FINAL REPORT

CONTROLS ON THE SUCCESSFUL USE OF DREDGED SEDIMENTS FOR THE RESTORATION AND REHABILITATION OF BRACKISH MARSHES ON THE BARATARIA BASIN LANDBRIDGE

BY

IRVING A. MENDELSSOHN[1], MICHAEL D. MATERNE[2], SEAN A. GRAHAM[1], SHEILA ROHWER[2], AND MANOCH KONGCHUM[2]

[1]DEPARTMENT OF OCEANOGRAPHY AND COASTAL SCIENCES
LOUISIANA STATE UNIVERSITY
BATON ROUGE, LA 70803
[2] LOUISIANA STATE UNIVERSITY AGRICULTURAL CENTER,
SCHOOL OF PLANT, ENVIRONMENTAL, AND SOIL SCIENCES
LOUISIANA STATE UNIVERSITY
BATON ROUGE, LA 70803

SUBMITTED TO:

MELISSA HYMEL
LOUISIANA COASTAL PROTECTION AND RESTORATION AUTHORITY
450 LAUREL STREET, SUITE 1500
P.O. BOX 44027
BATON ROUGE, LA 70804-4027

IN PARTIAL FULFILLMENT OF:
CPRA CONTRACT NO. 2503-11-39

DECEMBER 31, 2015

ACKNOWLEDGEMENTS

This final report was produced for the Louisiana Coastal Protection and Restoration Authority (CPRA) as a requirement of CPRA Contract No. 2503-11-39. The authors thank Ms. Melissa Hymel, CPRA project manager, for her conscientiousness and patience during the study. Ms. Hymel provided assistance throughout the project, but especially in acquiring transect elevation data collected for CPRA by various surveying and engineering firms. Thank you is likewise extended to Messrs. Ed Haywood and Greg Grandy for their guidance and support in the initial stages of the project. We also thank Dr. James Pahl and Ms. Hymel for reviewing earlier drafts of the final report. Their suggestions were extremely helpful and greatly improved the final report. Also, we thank Dr. Bruce Williamson for reviewing portions of the final report before submittal. Finally, we acknowledge and thank CPRA for providing the funds to conduct the research presented in this report.

EXECUTIVE SUMMARY

Deltaic marshes are some of the most productive ecosystems in the world and provide critical ecological functions and services. Unfortunately, these wetland systems are rapidly disappearing. The Mississippi River Delta system is a prime example with wetland loss rates of 43 km^2 (10,605 ac) per year from 1985 to 2010. Although a suite of natural and human-induced factors are responsible, excessive inundation resulting from an inability to maintain marsh elevations relative to local water levels due to sediment deficiencies is a common causative agent. Consequently, the majority of wetland restoration approaches proposed for the Mississippi River Delta system incorporates the deposition of sediments, either by sediment diversions or hydraulically dredged sediment-slurries, to maintain and restore marsh elevations to levels that will support emergent wetland plants. This report assesses the use of hydraulically dredged sediment-slurries to restore and rehabilitate brackish marshes as part of the Barataria Landbridge restoration project funded by the Coastal Wetlands Planning, Protection and Restoration Act program (BA-36, Dedicated Dredging on the Barataria Basin Landbridge) and constructed from September 2008 to April 2010. Our overall goal was to evaluate the effects of sediment-slurry application on the restoration of hydrology, soil quality, and vegetative establishment, cover and growth within deteriorating interior brackish marshes. We used high-resolution aerial imagery and on-the-ground sampling of multiple environmental and ecological variables to address restoration success relative to nearby reference marshes.

The Barataria Landbridge study area consisted of 799.2 ha of marsh separated by the Harvey Cutoff canal. The land area west of the Harvey Cutoff is known as *Temple Island* and the study areas east of the Harvey Cutoff is lease property, managed by the *Little Lake Hunting Club*; the two study areas are 255.1 ha and 544.5 ha, respectively. The study areas were further divided into nine treatment cells; six Confined sediment cells, two Unconfined sediment cells, and one Control cell. For the ground-truth component of the research, permanent stations were established to assess a suite of environmental and ecological response variables sampled in the following restoration treatment and reference marshes: (1) Confined High Elevation, (2) Confined Medium Elevation, (3) Unconfined Medium Elevation, (4) Unconfined Low Elevation, (5) Unconfined Very Low Elevation, (6) Degraded Reference (fragmented marsh with numerous ponds), and (7) Healthy Reference (visually intact with little fragmentation). Each of these treatment levels was replicated five times for a total of 35 permanent stations. Sampling events occurred in the fall of 2011, 2012, and 2013. Restoration success was based on two criteria: (1) the ability to reach ecological equivalency with the Healthy Reference marsh and (2) the ability to reach a condition indicative of ecological improvement compared to the Degraded Reference marsh.

Successional changes across the Barataria Landbridge project were determined by creating a geographical information system (GIS) project that integrated four (2008, 2010, 2011 and 2012) base maps. Based on the 2008 DOQQ (Digital Orthophoto Quarter Quads) aerials as a pre-construction starting point, surface features in the Barataria Landbridge marsh were typical of a rapidly subsiding and deteriorating deltaic marsh. Vegetation was generally weak and consisted of small fragmented communities of predominantly *Spartina patens* (marshhay cordgrass). Vegetation was generally shallow rooted in soft unconsolidated muck and often up-rooted in mass during storm events. Soil banks along major water bodies were severely eroded and had collapsed where vegetation had been under-cut by waves, tides, and storm surges. The ratio of

water-to-vegetation was disproportionately high, and numerous shallow-water open ponds dominated large areas of the marsh. In 2008, habitat types were limited and consisted of 62.4% water and 37.6% vegetation, or slightly less than a 2:1 water-to-vegetation ratio.

Spatial changes in the first year post-construction (2010) were dramatic. There was an overall 77.5% increase in vegetative cover and a 47.3% reduction in open water between 2008 and 2010. The Control cell was the only treatment unit within the study area to lose vegetative cover between 2008 and 2010. One notable spatial change in 2010 was the appearance of approximately 8,000 small germinating propagules within the two Unconfined sediment cells. Most of these propagules were vegetative fragments that had sheared away during sediment loading and anchored on top of- or penetrated through thin sediment sheets formed by free-flow sediment deposition.

Between 2010 and 2011 there were only minor habitat shifts within the study area as sediments continued to de-water and cells became increasingly drier. Notable changes in 2011 were the first appearance of large open areas of unvegetated soil in the Confined cell treatments and the rapid expansion of *Spartina patens* propagules in the Unconfined sediment cells. By the end of 2011, the Barataria Landbridge marsh consisted of 69% vegetative cover, 12% open water, and 19% unvegetated bare soils. Changes in habitat between 2010 and 2011 were a 4% increase in vegetative cover, a 65% decrease in open water, and a 100% increase in unvegetated bare soils.

In late August 2012, the Barataria Landbridge marsh was moderately impacted by Hurricane Isaac. Damage to the marsh was primarily flood-related, as we found no evidence of surface scouring or gully erosion within the study area. Damage to the above- and below-ground portions of standing plants were minimal. Buried plant materials, in and around concentrated wrack along shorelines and adjacent to open water, were yellow to light brown, but otherwise remained living. At the end of the 2012 growing season and post-Hurricane Isaac, the Barataria Landbridge project area consisted of 64.0% vegetative cover (51.4 ha), 24.9% unvegetated bare soils (198.7 ha), and 11.2% open water marsh (89.4 ha). The 2012 measurements, in general, showed a slight decline in vegetative cover across all of the project area and significant increases in unvegetated bare soils from 2011. The vegetative components in the Control cell continued to decline, as they had in the previous three years. From 2008 to 2012, the Barataria Landbridge marsh increased in vegetative cover by 71%, decreased surface water areas by 81.8%, and added 198.7 ha (24.9%) of elevated unvegetated soils. In comparison, the un-treated Control marsh lost 41.8% of its 2008 vegetative cover, increased its open water area by 29.9%, and added 3.8 ha (5.1%) of elevated, but unvegetated soils.

A long-term hydrologic and salinity dataset was compiled with the installation of three water data sondes within the project study area; water temperature (°C), specific conductance (µS/cm), salinity (ppt), and water levels (m) were measured on a continuous basis (half-hour intervals) for 24-months beginning July 2011. Water levels between the East Confined and Control treatment units were statistically the same, with a mean of 0.040 m and 0.036 m, respectively (elevations cited are relative to NAVD88, Geoid12A). Water levels within the East Unconfined treatment unit were highest with a mean of 0.162 m. Water levels within the East Confined and Control treatment units were primarily tidal driven and salinity levels were positively correlated with diurnal tides. Mean salinity was highest in the East Confined (5.2 ppt) and Control (3.8 ppt) treatment units, and lowest in the heavily impounded East Unconfined (3.6 ppt) unit. Based on the relatively low mean salinity levels and the post-construction floristic composition, the Barataria Landbridge marsh would be classified as an intermediate to brackish marsh.

Thirty-seven elevation transects were surveyed between 2012 and 2014 by T. Baker Smith and HydroTerra Technologies, LLC, engineering firms contracted by the Louisiana Coastal Protection and Restoration Authority. The number of transects within each treatment varied, but the total number of elevation points that makeup the surface elevation dataset is significantly large and contains approximately 5,900 elevations. We found sediment surface elevations within the four treatment units were variable, but were significantly different between sediment treatment sites. Mean elevations within the East- and West-Confined sites were highest at 0.331 m and 0.327 m, respectively. The East Unconfined unit had the lowest mean surface elevation of 0.180 m, and the West Unconfined had an intermediate mean elevation of 0.257 m. Elevations at the 35 permanent sampling stations had similar elevation trends in that the Confined High treatment had the highest (0.513 m) and the Degraded Reference marsh the lowest (0.149 m). The Confined and Unconfined Medium treatments were lower in elevation (0.309 m and 0.328 m, respectively) than the Confined High treatment, but higher than the Unconfined Low (0.24 m) and Very Low (0.176 m) treatments. Flooding frequency and flood durations within the treatment units and sample stations were a function of sediment elevation and watershed efficiency, with the higher and better drained marshes experiencing the least number of flood events. Flooding frequency within the elevated East Confined treatment unit (with the highest mean sediment elevation) was lowest at 91 flooding events over the 24-month monitoring period; flooding frequency within the East Unconfined site (with the lowest mean sediment elevation) was greatest at 378 events. Flooding frequency in the Healthy reference marsh was 95, and flooding frequency within the Degraded reference marsh was 219 flooding events over 24 months. Relative to the permanent sampling stations, flooding frequency was least within the High Confined treatment site (elevation 0.513 m) with 4 flooding events, and greatest in the Unconfined Very Low (elevation 0.76 m) and Degraded Reference (elevation 0.149 m) with 259 and 219 flooding events, respectively. The Confined Medium (elevation 0.309 m), Unconfined Medium (elevation 0.328 m), and Unconfined Low (elevation 0.024 m) sampling stations had intermediate flooding frequencies of 68, 52, and 130, respectively, over the 24-month monitoring period.

For the most part, we found length of flooding to be negatively correlated with sediment elevation in relatively open unconfined marshes and confounded in marshes that are impounded with restricted water exchange. In ascending order of total time flooded, we found that the East Confined (surface elevation, 0.331 m) had the least amount of cumulative flooding with 6.7%. The Healthy reference marsh (elevation, 0.264 m) was second in both surface elevation and in cumulative flooding, with 7.7%. The West Unconfined treatment unit (elevation, 0.257 m) was third in sediment elevation and third in total flooding with 8.4%. The Degraded reference marsh (elevation, 0.149 m) had the lowest surface elevation of the five units, but second-to-last in cumulative flooding with 20.0%. The East Unconfined treatment unit (elevation, 0.180 m) was second-to-last in surface elevation, but last in cumulative flooding with 42.9%. Total time flooded for the permanent sampling stations had similar trends as flooding frequency, with lowest total time flooded for the Confined High treatment at 1% and highest for the Unconfined Very Low treatment and Degraded Reference marsh (24.3% and 20%, respectively). Other restoration marshes were intermediate in total time flooded with the Unconfined Medium and Unconfined Low permanent sampling stations having total flooding durations of 3.8% and 10.5 %, respectively, and similar to that of the Healthy Reference marsh with 7.7%.

Sediment-slurry application to deteriorated brackish marshes altered soil physico-chemistry, plant recruitment, and ecosystem function, but the intensity and direction (positive or negative) of effects depended on the marsh elevation achieved and the particular response variable. In general, soil bulk density, mineral matter content, soil oxidation (Eh), total salts, and nutrient content increased with higher sediment-loads, while organic matter decreased. Considerable differences were still apparent between the physico-chemical conditions of the restoration sites relative to the Healthy Reference marsh, although final marsh elevations in the restored marshes were at, or approaching, statistical equivalency. Furthermore, there was limited improvement in physico-chemical variables relative to the Degraded Reference marsh; variables that did improve were marsh elevation, soil ammonium and phosphorus, as well as Fe, and potential stressors like conductivity. In fact, when soil chemical variables were combined in a factor analysis, we found equivalency to the reference marshes for some, but certainly not all, of the restored marshes.

Total and average plant species richness in the sediment-restored marshes were equivalent to the Healthy Reference marsh, with the exception of the Confined Medium treatment (total species richness) and Confined Medium and Unconfined Medium treatments (average species richness), which were lower. Also, the dominant plant species in the Healthy Reference marsh were dissimilar to those in the higher elevation restoration marshes, although species composition was more diverse relative to the Degraded Reference marsh. Total biomass (aboveground + belowground, live and dead) in the lower elevation restored marshes was equivalent to the Healthy Reference marsh and significantly higher than for the Confined High Elevation site. In contrast, aboveground biomass (live+dead) did not differ among restoration treatments and reference marshes suggesting equivalency, although the highest (Confined High) and lowest (Unconfined Very Low) restoration treatments tended to have lower aboveground biomass then restored marshes at intermediate elevations. Belowground biomass decreased markedly with increasing elevation and was not equivalent to the Healthy Reference marsh. Marsh elevation was a critical abiotic driver of plant growth response. At higher marsh elevations, soils were drier and more oxidized, and plant total biomass was lower. In addition, higher percentages of silt and clay as well as higher concentrations of soil salts corresponded to lower total biomass.

Functional responses such as accretion, primary production, and organic matter decomposition also responded to sediment-slurry restoration. Accretion rates in the restoration marshes were equal to those in the Degraded Reference marsh, although the Confined High treatment tended to be lower than the reference marsh and was significantly lower than the Unconfined Very Low Elevation treatment. Aboveground production within the restoration area tended to equal that in the Degraded Reference marsh; belowground production mirrored the aboveground trend. Although differences in organic matter decomposition occurred among treatments, these differences were not quantitatively large. Cellulose degradation in the restored marshes was equivalent to the Healthy Reference marsh, except for the Confined Medium Elevation treatment, which was higher, while there were no differences compared to the Degraded Reference marsh. Plant litter decomposition was either equivalent to the Degraded Reference Marsh (for belowground litter) or near equivalent (for aboveground litter). Unfortunately, comparisons of these variables between Healthy Reference and restoration marshes were not possible due to a fire that impacted the Healthy Reference marsh sometime between the 2011 and 2012 sampling periods.

In summary, we conclude that the restoration marshes have not yet reached ecological equivalency with reference marshes for many of the variables and processes measured in this research. However, the restored marshes are now very different for a number of response variables, including total vegetated land cover, compared to the Degraded Reference marsh, which was highly ponded and consisted of few plant species. Thus, sediment-slurry application was successful in converting degraded fragmented marshes into marshes with contiguous vegetative cover and high species richness, more closely resembling the Healthy Reference marsh than the Degraded Reference marsh. Nonetheless, more time for functional development of the restoration marshes is necessary before ecological equivalency with the surrounding higher-quality natural marshes is completely achieved, at least relative to the response variables measured in this research.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Coastal erosion and wetland deterioration are serious and widespread problems affecting Louisiana's coastal zone. With average coastal wetland losses of 4,292 hectares per year from 1985 to 2010 (Couvillion et al. 2011), the long-term social, environmental, and economic consequences will not only affect Louisiana, but also deprive the nation of vitally important navigation, energy production, wildlife, and other wetland-related economic and environmental benefits.

Coastal wetland losses are attributed to a variety of biotic and abiotic factors, such as subsidence, sea level rise, hydrologic modification, herbivory, storm surge, and a variety of anthropogenic disturbances (Boesch 1982, Reed 1995). Although both natural and human-induced factors have been cited as causing wetland loss, many of their effects are mediated through one common agent, sediment availability. Therefore, a sound approach to reducing wetland loss and restoring deteriorated wetlands is the addition and retention of sediment to increase marsh elevations to a level that will support emergent wetland plants (Mendelssohn et al. 1983, DeLaune et al. 1990, Wilsey et al. 1992, Mendelssohn and Kuhn 2003, Slocum et al. 2005, Schrift et al. 2008, Graham and Mendelssohn 2013).

However, a major constraint to the effective use of dredged materials for the creation and restoration of wetland habitat is a limited understanding of how different depths of added sediment affect the hydrologic and edaphic environments that control successful plant establishment and expansion. Too little sediment may have no beneficial or even negative effects (Graham and Mendelssohn 2013), while too much sediment may detrimentally modify the bio-geo-hydrodynamics essential for the maintenance, self-regulation, and sustainability of these systems (Slocum and Mendelssohn 2008, Stagg and Mendelssohn 2010). Thus, there is a need to better integrate biotic and physical elements into dredge-sediment engineering to maximize the successful and beneficial use of dredge material for both large- and small-scale restoration projects in Louisiana's deteriorating marshes. Only with better understanding of the hydrologic and edaphic environments that control successful wetland sustainability will successful rehabilitation of deteriorating wetlands using dredged sediments be predictable.

GOAL AND OBJECTIVES

The overall goal of this research was to assess the effects of sediment-slurry application on the restoration of hydrology, soil quality, and vegetation establishment and growth within deteriorating interior brackish marshes. We addressed three major research questions:

1. What are the effects of sediment-enhanced elevation on vegetative recovery over a range of microhabitats within a deteriorating marsh?
2. What are the physico-chemical factors that control successful vegetative establishment and growth, succession, and resultant ecosystem functions?
3. At what surface elevation does maximum plant recovery occur, and are wetland structure and function equivalent to or surpassing that of un-amended reference wetlands?

Specific objectives of the research were to:

1. Assess elevation effects on vegetative recovery over a range of sediment elevations within the project area and nearby reference marshes.
2. Assay the physico-chemical nature of the sedimentary environment within the project area and adjacent reference sites, and determine their relationship to vegetation recovery and wetland function.
3. Determine vegetative structural and functional recovery across sediment treatments compared to nearby reference marshes.
4. Quantitatively assess spatial and temporal changes across project treatments and nearby reference marshes.

MATERIALS AND METHODS

Study Site

The *Dedicated Dredging on the Barataria Basin Landbridge Project* (BA-36) is located in the Barataria Basin within Jefferson Parish, approximately 11 km southwest of Lafitte, LA, along the southern shorelines of Bayous Rigolettes and Perot on either side of the Harvey Cutoff (29° 34' 23" N, 90° 08' 10" W). The project site included six confined fill areas (target elevation, +0.762 m [+2.5 ft.] NAVD88) with containment dikes constructed from *in-situ* material, five unconfined fill areas with limited confinement immediately adjacent to five of the six confined fill areas, and three borrow areas (two areas in Bayou Rigolettes and one area in Bayou Perot). Sediment was hydraulically dredged from the three borrow areas into the 11 fill areas from March 2009 to March 2010 to fill open water areas, create new marsh, and nourish existing marsh. All dredging was completed on March 9, 2010 (Barowka and Bonura 2010).

