological evolution of restored and nourished sites, potentially resulting in inefficient management of sediment resources. Therefore, these studies used remotely sensed data and methods to evaluate ecosystem and project targets, quantify environmental benefits; assess the geomorphic factors and ecological processes that govern wetland creation and nourishment applications; and evaluate project performance and resilience. These measures are helpful for maximizing the application and adaptive management of sediment for wetland creation and nourishment.

The overarching goal of this study was to gain an improved understanding of the effects of sediment introduction into wetland systems and the factors influencing the establishment, function, evolution, and resilience of restored wetland vegetation communities. This was achieved by considering the coast- and basin-scale impacts of river connectivity and sediment availability on wetland productivity and stability—and the use of an exceptionally large data set to derive and evaluate soil characteristics and processes (i.e., carbon accumulation) as functions of wetland restoration. To evaluate more specific uses of sediment for ecosystem restoration, and their impacts on wetland form, function, and resilience, two project-scale studies were performed at the far-east (Chandeleur Islands) and west (Sabine Refuge) reaches of coastal Louisiana. These smaller scale-studies evaluated the effects of berm sediment redistribution on island elevation, habitat, productivity, and floristic quality; and the ability of constructed wetlands to reach structural and functional equivalency to reference wetlands.

The specific objectives of these studies were to: (1) assess correlations between wetland productivity and riverine influence; (2) evaluate trends in wetland stability and correlations to productivity and river connectivity; (3) compile a comprehensive set of soils data from restoration and reference wetlands in coastal Louisiana; (4) map the spatial distribution of relevant wetland soils characteristics; (5) compute carbon sequestration rates for restored and natural wetland sites; (6) compare soil function across type and age of restoration measure; (7) evaluate the redistribution of berm sediment within the northern Chandeleur Island system; (8) assess berm sediment impacts by evaluating the quantity and quality of existing and new emergent vegetation as a function of redistributed sediment; (9) evaluate and compare structural changes of restored wetlands to naturally occurring reference wetlands; (10) quantify the quality and functional changes of restored wetland and compare to reference wetlands; (11) assess the resilience and recovery of restored wetlands to short-term episodic events (i.e., tropical storms and salinity spikes); and (12) demonstrate the overall efficiency and effectiveness of remote sensing methods for wetland monitoring and consider the implications of these data for future restoration and adaptive management activities.

Due to their rapid growth rates and direct response to environmental stressors and disturbances plants are excellent indicators for assessing the condition and performance of wetlands. However, solely using standard field-collected data to assess wetland condition and restoration performance can fail to account for key structural components, and can be challenging, especially in studies with large spatial and temporal requirements. Fortunately, the ability to understand the spatial and temporal dynamics of wetland landscapes is enhanced by recent advancements in remote sensing systems and analytics.

Remote sensing and landscape analyses provided enhanced techniques for evaluating landscape features, and specifically wetland structure and vegetation productivity. Satellite imagery were used to perform multi-temporal and -spatial scale assessments to quantify wetland productivity and stability metrics, and evaluate changes and correlations to reference wetlands and select ecosystem presses and pulses. Specifically, remote sensing, field data, and landscape metrics were used in these studies to evaluate wetland area, elevation (digital elevation models), wetland edge (edge density), stability (aggregation index), vegetation productivity (Normalized Difference Vegetation Index), and vegetative quality (Floristic Quality Index) across varying restoration measures and scales in coastal Louisiana.

The results show that Louisiana wetland productivity is highly associated with seasonality and vegetation zones, susceptible to episodic events (i.e., hurricanes and floods), and significantly correlated to river connectivity. This was observed under baseline conditions, post-major flood events, and across short and long periods of observation. Positive correlations between landscape stability and river influence were also observed. Likewise, carbon accumulation rates in the coastal zone were generally correlated to hydrogeomorphology and distinctive trends were observed within the Middle Coast, and Chenier and Deltaic Plains. Comparisons of carbon accumulation within smaller-scale assessment units revealed higher rates generally occurred in zones of high river connectivity or in swamp or higher salinity tolerant marsh. Naturally occurring wetlands had significantly higher carbon accumulation rates than restored wetlands, though sediment diversion sites had significantly higher accumulation rates than all other sites. Though carbon sequestration is a relatively new focus of wetland restoration missions, it may prove to be a critical application for climate change management.