Geospatial Assessments, Hydrology/Salinity, and Sediment Elevations

Geospatial Imagery

The objective of the GIS project component was to assemble a chronosequence of high-resolution geo-referenced maps from which a geospatial dataset could be developed and subsequent surface feature changes accurately measured. Successional changes across the life of the study period were determined by creating a geographical information system (GIS) project and integrating four base maps (2008, 2010, 2011 and 2012) using the Environmental Systems Research Institute's (ESRI) ArcGIS 9.1 software. The 2008 pre-construction aerials were flown by the U.S. Geological Survey (USGS) and the 2010, 2011, and 2012 post-construction aerials were flown by Aero-Data Corporation, LLC (Baton Rouge, LA) under a personal services contract for this study. The 2008 pre-construction aerials form a multi-resolution seamless image database (MrSID) constructed from 12 digital orthophoto quarter-quads (DOQQ) flown at 1:12,000-scale and have a 1-meter pixel ground resolution. The remaining three base maps are low altitude aerials (2,000 feet) with a 1:4,000-scale, and are color-infrared photography. Images used in post-construction analyses are high-resolution scans (2,032 dpi), and have a 0.076-meter pixel ground resolution. All of the base maps were geo-registered to the North American Datum of 1983 (NAD83) and to identifiable ground control positions with coordinates acquired from ground survey using ERDAS Imagine 8.7. Figure 1a through Figure 2b are examples of treatment areas from the 2008 pre-construction and the 2010-2012 post-construction base maps. Common points for all four-base maps are included for reference.

Dataset Development and Interpretation

The study site is divided into two discrete land areas separated by the Harvey Cutoff canal. The land area west of the Harvey Cutoff is locally known as *Temple Island* and the land area east of the Harvey Cutoff is lease property, managed by the *Little Lake Hunting Club*. Temple Island is the smaller of the two study areas with 255.1 ha and the Little Lake Club has 544.5 ha. Temple Island and the Little Lake Club study area was further divided into an equal number of treatment cells (four each) and one control cell. Two additional cells found on Temple Island were a deep-water petroleum access canal and one small interior pond. Both are open water cells that were intentionally protected from sediment deposition and remained open throughout the life of the study. There were no equivalent open water sites found within the Little Lake Club study area.

The sediment retention levees, clearly visible on the 2010 aerial (Figure 1b), were used to establish the outer and interior cell boundaries for the confined sediment treatment areas for the east Little Lake treatment area (E-CT) and for the west Temple Island treatment area (W-CT). Because the Unconfined sediment and the control cells lacked physical boundaries (such as retention levees), they were formed by linking salient surface features together to circumscribe the cells. Two free-flow Unconfined sediment cells and one Control cell were defined using this method. Figure 3 is a 2010 aerial image of the entire project study area, showing the location and treatment status of the six Confined sediment cells, two Unconfined sediment cells, one Control cell, and the two open water cells located on Temple Island.

With ArcGIS 9.1 software, a single common geo-referenced control layer was created to restrict surface feature delineations (polygons) to within a single common and fixed boundary across all four base maps. In addition, having one common geo-referenced control layer provided quality control for completing calculations across multiple year base maps. As the project site is an artificially created marsh complex, study-specific nomenclature was identified for each study year and was ground-truthed as additional aerial images became available and surface features added. For example, the 2008 DOQQ aerial represents pre-construction, undisturbed marsh, and consists of only two habitats open water and small areas of relic emergent marsh. However, with each new construction phase, additional landform descriptors were added to better describe changing hydrology, soil, and vegetative characteristics. Maintaining consistent mapping and landform terminology was critical to accurately assessing the temporal and spatial changes throughout the four-year study period. Table 1, *Mapping and Surface Feature Terminology,* is a list of map acronyms, surface feature descriptions, and definitions used throughout the study. Using the landform elements outlined in Table 1, individual surface features, such as vegetation, open water, and bare-soils were drawn as contiguous polygons to create a single master surface feature layer; this procedure was completed for each aerial year included in the study. From an XTools Extension for ArcGIS, like-surface features (for example, all vegetative polygons) located on each master overlay were electronically clipped, separated from their surrounding polygons, and then moved to separate layers. Grouping like-features and moving them to individual layers by year, produced a series of indices that were compiled into a database for analysis.

Hydrology and Surface Elevation

One of the study objectives was to assess the long-term hydrologic/salinity equivalence within the three treatment areas. To accomplish this, three YSI model 600LS water sondes, equipped with multi-sensors and a vented level system, were installed to record changes in both water level and salinity values within each of the study treatment areas. Water temperature (°C), specific conductance (µS/cm), salinity (ppt), and water levels (m) were measured on a continuous basis (half-hour intervals) beginning in July 2011 and ending in July 2013. Water data were collected at three points within the study area: (1) in the Little Lake Camp Canal, immediately adjacent to the Control marsh and outside of any sediment treatment areas (N29.56804; W090.11900); (2) within the Confined sediment treatment area (N29.57488; W090.13335); and (3) within the Unconfined free-flow treatment area (N29.56830; W090.11900). See Figure 3 for the approximate location of the three data sondes.

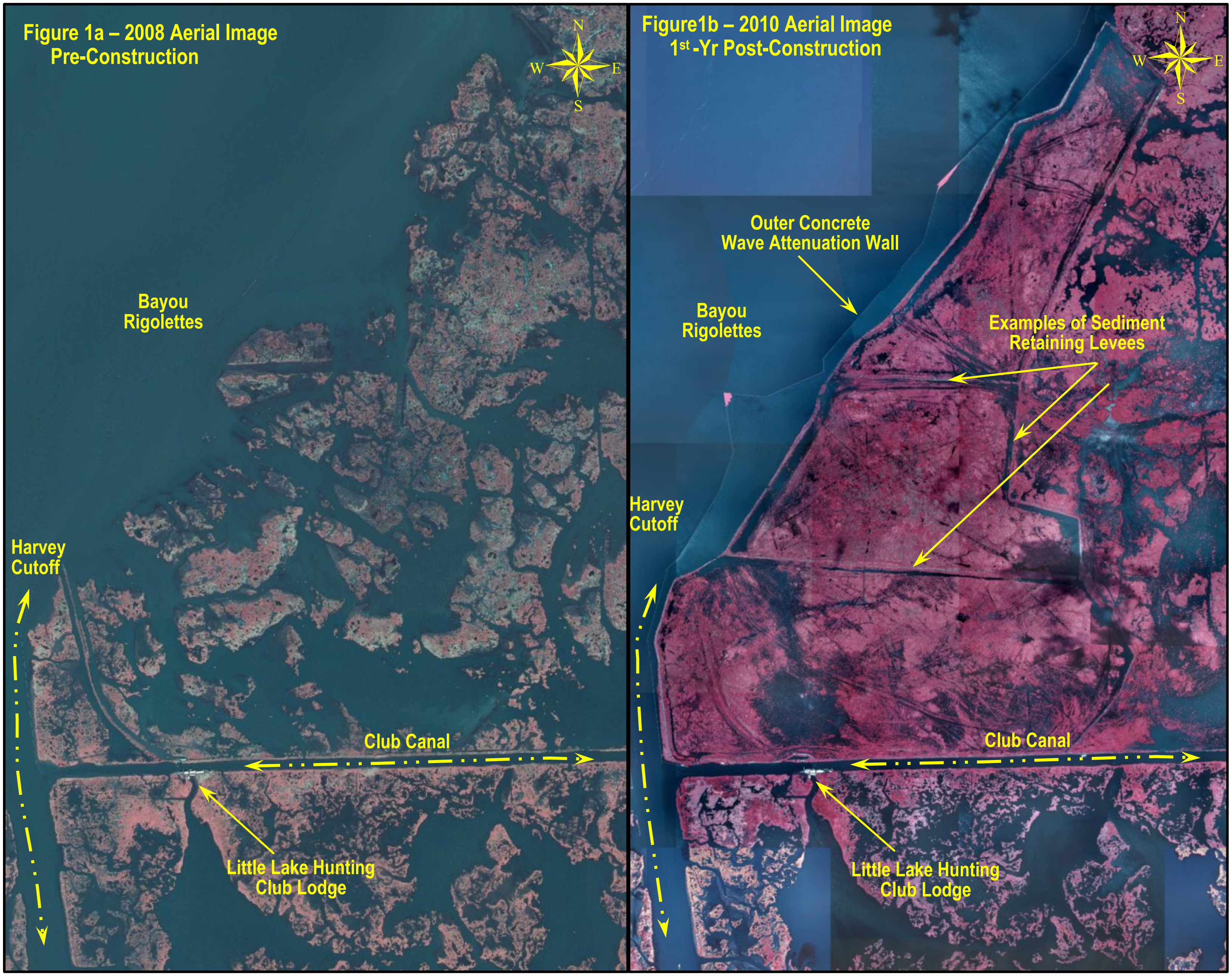


Figure 1a and Figure 1b – Sample areas of the Barataria Landbridge project area; Fig. 1a represents pre-construction marsh conditions and Fig. 1b represents 1st-year post-construction marsh conditions. Examples of the retaining levees are clearly seen in the 2010 image and were used to establish the spatial control boundary used throughout the study period.

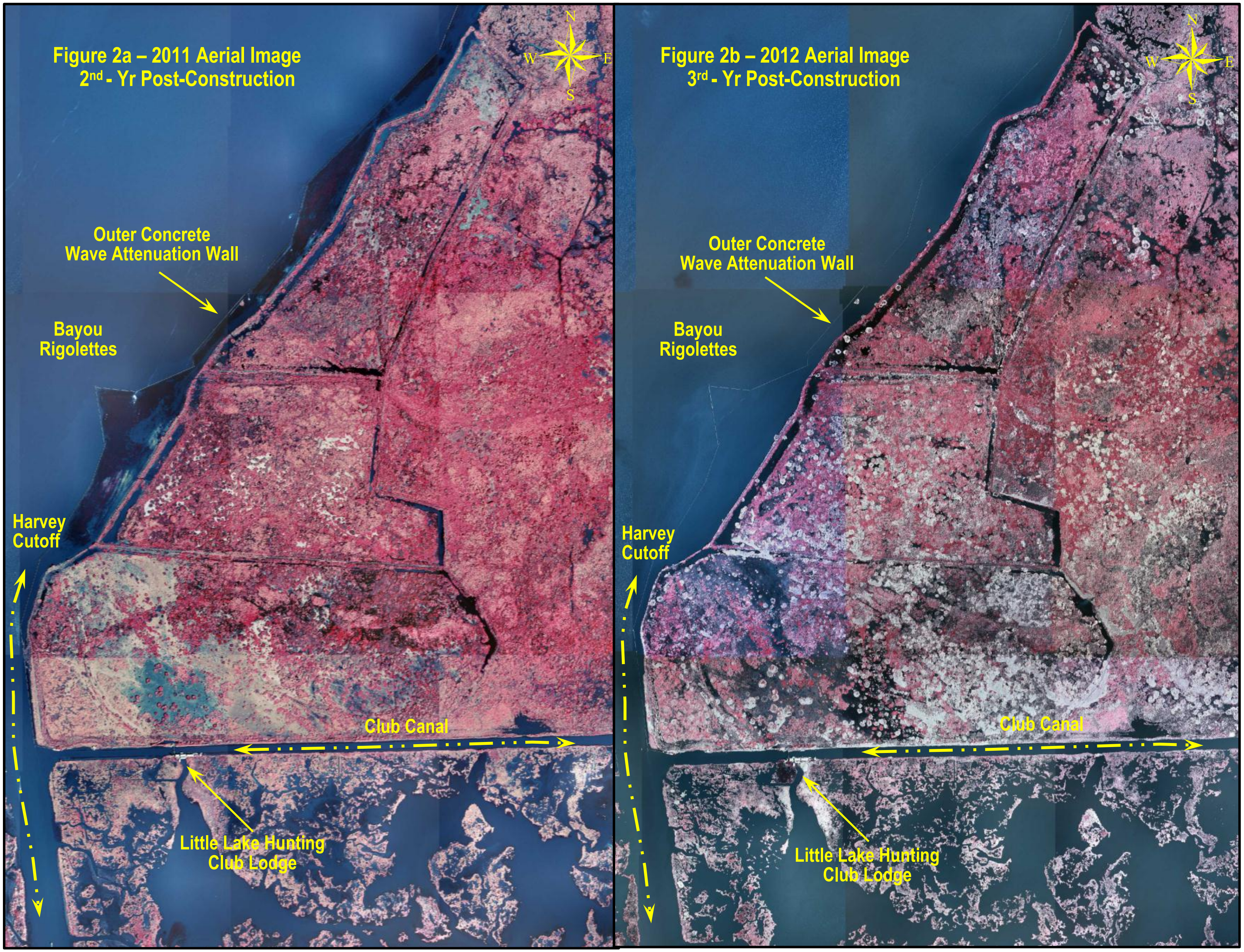


Figure 2a and Figure2b – Sample areas of the Barataria Landbridge project area; Fig. 2a shows the 2011 2nd-year post-construction marsh conditions and Fig. 2b shows the 2012 3rd-year post-construction marsh conditions. Common points are included for reference between aerial years.

Figure 3 – 2010 aerial image with exterior and interior control boundaries delineating 11 cells: six Confined sediment Treatment (CT#) cells, two free-flow Unconfined Treatment (UCT) cells, two Confined Un-Treated (CUT#) cells, and one un-treated Control (CTRL) cell. The locations of the three water sondes are also included.

Table 1 – Mapping and Surface Feature Terminology

SURFACE FEATURES	DEFINITION
Project Boundary	The Project Boundary is the outer boundary that circumscribes the project study area. The outer retention levees, based on the 2010 aerial imagery were used to delineate the Project Boundary. Cells lacking physical boundaries (such as retention levees) were formed by linking salient surface features together to circumscribe study treatment cells such as the Unconfined free-flow and Control cells. The Control Boundary Layer was subsequently used for all other study-year imagery. See Figure 3 for an example of the Project Boundary locations.
Project Sub-Boundaries – Treatment Cell Delineations	Project Sub-Boundaries are internal to the project study site. Sub-Boundaries circumscribe individual internal treatment cells, or other land areas of interest, such as access canals or interior ponds within the project site. Surface features such as vegetation would also be internal sub-boundaries.
Temple Island Study Area	Temple Island is a colloquial name given to the marsh island west of the Harvey Cutoff Canal and south of Bayou Rigolettes. Only the northern end of the Island is included in the study. Cells within the Temple Island study area are referenced with a W- for West Treatment Cells. See Figure 3 for the vicinity and relative location of the Temple Island land area included in the study and for West treatment cell locations.
Little Lake Club Study Area	The Little Lake Club study area is east of the Harvey Cutoff Canal and north of Club canal. The study area is leased property of mixed ownership and managed by the Little Lake Hunting Club organization. Cells within the Little Lake Club study area are referenced with an E- for East Treatment Cells. See Figure 3 for the vicinity and relative location of the Little Lake Club land area included in the study and for East treatment cell locations.
Confined Treatment Cells	The Confined Treatment Cells are referenced as CT# and are containment cells of varying size in which dredge-sediments were pumped until a uniform elevation within the cell was obtained. The study area contains six Confined Treatment cells, three in the Little Lake Club study area (E-CT#) and three in the Temple Island study area (W-CT#). Figure 3 shows the location of both East and West Confined Treatment cells.
Unconfined Treatment Cells	The Unconfined Treatment Cells are open ended, Unconfined cells, where dredge-sediments were allowed to free-flow. A relative flat elevation gradient formed from sheet-flow deposition, sloping from high to low in a west to east direction. Two Unconfined Treatment Cells are included in the study, one on Little Lake Club (E-UCT) and one on Temple Island (W-UCT). Figure 3 shows the location of both East- and West-Unconfined Treatment cells.

SURFACE FEATURES	DEFINITION
Confined Untreated Cells – W-CUT#	The Confined Untreated cells are two open water cells located on Temple Island and identified as W-CUT#. One is a deep-water petroleum access canal, which opens into Bayou Rigolettes, the other is a small shallow-water enclosed pond. Both areas were sheltered from dredge-sediments by protection levees and remained open through the life of the study. Figure 3 shows the location of both East and West Unconfined Treatment cells.
Control Cell – CTRL	One Control (CTRL) cell is designed for the study and is located south of the Little Lake Club marsh. The Control cell is south of the Club Canal and is the marsh area immediately behind and east of the Little Lake Hunting Club buildings. Both the dredging contractor and property managers were confident that this area of marsh was un-impacted by slurry sediments and best represent pre-construction unaltered marsh conditions. Figure 3 shows the location of Control cell.
Water	Water elements found within the study area include any standing open water that appears permanent or semi-permanent. Ephemeral puddle water in shallow surface depressions resulting from climatic events, from extreme tides, or from storm surges were not included. Areas with a predominance of vegetation growing as emergents in standing water were grouped with Vegetation.
Vegetation	Vegetation found within the study area. Vegetative polygons may include individual plants, if large enough to delineate, or an entire plant community. Vegetative polygons were selected on the presence or absence of significant plant materials. Any sparse or thin vegetation growing as an emergent in standing water and where the water element is dominate, was grouped with Water.
Bare Soils – Un-Vegetated	Bare Soils are soils located above normal water elevation and lack any discernible vegetative cover, or areas of sparse vegetation where bare soils are dominate. Mud flats or low marsh soils exposed during low water events were grouped with Water.

Sondes were housed in a vertical structure consisting of a 12' treated 4"x 4" post driven into the marsh to resistance (Figure 4). An individual sonde was attached to a 2" PVC pipe that was then placed within a 3" perforated PVC pipe mounted to the 4"x 4" post with lag bolts. The bottom of the sonde rested on a 6" long hitch-pin inserted horizontally through the 3" PVC and positioned approximately 2" off the marsh floor. A 10' vented data cable was threaded through the top of the 2" PVC and secured in a covered hard-plastic electrical box mounted by a flange on top of the 3" PVC. The sonde structure provided a relatively water-tight, protective housing that allowed for removing and replacing the sonde with minimal tools. Sondes were re-positioned at the same elevation after each quarterly data collection, cleaning, and calibration check, as the bottom of the sonde rested on the top of the fixed 6" hitch-pin located at the bottom of the 3" PVC.

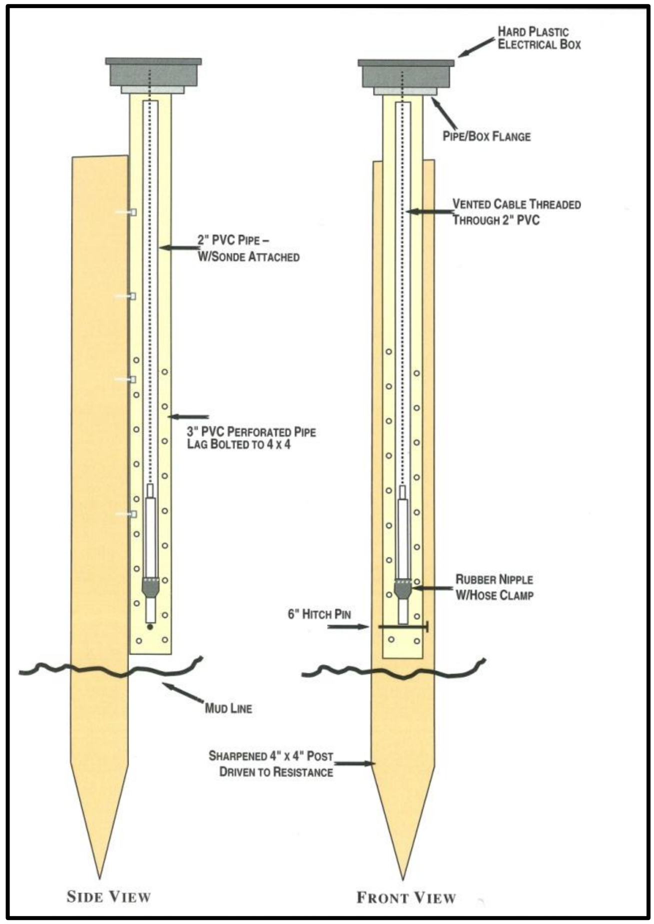


Figure 4 – Schematic of sonde structures

Once established, a differential survey between the three sondes was completed to determine a correction coefficient necessary to equilibrate water level readings across the three treatment units. On December 3, 2012, as part of the BA-125 Northwest Turtle Bay Marsh Creation Project survey, T. Baker Smith, LLC established a NAVD88 Geoid09 elevation with the Control marsh water sonde and with a temporary benchmark (TBM) associated with the Confined treatment cells. Both elevations were subsequently corrected to Geoid12A for consistence with all other ground surface surveys within the project area.

Water data were downloaded quarterly, and sondes were cleaned and recalibrated prior to each quarterly redeployment. For quality control/quality assurance, sondes were checked at the deployment site against a second calibrated instrument prior to re-deploying as an unattended field instrument. All subsequent water elevations were corrected to NAVD88, Geoid12A.

Ground Truth Assessment of Marsh Structure and Function

Sampling Design

Replicate sampling stations were established at specific elevation ranges within five of the six confined fill-areas, five unconfined fill-areas adjacent to each confined fill-area, and at five healthy and five degraded reference areas that did not receive sediment (35 stations in total; Figure 5). Prior to sampling station establishment, soil surface elevations were surveyed relative to NAVD88 (Geoid09) with a real time kinematic (RTK) GPS (see Soil Surface Elevation below) in an effort to locate areas within specific elevation ranges. NAVD88, Geoid09 elevations were converted to Geoid12A as described below and in Table A-1 to insure that all RTK-GPS elevations presented in this report were comparable. Within the five confined-fill areas, stations were established at high (0.42-0.62 m-NAVD88) and medium (0.26-0.34 m-NAVD88) elevations. Within the five unconfined-fill areas, stations were established at medium (same range as medium-confined), low (0.20-0.24 m-NAVD88) and very low (0.15-0.19 m-NAVD88) elevations. Within the healthy and degraded areas, five replicate stations were established at elevations that ranged from 0.22-0.26 and 0.09-0.16 m-NAVD88, respectively.

The seven study areas that served as treatment-levels (hereafter referred to as treatments) within our experimental design were as follows: (1) Confined High Elevation, (2) Confined Medium Elevation, (3) Unconfined Medium Elevation, (4) Unconfined Low Elevation, (5) Unconfined Very Low Elevation, (6) Degraded Reference, and (7) Healthy Reference. The "Healthy" Reference marsh, although appearing vigorous, had a soft, unstable substrate, which was difficult to traverse, and numerous small gullies, resulting in some fragmentation. Hence, healthy, in this context, is relative to the deteriorating Degraded marsh, which contained numerous small ponds.

Samples were collected from each of the 35 permanent stations (7 treatments x 5 replicates per treatment) in the fall of 2011, 2012 and 2013. However, a fire, which occurred sometime between the 2011 and 2012 sampling periods, Hurricane Isaac in 2012, and extensive mammal disturbance, impacted the Healthy Reference marsh and destroyed some stations. These events prevented estimates for accretion, above- and belowground production, and plant decomposition in the Healthy Reference marsh.

Soil Surface Elevation

During the late summer and early fall of 2011, marsh surface elevations were acquired at each of the 35 sampling locations (Figure 5) within the seven treatment areas. We used a Trimble R8 GNSS dual frequency receiver and a Trimble Zephyr 2 Geodetic antenna (Trimble Navigation Limited, Sunnyvale, CA). Connection to the Louisiana State University Center for GeoInformatics RTK Network (C4GNet) provided continuous centimeter-level accuracy for all horizontal and vertical measurements. All vertical elevations are presented in meters and relative to NAVD88 Geoid12A**.** Elevation transformations from earlier geoids, in this example Geoid09, to Geoid12A were calculated as follows:

$$H(12A) = H(09) + [h(2011) – h(2007)] – [N(12A) – N(09)],$$

where H(12A) is the orthometric height relative to Geoid12A, H(09) is the orthometric height referenced to Geoid09, h is the ellipsoid height for a particular ellipsoid, and N is the geoid height for a particular geoid (see Appendix Table A-1 for an example).

Because h and N are based on models as well as assumptions about what realization of NAD83 was actually used for the GPS-derived h, the transformed orthometric heights should be considered estimates (accuracy ± 3 cm). (Source: Michael Dennis, National Geodetic Survey, personal communication by email).

Figure 5 - Map of replicate sampling station locations. Abbreviations: C = confined, U = Unconfined, H = high elevation, M = medium elevation, L = low elevation, VL = very low elevation, DEG = degraded reference, REF = "healthy" reference

Temporal Changes in Soil Surface Elevation

The Louisiana Coastal Protection and Restoration Authority (CPRA) provided elevation survey data for the project area prior to and post construction for the following periods:

Preconstruction

T. Baker Smith, LLC (Houma, LA) surveyed the confined portions of the project area from October 29, 2008 to November 24, 2008. The unconfined sections of the project area and the control areas were not surveyed at this time. Horizontal datum was Louisiana State Plane, South Zone (1702), NAD83 in U.S. feet, and vertical datum was NAVD88, Geoid03 in U.S. feet. Vertical datum was transformed to NAVD88, Geoid12A as described above and converted to meters.

Post-Construction

Elevations were collected during the fall of 2009 and 2012 and during the spring of 2014. In the fall of 2009, HydroTerra Technologies, LLC (Scott, LA) surveyed elevations of the confined project areas 28 days after fill deposition was completed in each of the cells. Horizontal datum was UTM, NAD83, Zone 15, in meters, and vertical datum was NAVD88, Geoid03 in U.S. feet. Vertical datum was transformed to NAVD88, Geoid12A as described above and converted to meters.

T. Baker Smith, LLC surveyed the unconfined sections of the project area as part of CWPPRA Project BA-125 (Northwest Turtle Bay) in the fall of 2012 between October 30 and December 3, 2012. Horizontal datum was Louisiana State Plane, South Zone (1702), NAD83 in U.S. feet, and vertical datum was NAVD88, Geoid12A in U.S. feet. Elevation units were converted to meters.

In the winter of 2013 (December 11 and 12) and the spring of 2014 (March 18 – 26), HydroTerra Technologies, LLC surveyed the confined sections of the project area, similar to 2008 (pre-construction) and 2009 (immediately post-construction). Horizontal datum was UTM, NAD83, Zone 15 N, in meters, and vertical datum was NAVD88, Geoid12A in U.S. feet. Elevations in feet were converted to meters.

In summary, elevations in the confined-fill areas were surveyed four times (Fall 2008, Fall 2009, summer/fall 2011 (sampling stations), and spring 2014), while the unconfined-fill areas were surveyed only twice (summer/fall 2011 and fall 2012). Consequently, our ability to quantify changes in elevation over time was most robust for the confined sites. Degraded and Healthy reference marshes were surveyed only once during the study (summer/fall 2011). Survey points that fell in canals, marsh-adjacent bays, and on man-made structures were excluded from the analyses.

Soil Physico-chemistry

Two soil cores were collected from each station using a 5-cm diameter Russian Peat Borer (Aquatic Research Instruments, Hope, ID) for physico-chemical characterization in the fall of 2011, 2012, and 2013. One core was used for physical characterization of the soil, which included soil bulk density (Blake and Hartge 1986), percent moisture (Gardner 1986), percent organic matter (loss-on-ignition or LOI @ 550 ºC; Christensen and Malmros 1982), and particle size (pipette method; Gee and Bauder 1986). The second core was used to determine soil pH

(1:1 water), conductivity (1:2 water), and exchangeable nutrient concentrations. pH was determined with pH electrode and soil conductivity with a salt bridge. Total salts were derived from conductivity by dividing by 1.28 (Rhoades 1982) and converting to a soil volumetric basis, based on the soil bulk density. Ammonium-nitrogen (NH_4^+-N) was determined using the automated phenate method (APHA 2005) following 2M KCl extraction and 0.45-µm filtration (Mulvaney 1996). Additional mineral nutrients (calcium, magnesium, phosphorus, potassium, sodium, and sulfur) were measured with an inductively coupled plasma (ICP) spectrometer (SPECTRO/CIROS Model FTCEA000, Kleve, Germany) after extraction with a Mehlich 3 test solution (Mehlich 1984). Trace metals (copper, iron, manganese, and zinc) were extracted with diethylene triamine pentaacetic acid (DTPA) (Lindsay and Norvell 1978) and measured with an SPECTRO/ARCOS Model FHE12 ICP (Kleve, Germany). All nutrient samples were analyzed at field moistures, and soil moisture and bulk density correction factors were subsequently applied to present values on a dry volume basis (g cm^{-3}).

Soil oxidation-reduction potential (Eh) was measured *in situ* by inserting three bright platinum electrodes and a calomel reference electrode into the sediment of each plot to depths of 2 and 15 cm. Platinum electrodes were allowed to equilibrate for at least 30 minutes before reading the potential using a digital mV meter. The three readings at each depth were corrected for the reference electrode (+244 mV) and averaged. Eh was determined in the fall of 2011, 2012, and 2013.

Soil Accretion

Vertical accretion was measured at each station as the rate of sediment and organic matter accumulation above a feldspar marker horizon laid down within a 0.1-m^2 plot at each of the 35 sampling stations in October 2011. Vertical accretion was determined from two cores extracted at each station in fall of 2012 and 2013 using a 2.5-cm diameter Russian Peat Borer (Aquatic Research Instruments, Hope, ID). The depth to the feldspar horizon was measured at two separate locations around each core.

Plant Composition, Biomass, and Production

Aboveground biomass was estimated in 2011 by harvesting plants from a separate 0.25-m^2 clip plot. Clipped plant biomass was separated into live and dead categories, dried to a constant weight at 60^{o}C, and weighed. This same clip-plot was used to estimate net aboveground plant production at each station during the 2012-growing season by harvesting the biomass that re-grew into the 0.25-m^2 area after one year. At that time, a second clip-plot was established at each station, harvested, and utilized to estimate production during the 2013-growing season. In Fall 2013, a third clip-plot was established and clipped for a total of three aboveground biomass harvests and two annual estimates of aboveground production. Also, percent cover by species and mean canopy height were determined in 2011, 2012, and 2013 at each station within 1-m^2 quadrats. The species composition from both biomass and cover samples were used to determine total species richness, i.e., the total number of species for each of the seven treatments. Average species richness refers to the mean number of species for each treatment. Canopy height was measured in the field by visually aligning the canopy height with a graduated PVC pole and recording the height.

Belowground biomass was estimated in Fall 2011, Fall 2012, and Fall 2013 by collecting a 7.62-cm diameter x 30 cm long soil core from each station. The contents of each core were then sieved over 2-mm mesh, sorted into live and dead categories, dried at 60^{o}C, and weighed. Net belowground plant production was determined using in-growth bags (5-cm diameter by 30-cm long, 1.5 x 1.5-mm woven mesh bags, packed with peat ground to pass a 2-mm sieve) installed in duplicate at each station in October 2011 by removing soil cores of the same dimensions and replacing them with the in-growth bags. One in-growth bag was retrieved from each station in Fall 2012 by removing a slightly larger soil core that encapsulated the in-growth bags. Upon retrieval, the bags were rinsed of all mud, trimmed of external root/rhizomes, and washed over a 2-mm mesh screen to remove the peat packing material. The roots and rhizomes that remained were then dried to a constant weight at 60°C and weighed. Unfortunately, because of series of perturbations to the project area including hurricane damage and animal disturbance, as well as fire at the healthy reference marsh, in-growth bags were lost and belowground plant production could not be determined for 2013.

Plant Decomposition

Cellulose decomposition was determined in October 2011 using the cotton strip method (Harrison et al. 1988). This method, based on the decay of a standardized cotton fabric composed of 97% cellulose, evaluates decomposition by measuring loss of tensile strength of the cotton fibers making up the strips. At each station, 10 cm wide by 30 cm long strips of heavy artist canvas were inserted vertically into the soil substrate with the aid of a sharpshooter shovel and exposed to the soil environment for ~2 weeks, as described by Maltby (1987). Upon retrieval, additional control strips were inserted and removed immediately. Loss of tensile strength, calculated relative to control strips that were inserted into the ground and immediately removed, was then determined at 2-cm depth intervals down to a depth of 24 cm using a tensometer and force gauge (Dillon Quantrol SnapShot, Dillon, Fairmont, MN). This measurement was repeated at each station in Fall 2012 and 2013.

Above- and belowground plant decomposition were measured using 6-cm wide x 30-cm long litterbags constructed from 1-mm^2 nylon mesh and filled with either 7 g of oven-dried *S. patens* stems/leaves or 3 g of oven-dried *S. patens* roots/rhizomes. Litter bags were installed at each station in duplicate in October 2011. Bags containing aboveground material were placed on the marsh surface, while bags containing belowground material were inserted into the soil at a depth of 15 cm. One set of above- and belowground litterbags was retrieved in Fall 2012. Upon retrieval, the bags were washed of all mud, and any identifiable in-grown roots/rhizomes removed. The remaining material was then dried to a constant weight at 60°C and weighed. Unfortunately, because of series of perturbations to the project area including hurricane damage and animal disturbance, as well as fire at the healthy reference marsh, litter bags were lost and litter decomposition could not be determined for 2013.

Statistical Analyses

All statistical analyses were conducted using SAS (Statistical Analysis Systems) version 9.3 and JMP Pro 12.1 (products of SAS Institute, Inc., Cary, NC). We used univariate two-way mixed-model ANOVA with repeated measures to test the effects of each restoration treatment, year (repeated), and their interaction on above- and belowground biomass, total biomass, aboveground regrowth, plant canopy height, species richness, soil accretion, cotton tensile-strength loss per day, and soil physical properties (i.e., bulk density, moisture content, organic and mineral matter content, and percent sand, silt, and clay). Relationships between selected abiotic variables and total plant biomass were evaluated with regression analysis. Simple correlation coefficients (Pearson product-moment correlation coefficients) were determined and presented as needed to quantify specific relationships between variables. Single-year measurements of belowground ingrowth and above- and belowground litter decomposition were analyzed using one-way ANOVA to test the effects of each restoration treatment only. Correlated variables, including soil chemical properties and biomass-based composition of the dominant 10 plant species, were time-averaged and their dimensionalities reduced by factor analysis using the principal-axis method of extraction and squared multiple correlations of each variable with all other variables as prior communality estimates. Factors with eigenvalues greater than 1.0 were retained and orthogonally rotated with a varimax rotation (Kaiser, 1958). Variables with correlation coefficients ≥ 0.6 were used to define the retained factors. Factors scores generated for each sampling station were then analyzed using one-way ANOVA to test the effects of restoration treatment. For all ANOVA, differences among treatment-means were tested *post-hoc* using the Tukey-Kramer multiple comparison test. Unless otherwise noted, statistically significant differences were at $P \leq 0.05$.

For all statistical tests, assumptions of normality and homoscedasticity were verified by examining normal probability and residual plots, respectively. When necessary, these data were logarithmically, square root, or square transformed prior to analysis to validate model assumptions. For presentation of results, untransformed arithmetic means and standard errors (SE) were used.

RESULTS

Geospatial Assessments, Hydrology/Salinity, and Sediment Elevation

2008 Spatial Assessment

Successional changes across the Barataria Landbridge project area were quantified by examining four (2008, 2010, 2011, and 2012) base maps within a GIS framework. Based on the 2008 DOQQ aerials as a pre-construction starting point, surface features in the Barataria Landbridge marsh were typical of a rapidly subsiding and deteriorating deltaic marsh. Vegetation was generally weak and consisted of small fragmented communities of predominantly *Spartina patens* (marshhay cordgrass). Vegetation was generally shallow rooted in soft unconsolidated muck and often up-rooted in mass during storm events. Soil banks along major water bodies were severely eroded and had collapsed where vegetation had been under-cut by waves, tides, and storm surges. The ratio of water-to-vegetation was disproportionately high, and numerous shallow-water open ponds dominated large areas of the marsh.

Figure 6 is the 2008 pre-construction aerial, with the control boundary and treatment cell layers shown for emphasis. The total area included in the Barataria Landbridge study was 799.2 ha (1,975 ac), and included both Temple Island and the Little Lake Hunting Club marsh. In 2008, habitat types were limited and consisted of 498.5 ha (1,231.8 ac) of water (62.4%) and 300.7 ha (743.1 ac) of vegetation (37.6%), or slightly less than a 2:1 water-to-vegetation ratio. Vegetation was spatially disproportionate within the study area, with marsh areas adjacent to large water bodies containing significantly less vegetation than interior marsh areas located some distance away from open water bodies. For example, in the Little Lake Club marsh, treatment cells adjacent to Bayou Rigolettes (cells E-CT1, E-CT2, E-CT3, Figure 6) contained 73.4 ha (181.3 ac) of vegetation and 194.4 ha (480.3 ac) of water, with mean percentages of 27.4% and 72.6%, respectively, resulting in a water-to-vegetation ratio slightly greater than 2.5:1. The two interior treatment cells (E-UCT and Ctrl) contained 132.1 ha (326.3 ac) of vegetation (47.7%) and 144.5 ha (357.1 ac) of water (52.3%), or about 20% more vegetative cover than exterior cells and had a more balanced 1:1 water-to-vegetation ratio. The water-to-vegetation differential was similar within the Temple Island marsh. The two cells fronting Bayou Rigolettes (cells W-CT1 and W-CT2, Figure 6) contained a total of 39.5 ha (97.5 ac) of vegetation (31.3%) and 86.6 ha (213.9 ac) of water (68.7%), a slightly greater ratio of 2:1 water-to-vegetation. The more interior protected cell on Temple Island (cell W-UCT) contained almost the identical water-to-vegetation composition as the interior cells on Little Lake Club marsh. Specifically, the Temple Island interior cells consisted of 47.4% vegetation and 52.6% water and the Little Lake Club interior cells contained 47.3% vegetation and 52.7% water. The two-small protected open water cells on Temple Island, W-CUT1 and W-CUT2, consisted mostly of water with fringe vegetation along the banks. W-CUT1 contained 7.5 ha (18.5 ac) of water and 1.9 ha (4.6 ac) of vegetation and W-CUT2 contained 1.1 ha (2.7 ac) of water and 1.2 ha (2.9 ac) of vegetation.

Table 2 summarizes the 2008 spatial delineation for the entire project area by habitat and by cell. See Figure 6 for individual cell ID and the relative location of cells within the study area.

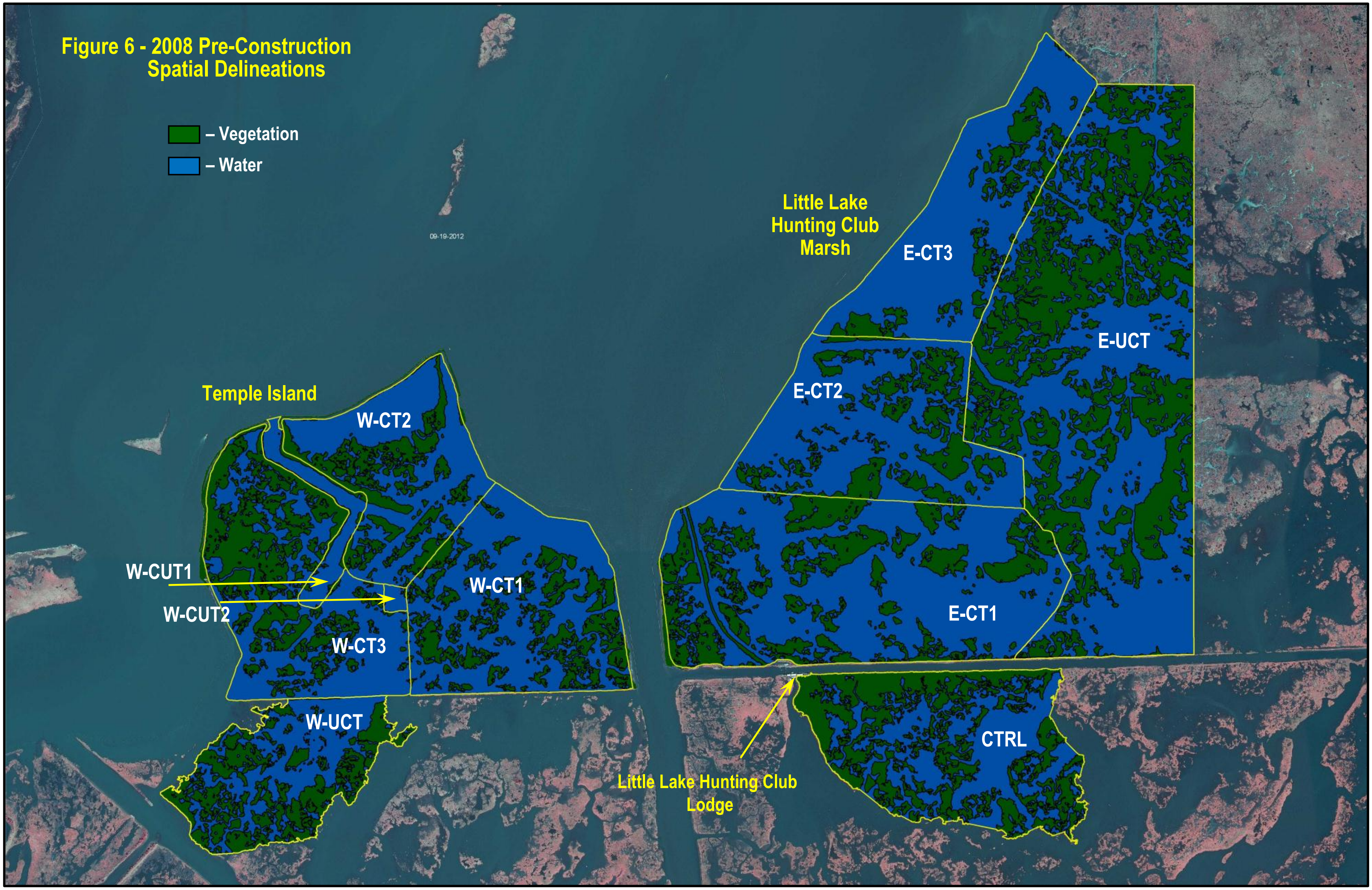


Figure 6 – The 2008 pre-construction marsh spatial delineations. Vegetation is heavily fragmented and retreating inward with large expanses of shallow open ponds dominating the study area. Measurements for spatial content by cells are included in **Table 2**.