The redistribution of berm sediments within the sediment-deprived northern Chandeleur Islands provided elevational lifts to existing back barrier island wetlands and shoals, and in some areas created new shoals in overwash- and drift-zones. Significant increases in the horizontal and vertical extent of existing and newly created features were largely due to the influx of high quality quartz sediments that were redistributed from the Eastern Barrier Berm. The redistributed sediments also provided island feature stability, both in the maintenance of areal extent and the quantity and quality of vegetation present. Across all measures, the restored wetlands that were evaluated as part of this study reached structural and functional equivalency to reference wetlands approximately three to five years after construction. With adequate maturity, the restored wetlands outperformed the reference wetlands, having higher percentage of land, wetland aggregation (i.e., spatial integrity), aboveground vegetation productivity, and floristic quality. The restored wetlands also demonstrated higher levels of stability, providing more resistance to disturbance events (i.e., hurricanes, inundation, and salinity events), and experiencing reduced levels of flux (i.e., transitional phases of invasive and disturbance species) during recovery periods.

From a management standpoint the studies presented herein underscore the importance of high-frequency monitoring to evaluate the development, performance, and resilience of ecosystem restoration measures. In addition to comparisons of restored to natural wetlands, these studies identified unique or differing restoration applications (i.e., planting versus natural vegetation establishment, early versus late reestablishment of hydrologic flow, off- versus near-shore sediment placement, and carbon accumulation across restoration types) to evaluate project and management effects on performance. These evaluations provide greater insight into, and linkages between, management impacts on wetland structure and function. These monitoring techniques and products will also serve as useful components for future inventory and gap analysis, as well as the development of monitoring parameters, rationale for adaptive management, and restoration objectives.

The results of these studies show the combination of remotely collected and *in situ* data, in addition to derived metrics and indices, provided adequate measures of wetland performance (i.e., structure and function) and were able to reflect wetland resilience to, and recovery from, disturbance events. These combinations also demonstrate the importance of river connectivity and sedimentation for wetland productivity and overall spatial integrity. The placement of dredged sediment for ecosystem restoration was also effective in reestablishing critical wetland processes, goods, and services. Therefore, it is recommended that wetland creation and nearshore beneficial use of dredged material not only remain a primary focus of wetland restoration in Louisiana, but these applications should be supplemented and nourished via increased river connectivity to wetland landscapes. Continued evaluations of wetland productivity and landscape configuration, along with other presses and pulses, will provide a greater understanding of ecosystem drivers and sediment importance for long-term management of wetlands and coastal resources.

VITA

Glenn Michael Suir is a native of Lafayette, Louisiana. He earned a Bachelor of Science degree in Environmental and Sustainable Resources from the University Louisiana at Lafayette in 1999, followed by a Master's of Science degree in Agronomy from Louisiana State University in 2002. After completing his master's degree, he spent several years working as a research associate at Louisiana State University. His research focused on spatial hydrology, water quality modeling, and best management practices for soil conservation. In 2004, he began work as a GIS specialist at the US Geological Survey's National Wetlands Research Center. Research there shifted to the use of landscape metrics for ranking the suitability of ecological processes and assessing spatial integrity indices for evaluating alternative wetland restoration action scenarios. In 2008, he began work as a coastal scientist at SJB Group Inc. Work there focused on formulating, screening, modeling, and designing large-scale barrier island restoration concept plans. Currently, he is a research agronomist with the US Army Engineer Research and Development Center where his primary duties include the development of novel methods for monitoring and assessing the performance and resilience of ecosystem restoration applications. In 2013, the Army Corps of Engineers selected Glenn for participation in the Long Term Training Program at Louisiana State University in pursuit of a doctorate in Oceanography and Coastal Sciences specializing in wetland restoration and ecosystem assessment. He expects to graduate with his PhD in May 2018.