Table 2 – 2008 pre-construction spatial delineation summary by cell. Prior to construction the study area was broken marsh and consisted of only two marsh habitats, large bodies of shallow water and small fragments of clumped vegetation.

2008				% of Total Cell	
Cell ID	**Vegetation (ha)**	**Water (ha)**	**Cell Subtotal (ha)**	**Vegetation**	**Water**
Ctrl	37.3	38.8	76.0	49.0%	51.0%
E-CT1	40.6	87.4	127.9	31.7%	68.3%
E-CT2	20.8	52.8	73.7	28.3%	71.7%
E-CT3	12.0	54.1	66.2	18.2%	81.8%
E-UCT	94.8	105.7	200.6	47.3%	52.7%
W-CT1	22.2	51.4	73.5	30.2%	69.8%
W-CT2	17.3	35.2	52.5	32.9%	67.1%
W-CT3	31.7	39.6	71.3	44.4%	55.6%
W-CUT1	1.7	7.5	9.2	18.7%	81.3%
W-CUT2	0.1	1.1	1.2	6.1%	93.9%
W-UCT	22.3	24.8	47.1	47.4%	52.6%
Total	**300.7**	**498.5**	**799.2**	**37.6%**	**62.4%**

2010 Spatial Assessment
In October 2010, a vegetative survey was completed to assay species composition and to associate infrared image signatures with treatment cell elements. All of the treatment cells on both Temple Island and the Little Lake Club were accessed from several entry points; however, access to the interior of both areas was limited and data were collected for only short distances from around the initial entry points. Based on observations of species frequency, soil moisture, and plant densities, plant community composition is generally driven by soil elevation, soil moisture, low salinity, and depth of sediment overburden over pre-construction standing flora.

In the Confined cells, for example E-CT1 and W-CT1, sediments were pumped to a higher, relatively uniform elevation, with soil moisture content varying from dry to standing water in surface depressions and borrow canals. The original flora (predominantly *Spartina patens* (marshhay cordgrass)) was completely buried by sediment overburden, and there was little evidence of *Spartina patens* regeneration. At the end of the first full growing season, vegetation within the Confined cells was predominantly annual species, which provided a fair measure of species richness, albeit seasonal. On dry to moist soils, *Echinochloa colona* (barnyardgrass), *Echinochloa walteri* (Walter's millet), *Leptochloa fascicularis* (bearded sprangletop), *Setaria pumila* (yellow bristlegrass), *Chenopodium ambrosioides* (Mexican tea), *Digitaria sanguinalis* (giant crabgrass), and *Rumex crispus* (curly dock) were common, though no one species was clearly dominant. Two twining legumes, *Vigna luteola* (deer pea) and to a lesser extent, *Centrosema virginianum* (coastal butterfly pea) were present. *Vicia ludoviciana* (Louisiana vetch), an annual legume more prevalent in the late spring to early summer, was present, although most plants were badly senesced. Woody semi-shrub species, such as *Croton capitatus* (wooly croton), *Sesbania drummondii* (rattlebox), *Sesbania herbacea* (sicklepod), *Aeschynomene indica* (jointvetch), *Ambrosia artemisiifolia* (common ragweed), *Ambrosia trifida* (giant ragweed), and *Symphyotrichum subulatum* (eastern annual saltmarsh aster), were scattered and co-mingled throughout the herbaceous community. In areas where the soil moisture ranged from wet-to-saturated, *Paspalum vaginatum*, (seashore Paspalum), *Panicum repens* (torpedo grass), *Suaeda linearis* (annual seepweed), and *Phragmites australis* (roseau cane) were common. On mudflats and in standing water (such as borrow canals) *Typha domingensis* (southern cattail), *Schoenoplectus americanus* (Olney threesquare), *Schoenoplectus californicus*, (California bullwhip), *Cyperus croceus* (Baldwin's flatsedge), and *Cyperus oxylepis* (sharpscale flatsedge) were common.

In the Unconfined treatment cells (E-UCT and W-UCT), where sediments were un-restricted, species composition varied along an elevation gradient from dry to wet and contained many of the same species found in the wetter areas of the contained cells. However, there was a notable presence of residual *Spartina patens* that was not found in the Confined cells. Most *Spartina patens* within the Unconfined cells appeared to be small clumps of vegetation or smaller plant fragments sheared off by free-flow sediments, and were randomly redistributed throughout the cell. A significant number of these fragments were in the early stages of re-colonizing. Two other plant species, *Sagittaria lancifolia* (bulltongue arrowhead) and *Distichlis spicata* (saltgrass), were also present, but in smaller quantity.

Vegetative composition in the Control cell (Ctrl) remained unchanged from its original pre-construction composition. The Control cell was dominated by *Spartina patens*, with tuffs of

Distichlis spicata and *Sagittaria lancifolia*. A few individual plants of *Lythrum lineare* (wand lythrum) were also found in the Control cell; however, they were sparse and limited to micro-sites where soil or wracks of organic matter had lodged, creating a slightly elevated environment above the mean water level.

2008-2010 Spatial Comparisons

Spatial changes in the first year post-construction were significant (Table 3). Prior to construction, the study area consisted of heavily fragmented marshes with 38% vegetation (300.7 ha; 743 ac) and 62% open water (498.6 ha; 1,232 ac). In the first year post-construction (2010), the same marsh area consisted of 67% vegetation (532.6 ha; 1,316 ac) and 33% open water (266.8 ha; 659.2 ac), for a vegetation-to-water ratio of 2:1. Overall, there was a 77.5% increase in vegetative cover and a 47.3% reduction in open water between 2008 and 2010.

Vegetative recovery in 2010 was relatively uniform within like treatment cells, but highest in the Confined cells, slightly lower in the Unconfined cells, and lowest in the Control cell. For example, within the six Confined cells (E-CT1, 2, 3 and W-CT1, 2, 3) the lowest vegetative cover value recorded was 70.3% in the East Confined treatment cell (E-CT1) and the highest vegetative cover of 86.5% was in the West Confined treatment cell (W-CT2). The mean vegetative cover across all Confined cells was 78.5%. Vegetative cover in the Unconfined cells (E-UCT and W-UCT) was slightly lower than the Confined cells. The mean vegetative cover value across both Unconfined treatment cells was 54.7%. The Control cell had the least amount of vegetative cover of all treatment cells with 39.5%. In addition, the Control cell was the only treatment cell within the study area to lose vegetative cover between 2008 and 2010.

Table 3 – 2010 post-construction spatial delineation summary by cell. Prior to construction, the study area was broken fragmented marsh, consisting of large bodies of shallow water and small clumps of vegetation. Habitat conditions changed significantly after the first full year with substantial vegetative growth and reduced number of open shallow water bodies.

	2010				% of Total Cell	
Treatment Cell	**Vegetation (ha)**	**% Change Veg from 2008**	**Water (ha)**	**% Change Water from 2008**	**Vegetation**	**Water**
CTRL	30.0	-19.5%	46.0	18.6%	39.5%	60.5%
E-CT1	89.9	121.7%	38.0	-56.5%	70.3%	29.7%
E-CT2	63.1	203.0%	10.6	-80.0%	85.7%	14.3%
E-CT3	49.3	310.5%	16.9	-68.9%	74.5%	25.5%
E-UCT	105.8	11.6%	94.8	-10.4%	52.7%	47.3%
W-CT1	57.7	160.0%	15.8	-69.2%	78.5%	21.5%
W-CT2	45.4	162.9%	7.1	-79.9%	86.5%	13.5%
W-CT3	59.8	88.9%	11.5	-71.1%	83.9%	16.1%
W-UCT	29.6	32.5%	17.5	-29.3%	62.8%	37.2%
W-CUT1	1.7	-2.2%	7.5	0.5%	18.3%	81.7%
W-CUT2	0.1	73.2%	1.1	-4.7%	10.5%	89.5%
Total	**532.5**	**77.5%**	**266.8**	**-47.3%**	**66.6%**	**33.4%**

Figures 7a and Figure 7b are side-by-side aerial image comparisons of vegetative cover within the Confined cells in 2008 (pre-construction) and in 2010 (first-year post-construction). A colored water overlay is included in both images for emphasis. Particularly noticeable is the change in individual plant community size and the number of discrete community units between 2008 and 2010. For example, in 2008 under pre-construction conditions, standing plant communities were smaller, fragmented, and generally migrating towards the interior. The three East Confined cells (E-CT1, 2, 3) contained a mean of 96 individual plant communities with a mean area of 0.24 ha (0.59 ac). Plant community size and structure within the three West Confined cells (W-CT1, 2, 3) were similar, with a mean of 101 individual plant communities and a mean area of 0.25 ha (0.61 ac). However, by 2010, plant communities within the East Confined cells had coalesced into larger units, reducing the number of individual plant units from 96 to 18, but increased in mean area from 0.24 ha (0.59 ac) to 3.6 ha (8.9 ac). Confined cells within the Temple Island marsh (W-CT1, 2, 3) followed the same growth patterns, with individual plant communities reduced from 101 in 2008 to 14 in 2010, with a mean area increase from 0.25 ha (0.61 ac) to 3.9 ha (9.6 ac).

The Unconfined cells also went through a significant change in community structure; however, moving in an opposite direction from the Confined cells. Recall that the plant community composition within the Unconfined cells were primarily *Spartina patens* (a perennial species) and consisted of small floating islands and plant fragments that were sheared away by moving sediments and randomly relocated across the Unconfined cells. Unlike the Confined cells, where in-situ plant materials were completely buried under sediment overburden, plant materials within the Unconfined cells were only partially buried and had begun a slow process of clonal regeneration and vegetative spread. Using the same methodologies in measuring individual plant communities within the Confined cells, we delineated 3,303 individual plant units within the East Unconfined treatment cell and 4,679 individual plant units within the West Unconfined treatment cell. By the end of the 2010 growing season, individual plant units within the Confined cells had grown together to form larger plant communities (mean area 3.6 ha; 8.9 ac), whereas plant units within the two Unconfined cells remained mostly fragmented, were individually small (mean area 0.016 ha; 0.04 ac), and had just begun to expand vegetatively. There were no unvegetated dry soil areas identified within any cells in 2010.

Figure 8 and Figure 9 are side-by-side aerial image comparisons of spatially delimited vegetative cover within the East Unconfined (Figure 8a-b) and the West Unconfined (Figure 9a-b) for 2008 (pre-construction) and 2010 (first-year post-construction). What appears as heavy black lines within the 2010 Unconfined cells in both the East- and West Unconfined cells are actually the outlines of thousands of small vegetative polygons blurred together when viewed at a small scale. When enlarged, the lines separate into several thousand small discrete vegetative units. Figure 9c-d are side-by-side aerial image comparisons of spatially delimited vegetative cover within the Control treatment cells. Of particular interest is the lack of any new vegetative growth between 2008 and 2010. Table 3 contains the 2010 spatial measurements for the nine treatment cells and includes the percent vegetative and water change between 2008 and 2010 within respective treatment cells. Figure 14, located at the end of this section, is a summary histogram comparing the vegetative compositions by treatment cell and by year.

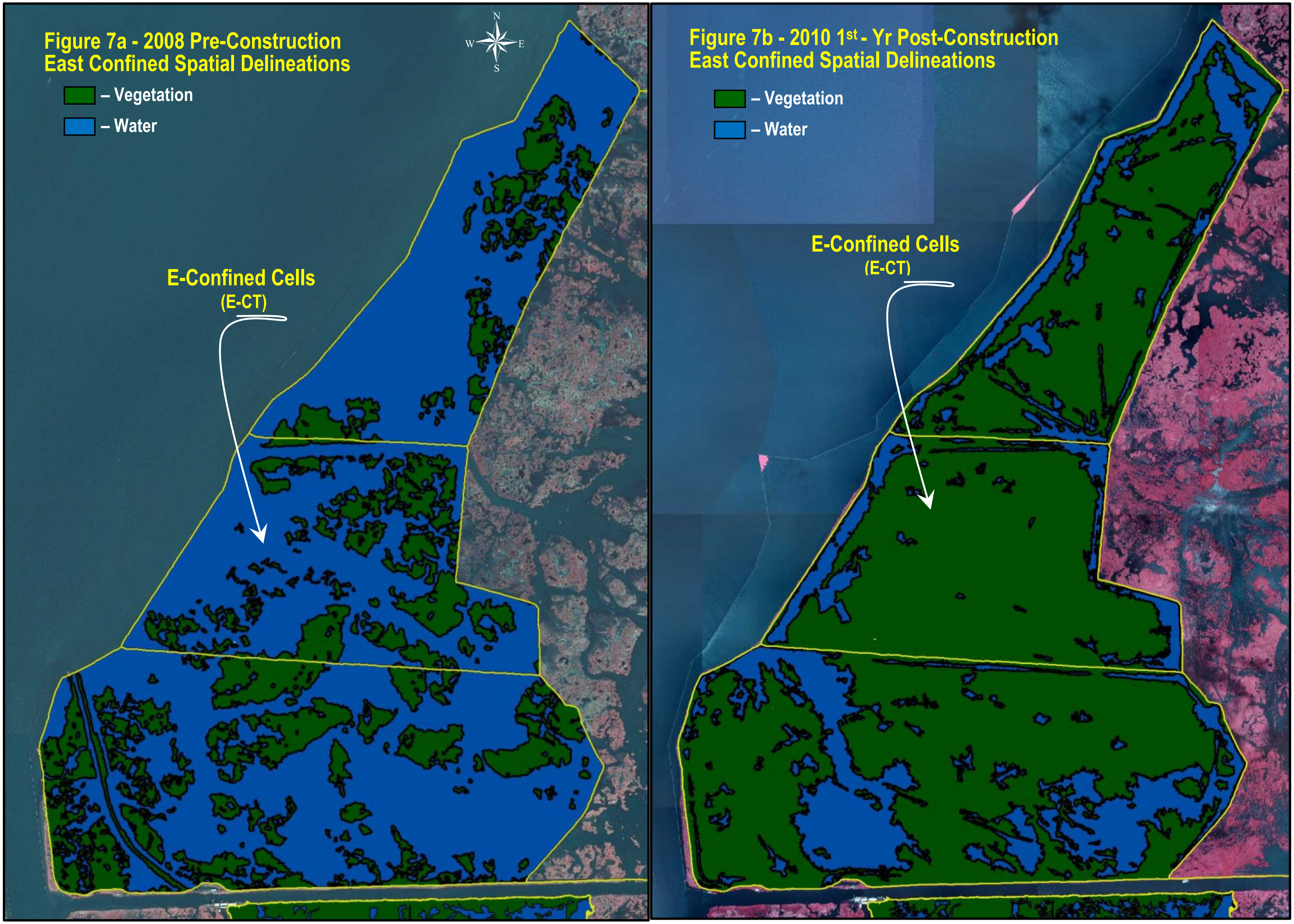


Figure 7a and **Figure 7b** – 2008 and 2010 East Confined treatment unit vegetative and water spatial components. Note the vegetative fragmentation and large bodies of open water in 2008, compared to the lack of fragmentation, increased vegetative communities, and reduced open water in 2010.

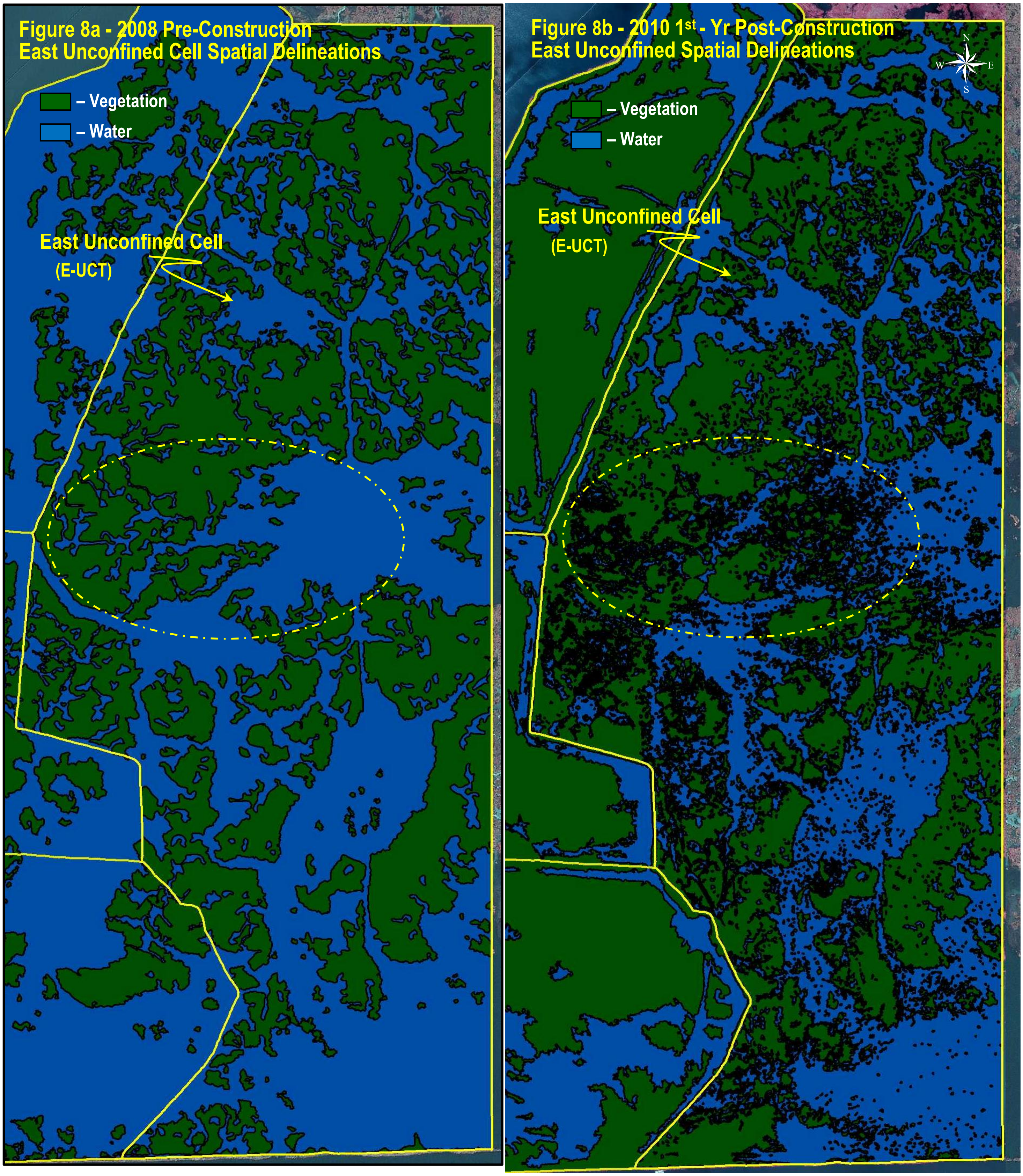


Figure 8a and **Figure 8b** – 2008 and 2010 East Unconfined treatment unit vegetative and water spatial components. Of primary interest are the heavy black matting seen in the 2010 aerial and not present in 2008. The black matting are the outer boundaries of several thousand individual vegetative polygons, most are small fragments of *Spartina patens* that sheared off during sediment loading and are in their initial stage of regenerating. Note the same matting patterns in the 2010 West Unconfined treatment cell, Figure 9b.

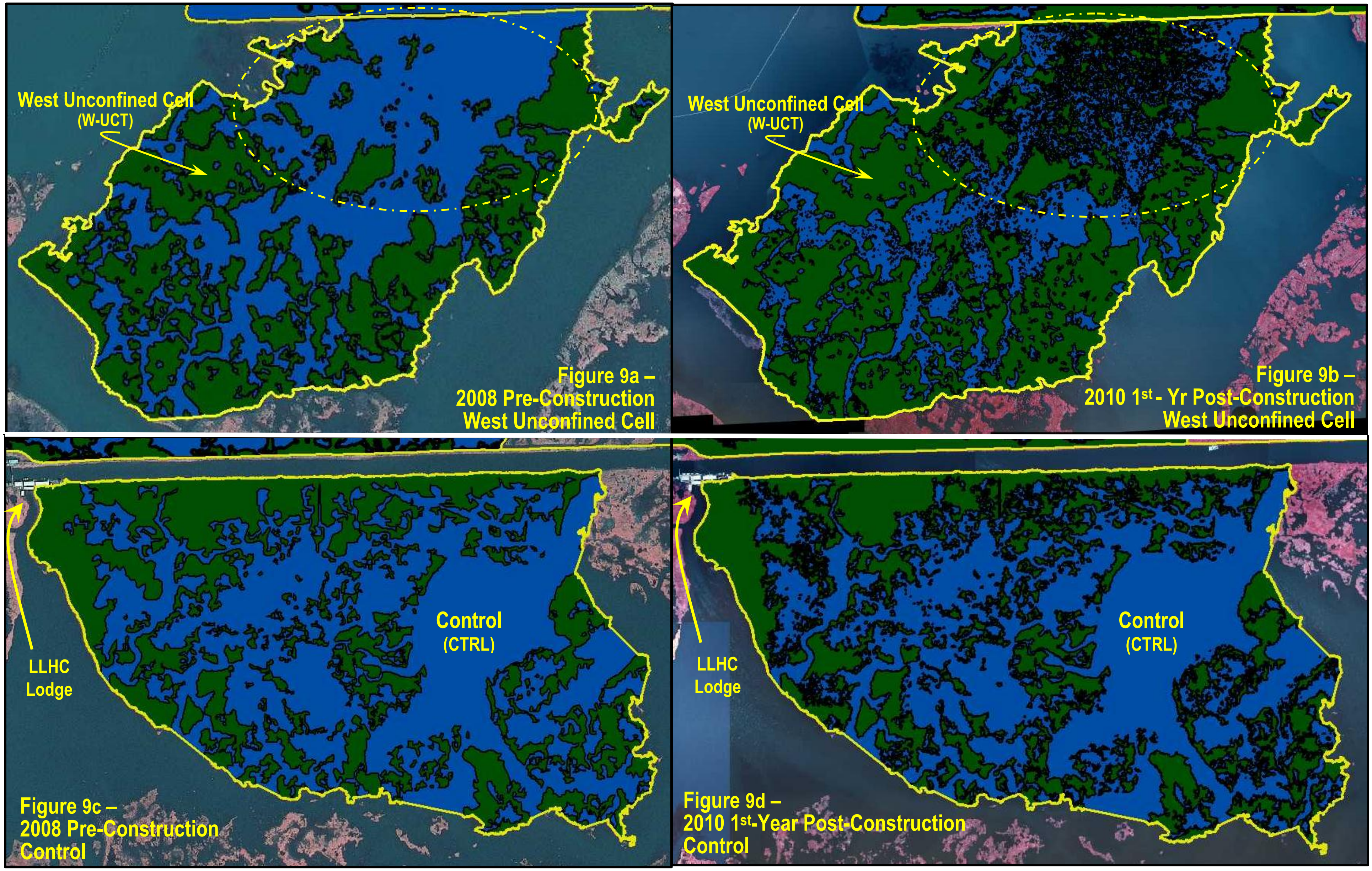


Figure 9a and **Figure 9b** – 2008 and 2010 West Unconfined treatment unit vegetative and water spatial components. Of primary interest are the heavy black matting seen in the 2010 aerial and not present in 2008. Regeneration of several thousand small fragments of *Spartina patens* were emerging at about the same rate and extent as that seen in the East Unconfined cell (Fig. 8b above). **Figure 9c** and **Figure 9d** – are the 2008 and 2010 Control cell spatial delineations. There was little change within the Control cell, either within the vegetative or the water components between 2008 and 2010. The Control treatment unit was the only treatment cell to actually lose vegetative cover between 2008 and 2010.

2011 Spatial Assessment

Spatial changes in the first year (2010) post-construction were dramatic (Table 4). With the addition of dredge-sediments, large areas of previously unproductive open shallow water marsh were converted to equally large areas of diverse vegetated marsh. The rate and extent of vegetative reclamation varied within like-treatment cells, but were considerably different between treatments cells.

In 2011, the second year post-construction, there were minor shifts within habitat as surface sediments continued to de-water and become drier. In addition, an Un-Vegetative mapping feature was added in 2011 to capture bare dry soils lacking any discernable vegetative cover. In 2010, there were no dry soil areas mapped within the project area, but by the end of the 2011 growing season, 19.1% or 150.7 ha (372.5 ac) of the project area was mapped as Unvegetated dry soils. Although all of the treatment cells contained some quantity of unvegetated areas, the six Confined treatment cells (E- and W-CT) contained the greatest amount of Unvegetated dry soils, with a 28.1% mean and a total of 130.9 ha (323.4 ac). The East- and West-Unconfined treatment cells contained smaller quantities of unvegetated dry soils with a mean of 7.4% within cells and a total of 18.4 ha (45.5 ac) across cells. The single Control cell contained the least amount of unvegetated bare soils with 1.9% or 1.4 ha (3.5 ac).

Initially it was thought that unvegetated soils were the result of receding surface water and/or sediment dewatering; however, we found a significant amount of lost habitat within areas that were previously mapped as vegetated, as well as areas that were previously mapped as surface water in 2010. For example in 2011, all treatment cells lost some surface water area except the Control cell, which actually added an additional 5.3% or 2.4 ha (6.0 ac). Mean water loss within the six Confined cells was 72.7% or 72.6 ha (179.4 ac) across all six cells. Water loss within the two Unconfined cells was the greatest, with a cell mean of 91.1% or 102.3 ha (252.8 ac) across both cells. In addition, all treatment cells except the two Unconfined treatment cells had a net loss of vegetative cover. The mean vegetative loss within the six Confined cells was 15.9% or 58.2 ha (143.9 ac) across all cells, and the Control cell lost 12.9% or 3.9 ha (9.6 ac) of its vegetative cover. However, the Unconfined cells had a mean gain in vegetative cover of 62.0% within the two cells or an increase of 83.9 ha (207.3 ac) across both cells.

Treatment effects, specifically within the Confined and Unconfined sediment cells, were of particular interest to the project sponsors. Confined cells represent a more typical design in dredge sediment engineering when creating new marsh, while Unconfined cells are more commonly associated with marsh enhancements. Building new marsh using Confined cells is generally expensive and involves building containment levees, where large quantities of dredge-sediments are loaded to a uniform elevation. All of, or at least a portion of, the newly created marsh are artificially vegetated to control sediment loss, protect the containment levees, and to provide a select source of plant materials for plant community development. Unconfined cells are comparably less expensive to construct, require minimum levee construction, and use smaller quantities of dredge sediment. Sediments within Unconfined cells move through the cell unit as sheet-flow, forming low elevation soil gradients, rather than compacted sediments that are stacked to a uniform elevation. In Unconfined cells, coarse particle sediments mound nearest the sediment source and the smaller particles form thin veneer layers at the farthest ends of the soil gradient. There is an added-value in free flow sediments in that ultra-fine soil particles, such as clays and low density organic matter, stay in suspension longer and form nutrient fronts that potentially benefit areas beyond the primary sediment field. Consequently, it was particularly interesting to see the considerable vegetative difference between the Confined and Unconfined treatment cells following the 2011 growing season.

Recall that in 2010, the Confined cells were higher in elevation, seasonally dryer, heavily populated with annuals and had an average 78.5% vegetative cover. Within the same time-period, the Unconfined cells were lower in elevation, seasonally wetter, populated with fragmented perennial plant materials with only about half (mean 54.7%) of the cell covered with vegetation; neither treatment area was artificially planted. However, by the end of the 2011 growing season, the Confined cells had lost 15.9% (58.1 ha; 143.9 ac) of its 2010 vegetative cover, while the Unconfined cells had a net 33.9% (83.9 ha; 207.3 acres) increase in vegetative cover between 2010 and 2011. Vegetative cover within the Confined cells was heavily dependent on recurring seed germination from annual species, while plant species within the Unconfined cells were primarily small fragment of *Spartina patens* (a perennial). After a one-year growth lag, the Unconfined cells had 22.5% (55.8 ha; 137.9 ac) greater vegetative cover than that found within the Confined cells.

Table 4 contains the 2011 spatial measurements for the nine treatment cells and the percent of change between 2010 and 2011. Figure 14, located at the end of this section, is a summary histogram comparing the vegetative compositions by treatment cell and by year. Figure 10 and Figure 11 are side-by-side aerial image comparisons of spatially delimited vegetation, bare soil, and water areas for the East Confined (Figure 10a-b) and the East Unconfined (Figure 11a-b) treatment cells; Figure 12a-d are 2010 and 2011 side-by-side image comparisons for the West Unconfined and Control treatment unit spatial changes. Of particular interest is the amount and location of the newly appearing unvegetated dry soil areas. Although all the treatment units contain some dry soils, the largest areas appeared in the higher elevated East-Confined unit.

Table 4 - 2011 Spatial delineation summary by treatment cell with the percent change from the previous study year and percent totals.

2011						% of Total Cell		
Treatment Cell	**Veg (ha)**	**% Change Veg from 2010**	**Water (ha)**	**% Change Water from 2010**	**Un-Veg (ha)**	**Veg**	**Water**	**Un-Veg**
CTRL	26.1	-12.9%	48.4	5.3%	1.4	34.4%	63.7%	1.9%
E-CT1	66.6	-25.9%	9.5	-74.9%	51.8	52.1%	7.4%	40.5%
E-CT2	60.9	-3.4%	3.9	-63.0%	8.8	82.7%	5.3%	12.0%
E-CT3	48.3	-2.0%	7.1	-58.0%	10.8	73.0%	10.7%	16.3%
E-UCT	176.4	66.8%	7.8	-91.8%	16.3	88.0%	3.9%	8.2%
W-CT1	52.0	-10.0%	4.0	-74.9%	17.6	70.6%	5.4%	24.0%
W-CT2	28.5	-37.2%	2.1	-69.5%	21.8	54.3%	4.1%	41.6%
W-CT3	50.7	-15.3%	0.6	-94.6%	20.0	71.1%	0.9%	28.1%
W-UCT	42.9	44.7%	2.2	-87.3%	2.0	90.9%	4.7%	4.3%
W-CUT1	1.5	-14.0%	7.8	3.1%	0.0	15.7%	84.3%	0.0
W-CUT2	0.2	51.8%	1.0	-6.1%	0.0	16.0%	84.0%	0.0
Total	**554.1**	**4.1%**	**94.5**	**-64.6%**	**150.7**	**69.3%**	**11.8%**	**18.9%**

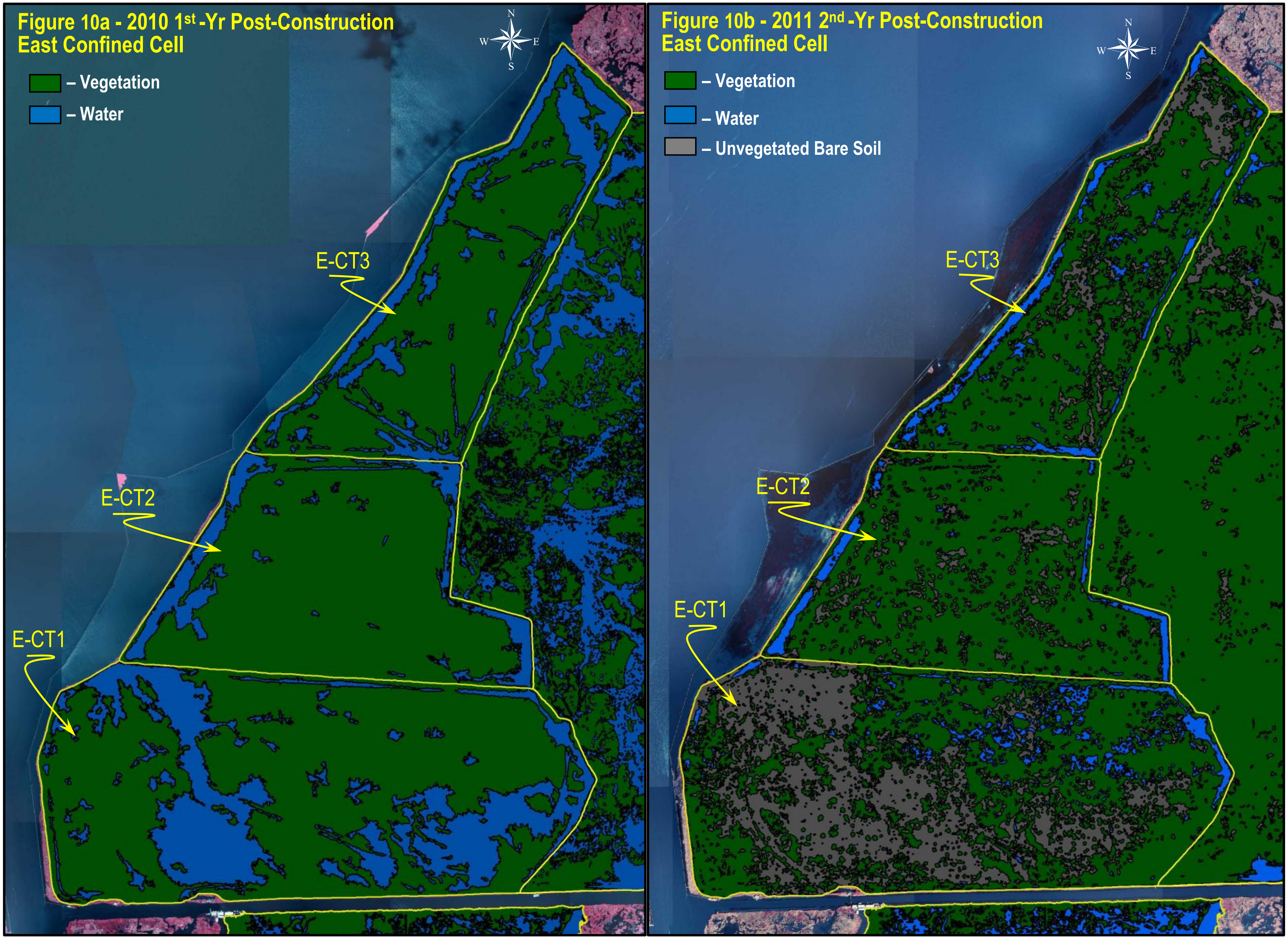


Figure 10a and **Figure 10b** – 2010 and 2011 vegetative and water spatial components for the East Confined Treatment (E-CT 1-3) cells. Unvegetated bare soils first appeared in 2011, mostly in the slightly lower elevated sediment flats that were previously (2010) water. Unvegetated soils were particularly noticeable in the E-CT1 cell in 2011.

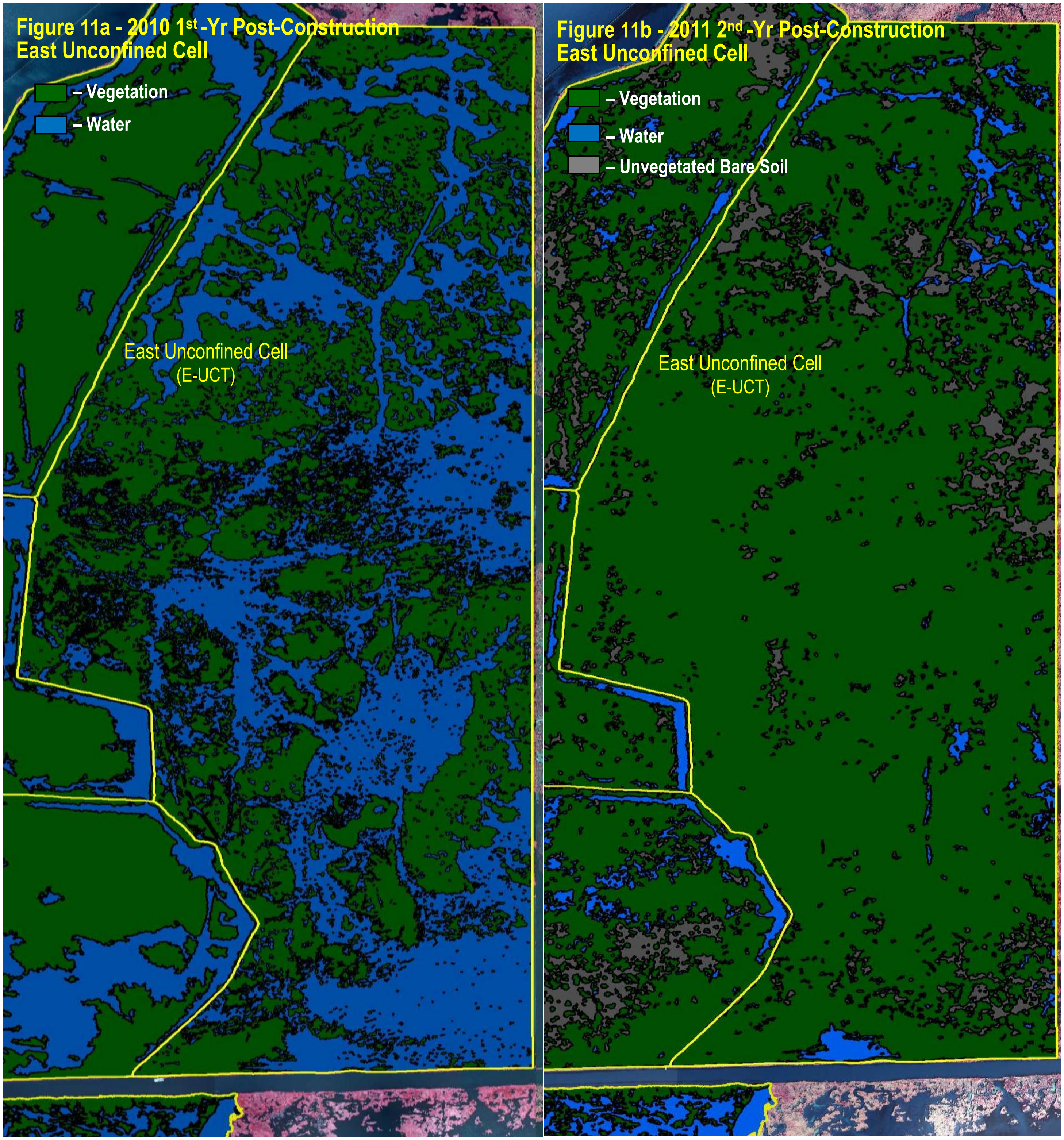


Figure 11a and **Figure 11b** – 2010 and 2011 vegetative and water spatial components for the East Unconfined Treatment cell. Although there are a few small areas of Unvegetated bare soils mapped in the Unconfined unit in 2011, the most notable feature is the survival and maturation of thousands of small plant fragments seen as black matting in 2010. In 2010 there were 3,303 vegetative polygons with a mean area of 0.032 ha (0.079 ac) in the East Unconfined cell. By 2011, the 3,303 vegetative polygons had coalesced into 251 polygons with a mean area of 0.704 ha (1.74 ac). There were no Unvegetated areas in 2010, but by 2011 there were 776 polygons with a mean area of 0.021 ha (0.052 ac) .

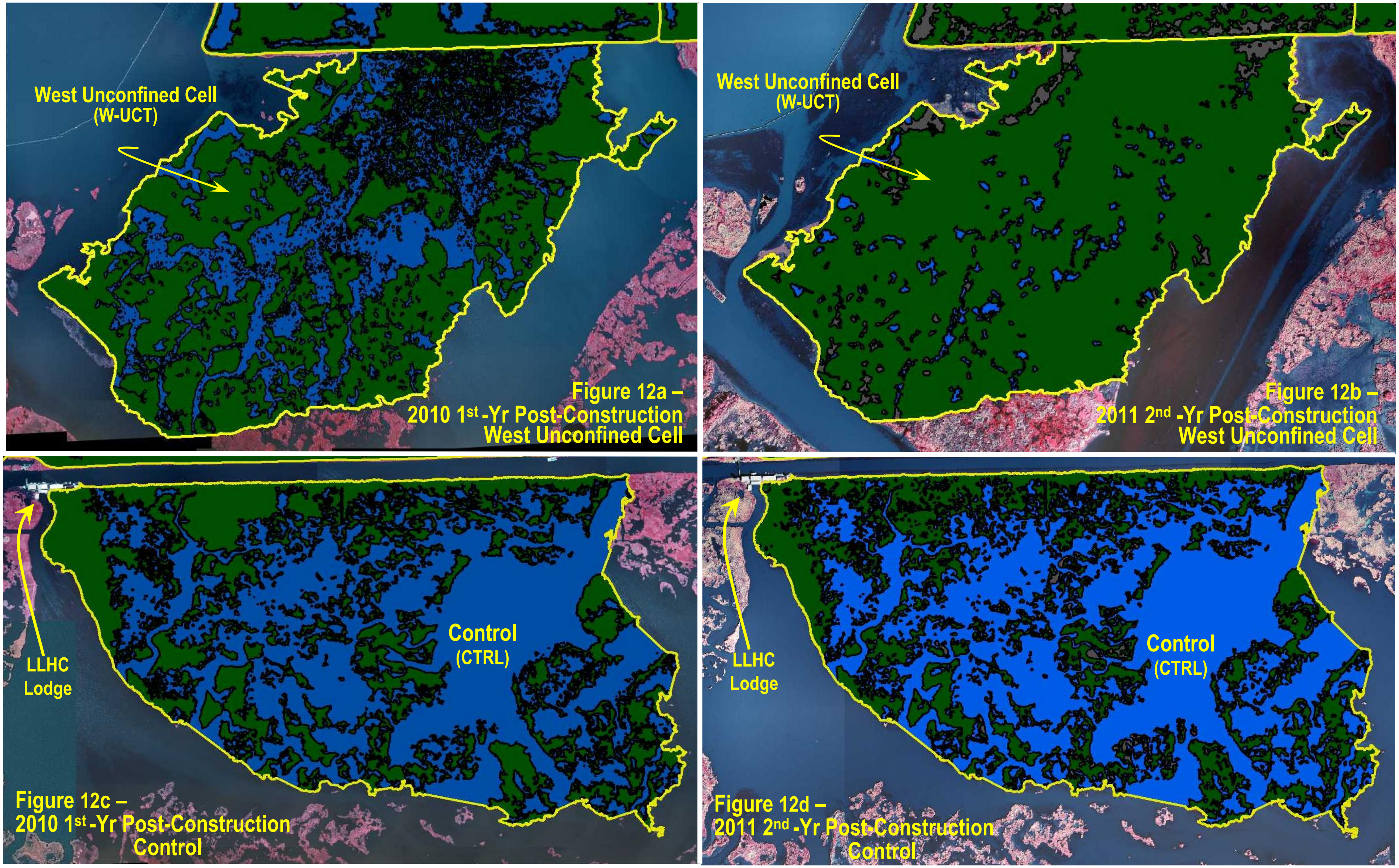


Figure 12a and **Figure 12b** – 2010 and 2011 vegetative and water spatial components for the West Unconfined Treatment cell. Although there were a few small areas of Unvegetated bare soils mapped in the Unconfined unit in 2011, the most notable feature was the survival and maturation of the thousands of small plant fragments seen in 2010. In 2010 there were 4,679 vegetative polygons with a mean area of 0.006 ha (0.015 ac) and by 2011 the vegetative polygons had coalesced into 37 vegetative polygons with a mean area of 1.157 ha (2.86 ac). There were no Unvegetated areas in 2010, but by 2011 there were 185 polygons with a mean area of 0.015 ha (0.039 ac). **Figure 12c** and **Figure 12d** – are the 2010 and 2011 Control cell aerials. There was little change within the Control cell, either within the vegetative or the water components.

2012 Spatial Assessments

In August 2012, Hurricane Isaac made landfall slightly west of Port Fourchon in Lafourche Parish, Louisiana. Hurricane Isaac was a modest category 1 hurricane, with wind gusts of 85 mph measured at Grand Isle, Louisiana and with a tidal surge of 11 feet measured at Shell Beach, Louisiana. Hurricane Isaac was a slow moving storm that produced a significant amount of rain, resulting in record damages to home and property for a large part of southeast Louisiana. Damages to the Barataria Landbridge marsh were primarily flood-related as well, and there was no evidence of surface scouring or gully erosion within the study area. The aerial image in Figure 13 was taken September 19, 2012, approximately five weeks post-Hurricane Isaac. Impacted vegetation appear on the color-infrared image as circular white-grey features, and areas of concentrated wrack (same color signature) appear as linear patterns along shorelines, within narrow channels, and along most embankments.

In October 2012, a two-day field trip was conducted to correlate aerial infrared signatures to their respective ground value. The standing vegetation and wrack impacted areas were of particular interest, as it was unclear from the imagery whether impacted areas were standing dead and should be included as a separate additional mapping feature or just damaged and would potentially return as live vegetative cover in the next growing season. After a two-day field assessment, it was determined that within the standing perennial plant communities, damage was limited to the above-ground portion of the plants and was primarily mechanical damage; that is, stems were either broken off somewhere above the crown or were still attached to the crown and lodged over. Damage to the below-ground portion of the plant was minimal, and for most samples non-existent; that is, root and rhizomes were firm, and they were of applicable color and intact. There was no odor in the immediate root zone to suggest any deteriorating tissue. Wrack damaged plants in many cases were simply covered with detrital material and were yellow or light brown from lack of sunlight. Many had no mechanical damage, but were simply lodged

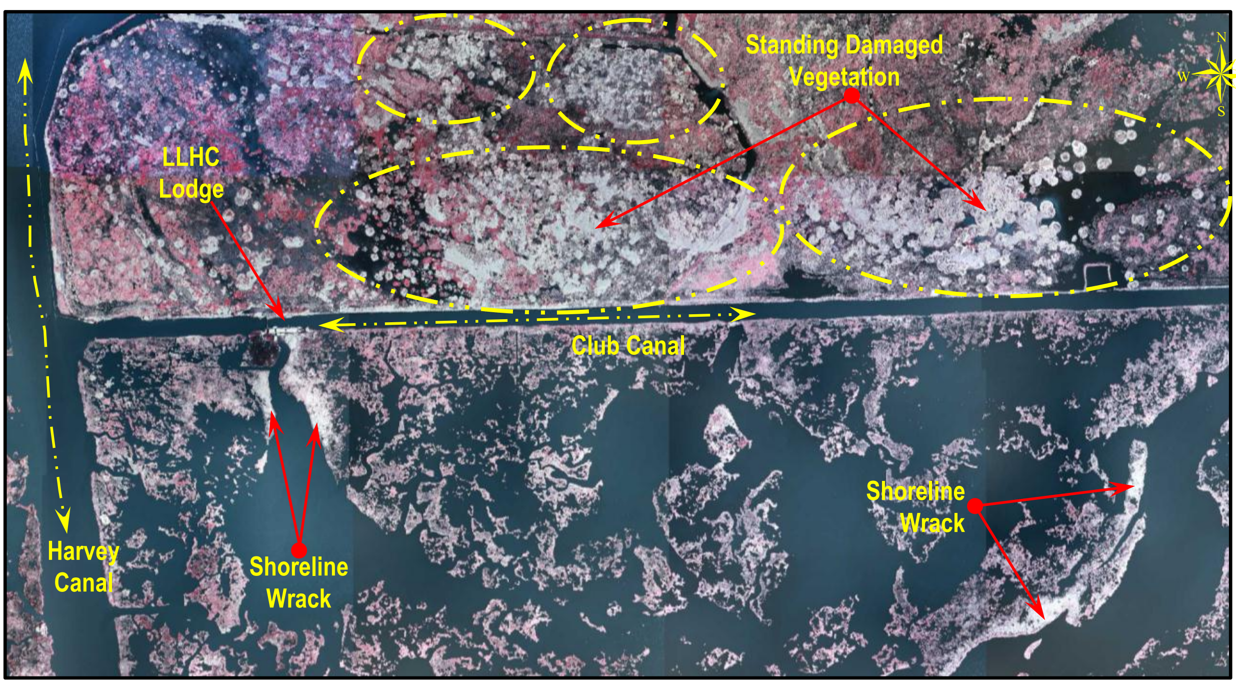


Figure 13 – 2012 aerial image taken five-weeks post Hurricane Isaac. The light whitish-silver signatures are areas of flood impacted vegetation and areas of concentrated wrack debris. Based on a two-day site survey, damaged vegetation was yellow to brown, but remained standing and living, and mortality was minimal.

over and smothered under overburden. Root cores from the wrack areas were virtually the same as from the standing damaged areas; that is, no appreciable root-rhizome mortality was evident. Consequently, the standing and wrack-damaged areas were mapped as live vegetative cover.

At the end of the 2012 growing season and post-Hurricane Isaac, the Barataria Landbridge project area consisted of 64.0% vegetative cover (511.4 ha; 1,263.6 ac), 24.9% unvegetated bare soils (198.7 ha; 490.9 ac), and 11.2% open water marsh (89.4 ha; 220.9 ac) (Table 5). The 2012 measurements in general, showed a slight decline in vegetative cover across all of the project area from 2011, except for two Confined cells on Temple Island, W-CT1 and W-CT3; these two cells had a mean increase of 14.8% in vegetative cover. Although there were vegetative losses in seven of nine treatment cells from 2011, losses were relatively small, with a high of 18.3% in the West Unconfined cell and a low of 2.6% in W-CT2. There were, however, significant increases in unvegetated bare soils in 2012, ranging from 411.2% in the West Unconfined unit to 1.8% in cell W-CT2. The only two cells with no increased area of unvegetated soils were cells W-CT1 and W-CT3. These two cells were also the only cells that contained area increases in both Vegetative Cover and Surface Water. The vegetative component in the Control cell continued to decline, as it had in the previous three years. Table 5 contains the 2012 spatial measurements for the nine treatment cells and includes the percent change within cells between project years 2011 and 2012. Figure 14 is a histogram comparing vegetative component for 2008 to 2012, by treatment cell and by treatment year. Figure 15 and Figure 16 are the 2011 and 2012 project areas, respectively. The two aerial figures include individual treatment cell boundaries and within cell habitat delineations. The two aerials were taken approximately the same time in their respective years and provide a sense of relationship, proportion, and spatial distribution within and between treatment cells, and within the project area as a unit.

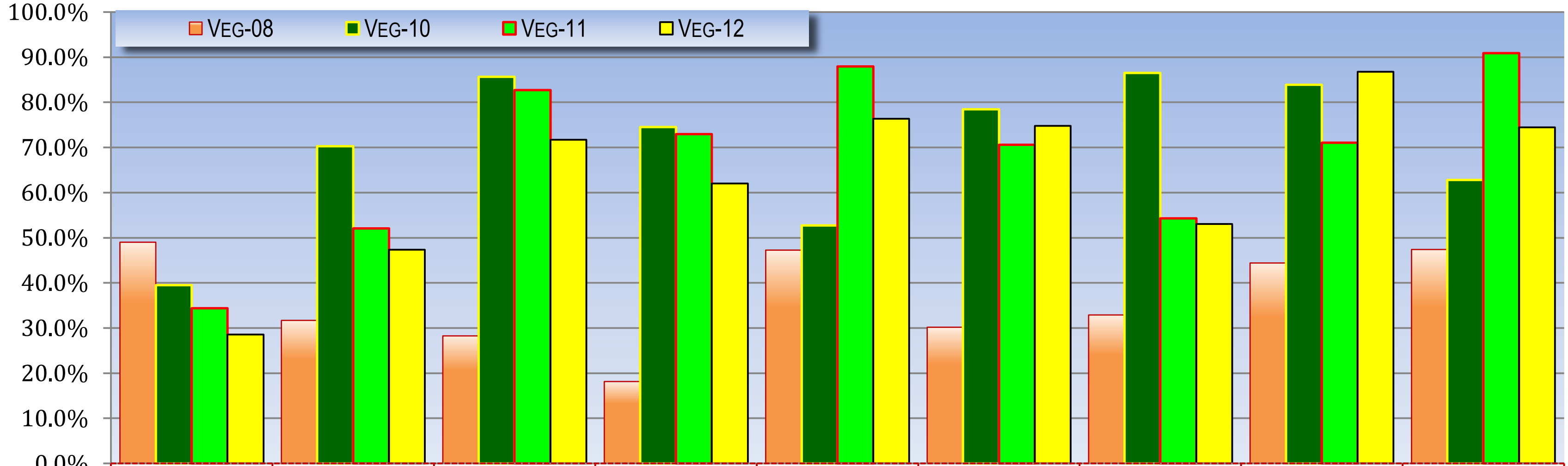
Marsh conditions prior to implementation of the project were deleterious. The Barataria Landbridge marsh in 2008 was dominated by shallow and unproductive open water with a weak and declining vegetative community. In the fall of 2008, the Barataria Landbridge marsh consisted of 498.5 ha (1,231.8 ac; 62.4%) open water and 300.7 ha (743.1 ac; 37.6%) vegetative cover. In the fall of 2012, the last year of the project study, the same marsh area consisted of 511.4 ha (1,263.6 ac; 64.0%) of vegetative cover, 89.4 ha (220.9 ac; 11.2%) open water, and 198.7 ha (491.0 ac; 24.9%) elevated, but unvegetated soils. From 2008 to 2012, the Landbridge marsh increased in vegetative cover by 71%, decreased surface water areas by 81.8%, and added 198.7 ha (491.0 ac; 24.9%) of elevated unvegetated soils. In comparison, the un-treated Control marsh lost 41.8% of its 2008 vegetative cover, increased its open water area by 29.9%, and added 3.8 ha (9.5 ac; 5.1%) of elevated, but unvegetated soils. Table 6 compares treatment and Control marsh conditions and subsequent changes for the first year prior to the Landbridge sediment treatment project, the first year post-treatment, and the final year of the project study.

Table 5 – 2012 Spatial delineation summary by treatment cell with percent change from the previous study year.

2012									
							% of Total Cell		
Treatment Cell	Veg (ha)	% Veg Change from 2011	Water (ha)	% Water Change from 2011	Un-Veg (ha)	% Un-Veg Change from 2011	Veg	Water	Un-Veg
CTRL	21.7	-17.1%	50.3	3.9%	3.8	168.7%	28.6%	66.4%	5.1%
E-CT1	60.5	-9.2%	2.1	-78.4%	65.2	25.9%	47.3%	1.6%	51.0%
E-CT2	52.7	-13.5%	3.7	-5.2%	17.1	93.7%	71.7%	5.0%	23.3%
E-CT3	41.0	-15.2%	6.5	-14.4%	19.0	75.9%	62.0%	9.2%	28.8%
E-UCT	153.1	-13.3%	3.5	-54.6%	43.9	168.0%	76.4%	1.8%	21.9%
W-CT1	54.9	5.6%	3.3	-17.5%	15.2	-13.5%	74.8%	4.5%	20.8%
W-CT2	27.8	-2.6%	2.3	8.8%	22.2	1.8%	53.1%	4.5%	42.4%
W-CT3	62.9	24.2%	8.2	1236.7%	1.4	-93.0%	86.7%	11.3%	1.9%
W-UCT	35.0	-18.3%	1.6	-29.9%	10.4	411.2%	74.5%	3.3%	22.2%
W-CUT1	1.6	0	7.5	0	0.3	0	16.8%	79.8%	3.4%
W-CUT2	0.3	0	0.8	0	0.0	0	27.7%	68.6%	3.6%
Total	**511.4**	**-7.4%**	**89.4**	**4.3%**	**198.7**	**31.8%**	**64.0%**	**11.2%**	**24.9%**

Table 6 – Compares Sediment Treatment cells and Control marsh conditions prior to dredge sediment treatment (pre-construction 2008), first year post-sediment treatment (2010), and final year (3rd year post-sediment treatment (2012)).

Treatment Cells	Veg (ha)	Veg (%)	Water (ha)	Water (%)	Un-Veg (ha)	Un-Veg (%)
Year						
2008- 1st year pre-project	263.4	36.4%	459.7	63.6%	0	0
2010- 1st year post-project	502.5	69.5%	220.8	30.5%	0	0
2012- Last year post-project	489.7	67.7%	39.1	5.4%	194.9	26.9%
+/- Change from 2008 to 2010	239.1	90.8%	-238.9	-52.0%	0	
+/- Change from 2008 to 2012	226.3	85.9%	-420.6	-91.5%	194.9	100%
Control						
Year						
2008- 1st year pre-project	37.3	49.0%	38.8	51.0%	0	0
2010- 1st year post-project	30.0	39.5%	46.0	60.5%	0	0
2012- Last year post-project	21.7	28.6%	50.3	66.3%	3.8	5.1%
+/- Change from 2008 to 2010	-7.3	-19.6%	7.2	18.6%	0	0
+/- Change from 2008 to 2012	-15.6	-41.8%	11.5	29.6%	3.8	100%



	Ctrl	E-CT1	E-CT2	E-CT3	E-UCT	W-CT1	W-CT2	W-CT3	W-UCT
Veg-08	49.0%	31.7%	28.3%	18.2%	47.3%	30.2%	32.9%	44.4%	47.4%
Veg-10	39.5%	70.3%	85.7%	74.5%	52.7%	78.5%	86.5%	83.9%	62.8%
Veg-11	34.4%	52.1%	82.7%	73.0%	88.0%	70.6%	54.3%	71.1%	90.9%
Veg-12	28.6%	47.3%	71.7%	62.0%	76.4%	74.8%	53.1%	86.7%	74.5%

Figure 14 – Vegetative comparison within treatment cells by year. 2008 represents pre-construction vegetative conditions in a common deteriorating deltaic marsh, 2010 represents the first full growing season post-construction, and 2012 is the third full growing season post-construction and following a modest category 1 hurricane (Isaac, 08/12).

Figure 15 – The 2011 Barataria Landbridge study area marsh. Figure 15 includes individual treatment cell boundaries and within cell habitat delineations. Compare the 2011 (Fig. 15) aerial to the 2012 (Fig. 16) aerial of the same area with habitat polygons. The two aerial images were taken approximately the same time period within their respective years and shows relationship, proportion, and spatial distribution within and between treatment cells and the Barataria Landbridge marsh as a unit.

Figure 16 – The 2012 Barataria Landbridge study area marsh. Figure 16 includes individual treatment cell boundaries and within cell habitat delineations. Compare the 2012 (Fig. 16) aerial to the 2011 (Fig. 15) aerial of the same area with habitat polygons. The two aerial images were taken approximately the same time period within their respective years and shows relationship, proportion, and spatial distribution within and between treatment cells and the Barataria Landbridge marsh as a unit. The 2012 aerial image was taken approximately two-months post Hurricane Isaac (08/12).

Hydrology Assessments

Objectives of the Hydrology, Salinity, and Sediment Surface Elevation tasks were to assess the long-term hydrologic/salinity relationships within the three treatment areas and to assay sediment treatment and sample site elevation relationships. To determine water elevations, three YSI model 600LS water sondes, equipped with multi-sensors and a vented level system were installed to record changes in both water level and salinity values within three study treatment areas. Water temperature (°C), specific conductance (μS/cm), salinity (ppt), and water levels (m) were measured on a continuous basis (half-hour intervals) beginning on July 14, 2011 and ending on July 2, 2013. Water data were collected at three points within the study area (Figure 3): (1) in the Little Lake Camp Canal, immediately adjacent to the Control marsh and outside of any sediment treatment areas (N29.56804; W090.11900); (2) within the Confined sediment treatment area (N29.57488; W090.13335); and (3) within the Unconfined free-flow treatment area (N29.56830; W090.11900). Once established, a differential elevation survey between the three sondes was completed to determine a correction coefficient necessary to equilibrate water level readings across the three treatment units. On December 3, 2012, as part of the BA-125 Northwest Turtle Bay Marsh Creation Project survey, T. Baker Smith, LLC established a NAVD88 Geoid09 elevation to the Control marsh sonde and to a secondary temporary benchmark (TBM) associated with the East Confined treatment cells. The sonde and the TBM elevations, as well as all subsequent water level data, were corrected to NAVD88 Geoid12A and converted to meters for consistence.

When we compared mean water levels across the three treatments units, we found no significant difference ($p>0.05$) between the Control and the East Confined treatment cells. However, we did find a significant difference ($p= <0.0001$) in mean water level between the East Unconfined, the Control, and the East Confined treatment area. Mean water levels (n=34,502) for both the Control marsh and the East Confined area were 0.036 m (SE=0.001) and 0.040 m (SE=0.001), respectively; mean water for the East Unconfined cell was 0.162 m (SE=0.001). For the 24-month monitoring period, the mean water level within the East Unconfined area was 0.126 m higher, or about 350% greater than either the Control marsh or the East Confined cell. In addition, we found that water exchange between the East Confined cell and Control marsh was moderately restricted by the remaining low retention levees and that there was about a two-hour exchange lag between the two systems. Based on the sonde dataset, we found a consistent synchronous ebb and flow pattern between the Control marsh and the East Confined cell with only a slight mean elevation differential (0.004 m, 0.013 ft) between the two treatments. We also found that water levels within the Control and East Confined marshes were primarily driven by rising and falling tides and were not appreciably influenced by low to moderate rainfall events. However, the East Unconfined sediment cell is an impounded marsh, formed by a series of protection levees and five fixed crest weirs, and differs from the Control and the East Confined cell in both mean water level and in tidal influence. Hydrology within the East Unconfined treatment unit is primarily dependent on overflow from high water events initiating from the Little Lake Club Canal and from captured water collected from rainfall events.

Although the elevations for the fixed crest weirs were not available for this study, we determined that weir crests were set at approximately 0.11 m (0.36 ft) above NAVD88. This places the spill elevation at 0.07 m (0.23 ft) above the Control marsh mean water, and establishes the threshold elevation for water exchange between the Little Lake Club Canal (Control hydrology) and the

East Unconfined impounded marsh. A 0.11 m (0.36 ft) weir elevation is high enough to maintain a shallow water pool within the East Unconfined marsh during dry periods, but not so high as to functionally eliminate water exchange between the outside Control marsh and the East Unconfined impounded marsh. Water levels within the Control marsh exceeded the fixed crest weirs (creating a flooding event) 296 times over the 24-month monitoring period (Figure 17). The mean flood duration was 15.8 hours per event, and cumulatively represents 27.2% of the total monitoring period (Figure 17). One objective for installing low, fixed crest weirs was to create and maintain a shallow water pool within the impoundment, thus minimizing the effects of prolonged dry periods and maintaining a better water balance. Over the 24- month monitoring period, excluding events where water levels exceeded the fix crest weirs, the interior East Unconfined mean pool stage was 0.083 m (0.27 ft) above NAVD88. In addition, water levels within the East Unconfined treatment unit never dropped to or below 0.0 NAVD88 for the entire monitoring period. However, the Control and East Confined units recorded water levels that were at or below 0.0 NAVD88 for 40.9% and 38.2% of the monitoring period, respectively. Figures 17 are hydrographs for the periods 07-14-11 to 06-30-12 and 07-01-12 to 06-30-13 and represent daily mean water levels for the three study treatment areas. Also included in both figures is a static elevation line representing the spill elevation for the fixed crest weirs separating the East Unconfined marsh and the Little Lake Club Canal. It is apparent from the repetitive flattening curve at around the 0.11m elevation (Figure 17) that the East Unconfined treatment unit is sheltered from diurnal tidal sequencing at or near the 0.11 m NAVD88 elevation.

Salinity Assessments

Water salinities within the three study treatment units were noticeably different from each other. The East Confined treatment unit had the greatest mean salinity of 5.2 ppt (SE=0.086), the East Unconfined sediment cell had the lowest mean salinity of 3.6 ppt (SE=0.026), and the Control marsh had an intermediate mean salinity of 3.8 ppt (SE=0.084); n for all treatment salinity means was 34,503 measurements. The Control, Healthy, and Degraded reference marshes shared a common unimpaired hydrologic system. It is reasonable to assume that salinity within the two-reference marsh sites is equivalent to that of the Control marsh, that is, 3.8 ppt. Salinity variation was greatest within the two most open free-flow marsh units, both of which were influenced by the same hydrologic regime. Within the East Confined marsh, salinity variability was slightly greater (SD=2.33; range 12.2) than that found in the Control marsh (SD=2.27; range 11.0). The East Unconfined treatment unit, which is completely impounded and sheltered from outside diurnal hydrology 72.8% of the time, had the least amount of salinity variation (SD=0.72; range 4.02).

Salinity in the Control and East Confined treatment cells appears to be driven by fluctuating tides, whereas in the East Unconfined treatment cell rainfall is the primary influence. We found positive correlations between tidal fluctuation and salinity within the Control marsh (r= 0.17) and in the East Confined sediment marsh (r= 0.19), but a negative correlation (r= -0.20) within the East Unconfined sediment cell. Figure 18 shows mean daily salinity and water levels for the Control and the East Unconfined unit between July 2011 and June 2012; only a single year of data is included for clarity. Parallel ebb and flow patterns between water and salinity in the Control marsh suggests a positive correlation between fluctuating tides and salinity. However, a near-opposing pattern between water levels and salinity within the East Unconfined unit, where rising water levels are followed by reduced salinity, suggests that salinity within the East

Unconfined unit is influenced by rainfall rather than tides. The large rainfall event, shown in late August-early September 2011, lowered the salinity within the East Unconfined unit by more than 50% over a five-day period and was associated with an increase in salinity within the Control marsh by over 200% during the same time-period. It is reasonable to assume that the large increase in salinity within the Control marsh was not the result of additional freshwater from rainfall, but likely due to strong southerly winds pushing saline water into the Control marsh as the storm approached the study area from the west. Figure 19 shows salinity plotted over water level for the Control and East Unconfined treatment units for the full two-year monitoring period.

Based on the relatively low mean salinity levels and the post-construction floristic composition, marshes within the project area would be classified as an intermediate to brackish. Salinities (even at spike levels) are low enough not to pose any major salt related stress to prevent the establishment or recovery of intermediate to brackish vegetation. However, the sonde-salinities are for open water and may not exactly reflect soil salinities, which are often higher than open water salinities and are more likely related to plant response. Table 7 is a summary of monthly mean water levels within year, and Table 8 shows the mean monthly salinities summarized across the 24-month monitoring period.

Surface Elevation and Hydrology

The primary objective of the Surface Elevation and Hydrology task was to assess the relationships between marsh elevation and hydrology for the four sediment treatment units and the seven Sample Station treatments. We asked the question: How did treatment sites differ in flooding frequency, flood event duration, total hours flooded and percent of total time flooded?

T. Baker Smith and HydroTerra Technologies, engineering firms contracted by the Louisiana Coastal Protection and Restoration Authority, surveyed all treatment surface elevations. Thirty-seven elevation transects, 16 within the East- and West-Confined treatment units and 21 within the East- and West-Unconfined treatment units, were completed between 2012 and 2014. The number of transects within each treatment varied with 11 transects in the East Confined, seven in the East Unconfined, five in the West Confined, and 14 in the West Unconfined treatment units. Orientation of transects within the sediment treatment groups varied somewhat. Transects in the East- and West-Confined units were slightly diagonal from west to southeast, while transects in the East- and West-Unconfined units were positioned west to east, with the exception of three transects in the West Unconfined unit that were oriented north to south. The number of elevation stations that comprised the sediment surface elevation dataset was large and included approximately 5,900 elevations. Figure 20 and Figure 21 are 2012 aerial images of the East project area (Little Lake Hunting Club marsh) and West project area (Temple Island marsh), respectively. Individual elevation points are plotted within transect, transects are plotted within their respective subunits, and transects are labeled for reference.

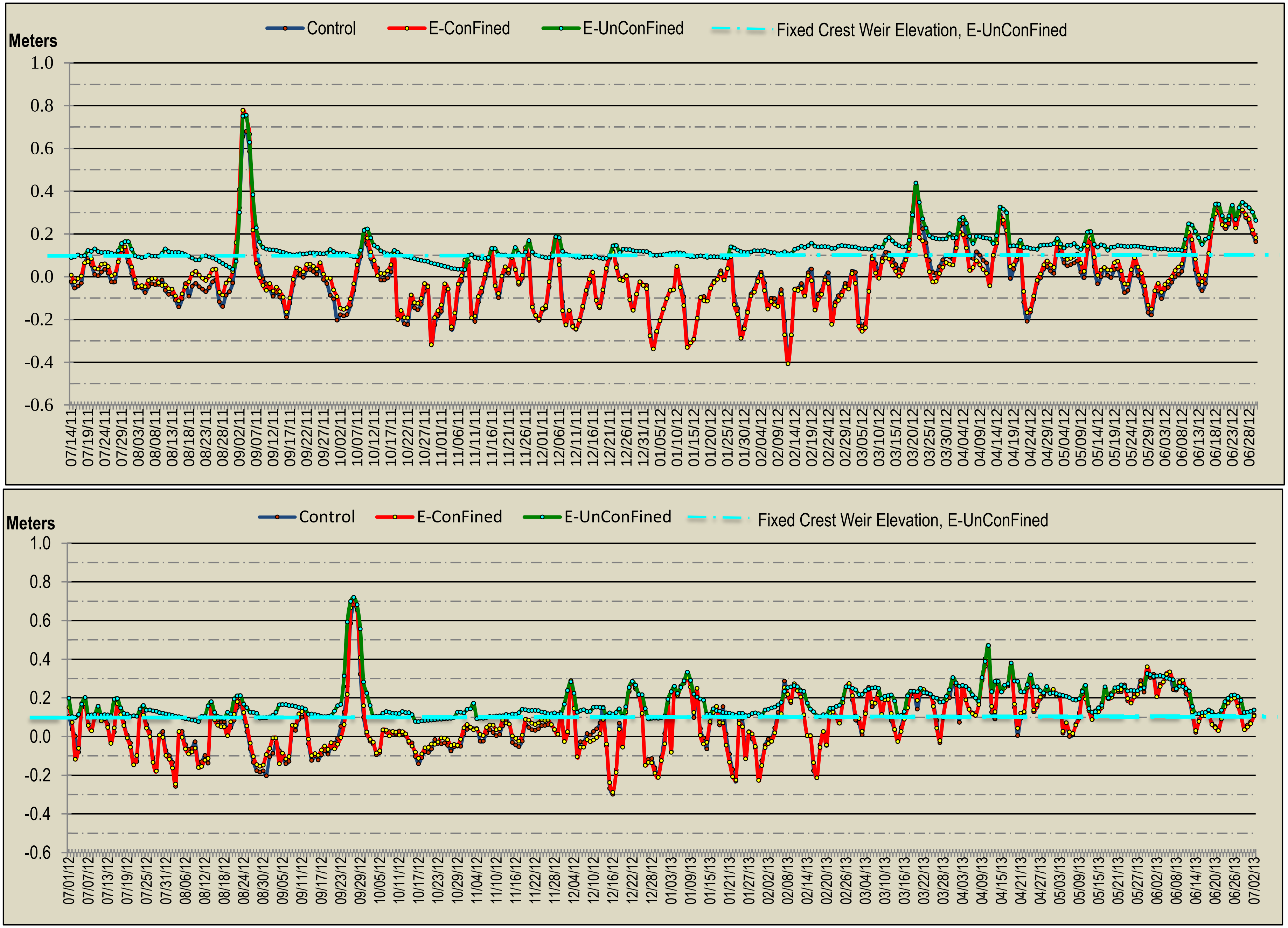


Figure 17 – Hydrologic graphs of daily mean water levels within treatment units. For clarity, the two-year summary is divided into two, one-year graphs. A static elevation line is included on both graphics that represents the elevation of the fix crest weirs separating the East Unconfined treatment unit and the Control marsh hydrology. Water levels are relative to NAVD88.

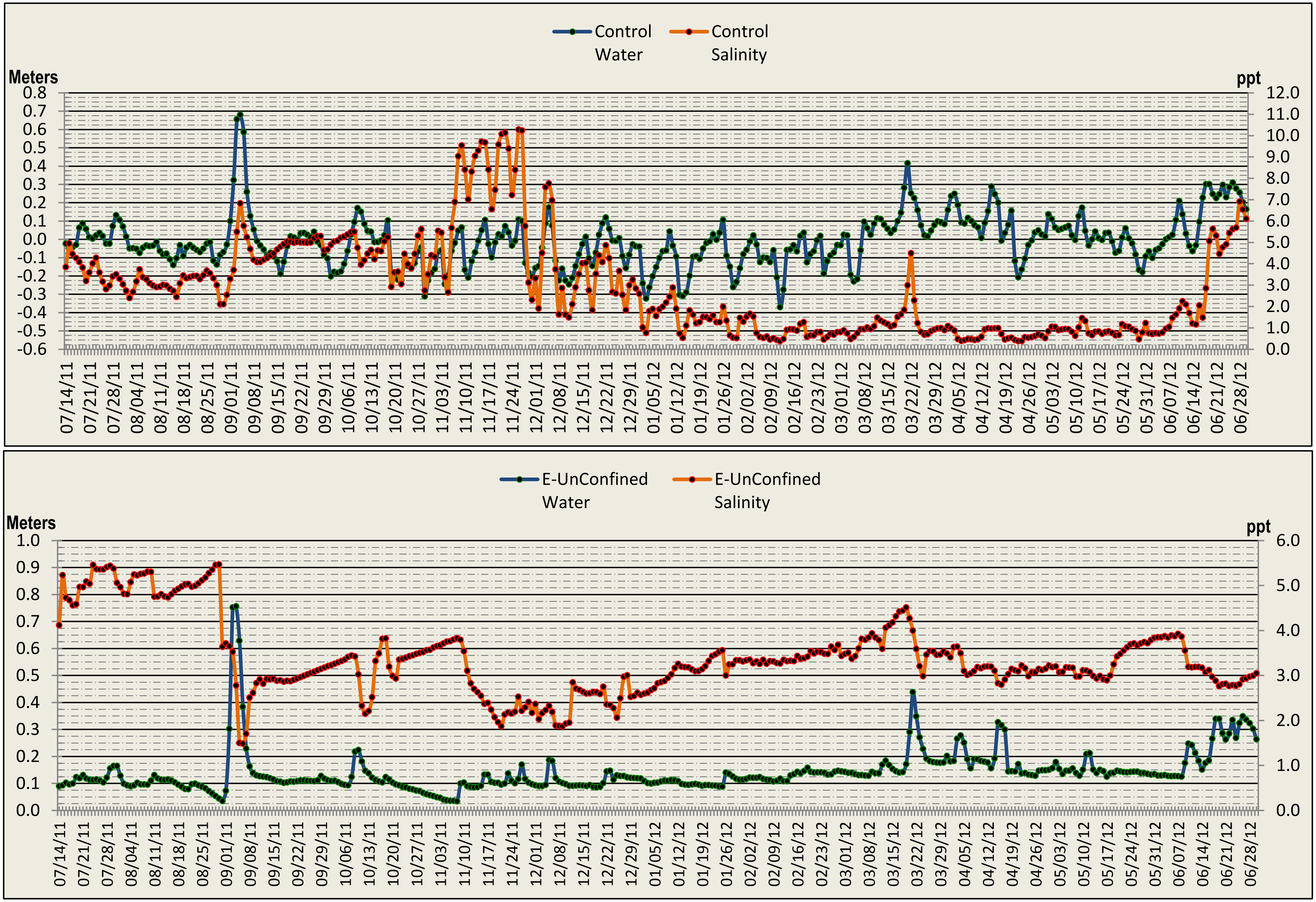


Figure 18 – Hydro-Salinity graphs of the Control (top graph) and East-Unconfined treatment unit (bottom graph) with daily mean salinity plotted over daily mean water levels. Salinity-water movement within the Control treatment units is heavily influenced by tides, demonstrated by a synchronous movement with rising and falling water levels. Whereas salinity within the more impounded East-Unconfined treatment unit, outside of tidal influence (bottom graph), is more positively correlated with rainfall and shows a negative response to increased water levels. Only one-year of data is presented for separation and clarity. Water levels are relative to NAVD88.

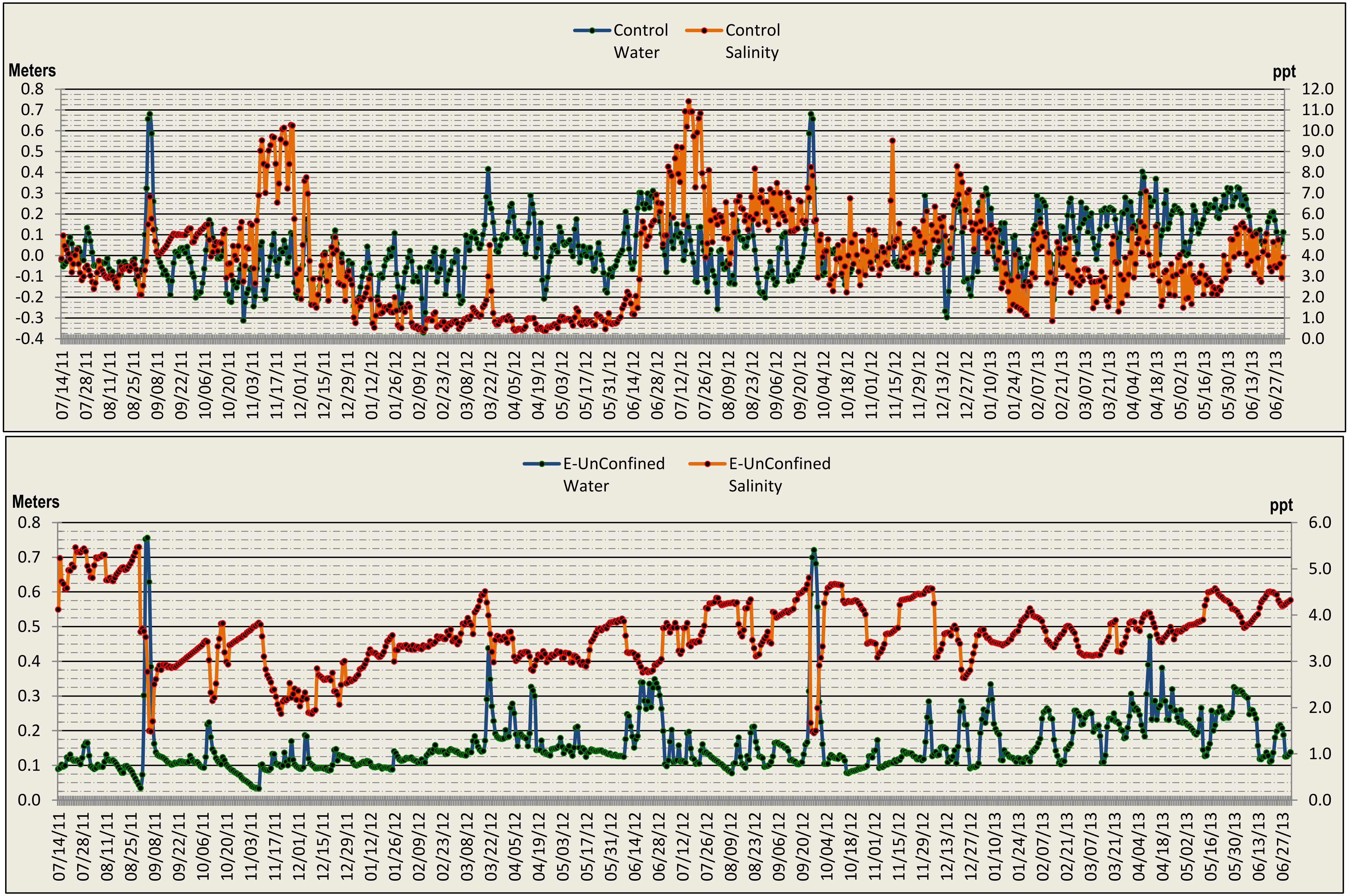


Figure 19 – Complete two-year data set of daily mean salinity plotted over daily mean water levels. Data from the Control (top graph) and East Unconfined (bottom graph) treatment units were selected because the hydrology-salinity relations were noticeably different. Water levels are relative to NAVD88.

Table 7 – Monthly mean salinity and water level (± NAVD88) within treatment units and by year. Monitoring period was 24 months beginning in July 2011 and ending July 2013; sample size n=34,503.

	2011						2012						2013					
	Control		East Confined		East Unconfined		Control		East Confined		East Unconfined		Control		East Confined		East Unconfined	
	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)	Water (m)	Salinity (ppt)
January							-0.12	1.48	-0.13	3.14	0.10	3.10	0.06	3.76	0.05	5.23	0.17	3.60
February							-0.08	0.80	-0.11	2.38	0.13	3.40	0.10	3.60	0.09	4.96	0.18	3.65
March							0.07	1.24	0.04	2.53	0.18	3.76	0.15	2.86	0.15	4.29	0.21	3.36
April							0.07	0.64	0.07	2.22	0.19	3.15	0.20	3.65	0.21	5.39	0.27	3.71
May							0.01	0.88	0.04	2.48	0.15	3.29	0.18	2.80	0.18	4.10	0.22	4.14
June							0.12	3.06	0.15	4.61	0.23	3.23	0.17	4.26	0.17	5.64	0.20	4.15
July	0.03	3.68	0.05	4.53	0.12	5.04	0.04	7.86	0.03	9.26	0.13	3.70	0.09	3.41	0.09	4.53	0.13	4.31
August	-0.06	3.04	-0.03	3.93	0.09	5.00	-0.03	5.91	-0.03	7.19	0.12	3.89						
September	0.06	4.80	0.09	6.40	0.19	2.85	0.06	6.27	0.07	7.62	0.23	3.70						
October	-0.08	4.48	-0.05	5.96	0.11	3.23	-0.04	3.91	-0.02	5.15	0.11	4.19						
November	-0.05	7.42	-0.03	8.65	0.09	2.80	0.03	4.44	0.05	5.92	0.12	3.92						
December	-0.06	3.48	-0.06	5.07	0.11	2.41	0.01	5.59	-0.01	6.98	0.16	3.50						

Table 8 – Mean monthly salinity and water values within treatment and summarized across year for the 24-month monitoring period July 2011 to July 2013.

	July 2011 — July 2013					
	Water (m, ±NAVD88)			Salinity (ppt)		
	Control	E-Confined	E-Unconfined	Control	E-Confined	E-Unconfined
January	-0.031	-0.043	0.135	2.62	4.19	3.35
February	0.005	-0.010	0.155	2.18	3.65	3.52
March	0.107	0.093	0.197	2.05	3.41	3.56
April	0.137	0.138	0.228	2.15	3.81	3.43
May	0.098	0.111	0.184	1.84	3.29	3.72
June	0.148	0.158	0.216	3.66	5.13	3.69
July	0.036	0.037	0.127	6.21	7.41	4.19
August	-0.045	-0.027	0.107	4.48	5.56	4.45
September	0.060	0.081	0.212	5.53	7.01	3.27
October	-0.057	-0.038	0.110	4.20	5.56	3.71
November	-0.010	0.008	0.105	5.93	7.29	3.36
December	-0.025	-0.033	0.134	4.54	6.02	2.95
Mean	**0.035**	**0.040**	**0.159**	**3.75**	**5.16**	**3.59**

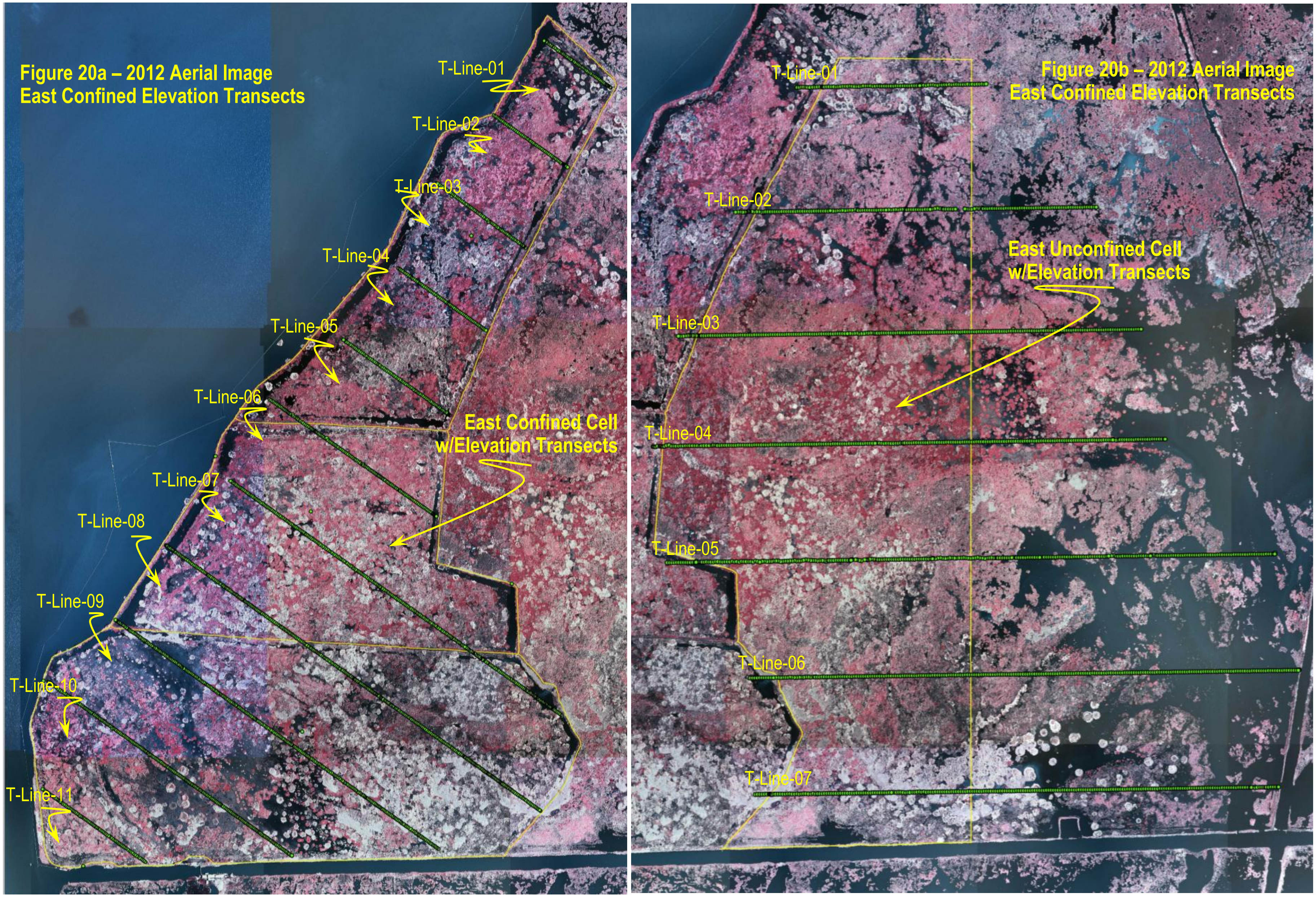


Figure 20a and Figure 20b – Elevation transect lines for the East Confined and East Unconfined treatment units. Elevation points are geo-referenced within transects and transect lines are geo-referenced within sub-units.

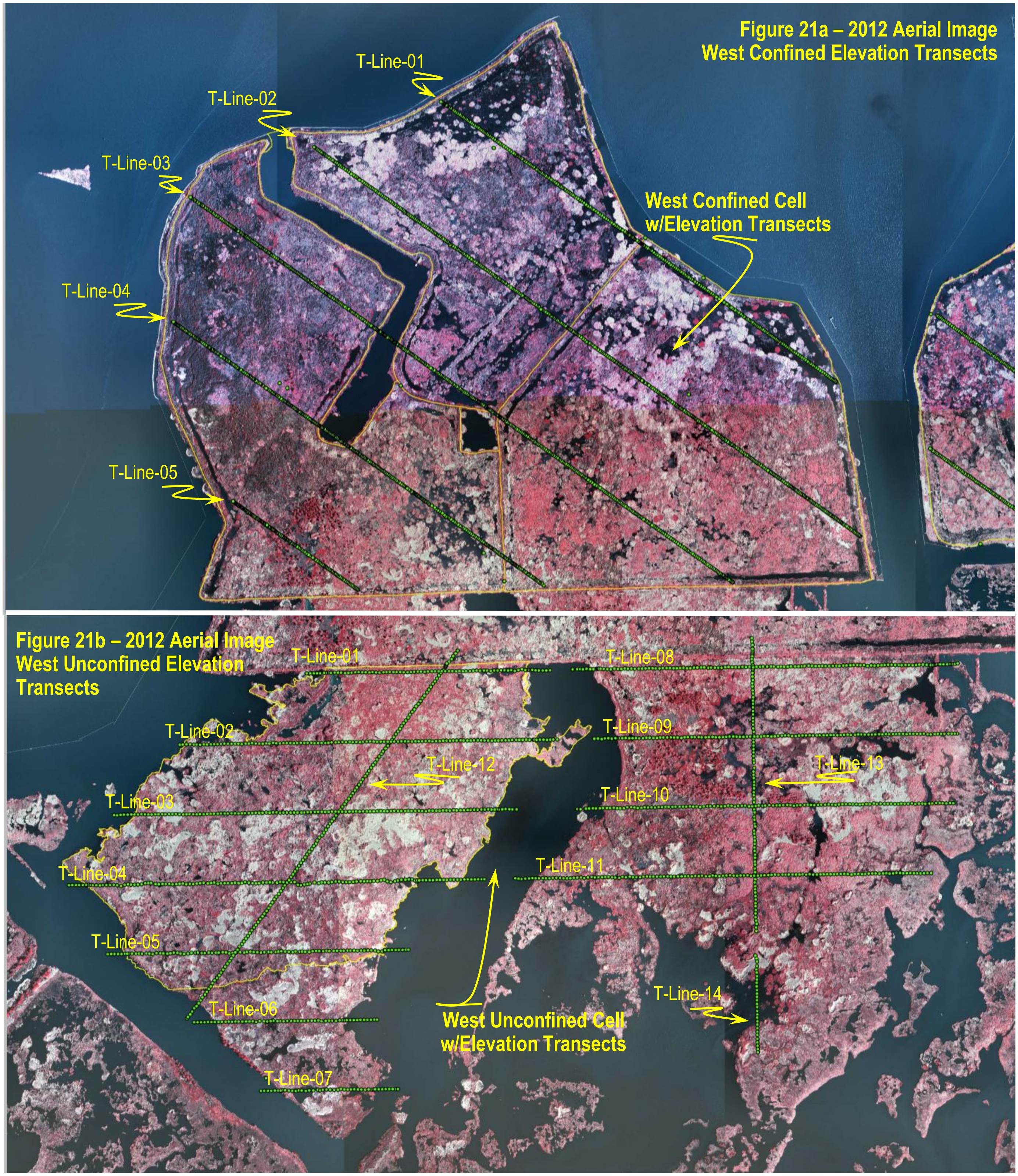


Figure 21a and Figure 21b – Elevation transect lines for the West Confined and West Unconfined treatment units. Elevation points are geo-referenced within transect lines and transect lines are geo-referenced within sub-units.

Table 9- Elevation transect means and descriptive statistics. Table 9 is separated by marsh units (Little Lake and Temple Island) with treatment units (Confined and Unconfined) nested within marsh units. Transect lines were geo-referenced, plotted, and labeled on Figure 20 and 21. Color shading within table indicates statistical similarity and differences between mean elevations of transect lines within treatment unit. Total Unit (tan row) are summary statistics across transects and within treatment units. Elevations are relative to NAVD88.

		Transect	Mean (m)	Standard Error	Standard Deviation	Minimum	Maximum	Range	Count
Little Lake Hunting Club Marsh	East Confined Unit	10	0.463	0.007	0.108	-0.078	0.831	0.910	211
		11	0.400	0.010	0.090	0.097	0.606	0.509	83
		02	0.394	0.033	0.250	-0.434	0.738	1.172	56
		03	0.344	0.024	0.206	-0.499	0.811	1.310	74
		09	0.340	0.009	0.145	-0.314	0.806	1.120	277
		06	0.326	0.014	0.167	-0.409	0.786	1.195	144
		04	0.325	0.015	0.119	0.073	0.735	0.663	67
		08	0.309	0.009	0.162	-0.383	0.998	1.381	333
		07	0.275	0.011	0.182	-0.327	0.984	1.311	278
		05	0.267	0.020	0.187	-0.482	0.603	1.085	87
		01	0.134	0.036	0.278	-0.582	0.464	1.045	59
		Total Unit	0.331	0.004	0.179	-0.582	0.998	1.580	1670
	East Unconfined Unit	01	0.290	0.016	0.140	-0.196	0.567	0.763	81
		04	0.272	0.006	0.087	-0.023	0.618	0.641	226
		03	0.224	0.007	0.098	-0.038	0.799	0.837	198
		05	0.153	0.014	0.234	-0.296	0.871	1.166	267
		07	0.135	0.010	0.149	-0.555	0.500	1.055	226
		06	0.127	0.009	0.139	-0.223	0.447	0.670	228
		02	0.125	0.017	0.217	-0.492	0.756	1.248	161
		Total Unit	0.180	0.005	0.173	-0.555	0.871	1.426	1387

Table 9 (cont.) - Elevation transect means and descriptive statistics. Table 9 is separated by marsh units (Little Lake and Temple Island) with treatment units (Confined and Unconfined) nested within marsh units. Transect lines were geo-referenced, plotted, and labeled on Figure 20 and 21. Color shading within table indicates statistical similarity and differences between mean elevations of transect lines within treatment unit. Total Unit (tan row) are summary statistics across transects and within treatment units. Elevations are relative to NAVD88.

		Transect	Mean (m)	Standard Error	Standard Deviation	Minimum	Maximum	Range	Count
Temple Island Marsh	West Confined Unit	05	0.502	0.010	0.113	0.187	0.963	0.777	132
		04	0.384	0.017	0.310	-1.113	1.272	2.385	332
		02	0.345	0.005	0.110	-0.203	1.176	1.378	408
		03	0.309	0.014	0.288	-0.995	1.259	2.254	445
		01	0.165	0.012	0.186	-0.504	0.738	1.242	256
		Total Unit	0.327	0.006	0.247	-1.113	1.272	2.385	1573
	West Unconfined Unit	13	0.313	0.011	0.095	0.053	0.532	0.479	80
		10	0.309	0.013	0.131	-0.121	0.597	0.718	109
		11	0.292	0.013	0.138	-0.349	0.633	0.983	116
		06	0.287	0.015	0.092	0.058	0.536	0.477	38
		12	0.284	0.007	0.077	0.010	0.542	0.532	120
		08	0.283	0.019	0.214	-0.474	0.857	1.331	122
		09	0.272	0.014	0.138	-0.138	0.955	1.092	102
		03	0.254	0.009	0.096	-0.050	0.533	0.583	107
		04	0.239	0.022	0.229	-0.589	0.607	1.196	108
		02	0.237	0.012	0.126	-0.147	0.580	0.727	111
		05	0.205	0.025	0.224	-0.514	0.512	1.026	83
		14	0.192	0.017	0.090	0.066	0.341	0.275	29
		01	0.146	0.026	0.223	-0.601	0.644	1.245	76
		07	0.134	0.034	0.202	-0.187	0.783	0.970	35
		Total Unit	0.257	0.005	0.166	-0.601	0.955	1.556	1236

To determine flooding frequencies, we first defined what constitutes a flood event. A flood event was defined as a rising and falling water cycle that begins with a water level below a reference elevation, rises to or exceeds the reference elevation, and ends when water falls below the initial elevation; one full cycle (regardless of the flood duration) would constitute one flood event for determining frequency. For example, if 5 cm above ambient marsh is the reference elevation and the sample period is four days, we would count two flooding frequencies if on day one, water rises from less than 5 cm to ≥5 cm and stays at that level for two days, then falls to less than 5 cm on the third day and rises again to ≥5 cm on day four. Because water levels rose from below the target elevation, then equaled or exceeded the target elevation, then subsequently fell below the target elevation twice within the sample period, two flooding events (or frequencies) had occurred.

To acquire a set of target elevations needed to complete the hydrology-sediment analysis, using transect data, a series of ANOVA tests were completed to determine mean variability between and among treatment units. We found that there was a significant difference in sediment elevation ($p=0.001$) between the Confined and Unconfined treatments, a significant difference between the East- and West-Unconfined treatments ($p=0.001$), but no significant difference in mean elevation between the East and West Confined treatments ($p>0.05$,). When we compared transect means within specific treatments, for example, within the East Confined treatment unit, we found a significant difference ($p<0.0001$) among transect elevations, as was the case in each of the four treatment units. For clarity, we constructed a number of pair-wise comparison tables to identify which transects are statistically alike and which are different. Descriptive statistics and the comparison results are contained in Table 9. Results are separated into East (Little Lake Hunting Club) and West (Temple Island) marsh units with sediment treatment levels (Confined and Unconfined) nested within marsh units. Transects are listed in descending order by means and transects linked with like-color shading (within treatment units) were not significantly different. For example, within the East-Confined treatment unit (Little Lake Hunting Club marsh Table 9), 11 transects formed five groups based on their statistical equivalency, with transects 10, 11, and 02 (yellow shading) at the high end of the elevation scale. Transect-01 (gray shading) is a single transect group member at the lowest end of the elevation scale. Three additional groups (orange, green, blue) formed separate groupings at the mid-high, mid, and mid-low elevation range. The number of observations within each transect (n= elevation pts.) is listed in the last column under Count.

Target elevations for the hydrology-sediment comparisons were calculated for only three of the four treatment units, as the East- and West-Confined units were statistically equivalent. Of the two units, we selected the East Confined as the more applicable unit, as one of the three water sondes was located directly within the East Confined unit boundaries. To broaden the range of assessments, we calculated three target elevations (mean-high, mean, and mean-low) for each of the three treatment units using elevations from 31 transects. The calculated values were then matched to a specific transect with a mean value closest to that of the calculated target value. For the seven Sample Station treatments, we used the calculated mean based on survey data taken specifically within the seven treatments. The calculated target elevation values, their respective paired transects, and Sample Station mean values used in the hydrology-sediment-elevation assessments are listed in Table 10; elevation values are expressed in meters and are NAVD88 Geoid12A corrected.

Using the transect elevation data, we found that sediment elevations were significantly different between sediment treatment sites. Exceptions were the two primary treatment East- and West-Confined units, which were statistically equivalent. Elevations within the East- and West-Confined sites (on average) were highest at 0.331 m (SE=0.004, n=1670) and 0.327 m (SE=0.006, n=1573), respectively. The East Unconfined unit contained the lowest mean elevation at 0.180 m (SE =0.005, n=1387), with the West Unconfined at an intermediate mean elevation of 0.257 m (SE =0.005, n=1236). Mean transect elevations within individual treatment sites were significantly different as well, and often contained a significantly wide mean range between transects. For example, in the East Confined treatment site, transect-10 had the highest mean elevation at 0.463 m (SE =0.007, n=211) and transect-01 had the lowest mean elevation at 0.134 m (SE =0.036, n=59), a 0.329 m (245%) differential in mean elevation. A complete list of transect mean elevations, standard errors, and n observations are contained in Table 9 and visually represented in Figure 22.

When assessing critical flooding events, such as frequency and duration of flooding, we found an interaction between treatment hydrology, elevation, and containment (levees). For example, within the mean-low elevation, flooding frequency increased as elevation decreased across treatment sites. Over the 24-month sampling period, the higher elevated East Confined site experienced 91 flooding events, considerably fewer than the intermediately elevated West Unconfined site with 250 events. The East Unconfined site, with the lowest mean sediment elevation, had the greatest number of flooding events with 378. Flooding frequency in the Healthy Reference marsh, with a slightly higher (+0.05 m) elevation than the East Confined mean-low elevation, had 95 flooding events, a commensurable increase in flooding events given the slight differences in elevations. The elevation-to-flood frequency relationship was accurate across all treatment-by-elevation combinations, with the exception of the mean-high elevations in the East- and West-Unconfined treatment units, where flooding frequency of the two sites was equal. Although the East- and West-Unconfined units were significantly different in elevation, equilibrium in flooding frequency occurred between the two units when the outside hydrology exceeded the fixed elevation protection weirs separating the two units. During high water events, water from outside of the East Unconfined unit overtops the low-water protection weirs and merges with impounded water to form a single water body. Consequently, flooding frequency then becomes a function of high water confluence and sediment elevations, with the retaining levee influence removed.

All of the sediment treatment units were initially either complete or semi-impoundments prior to sediment loading. The hydrologic efficiency of the remaining levee systems (post-construction) to impound or exchange water with the larger outside marsh factors into many of the unit’s hydrologic features. The West Unconfined unit, for example, was constructed with a minimum number of low elevation retaining levees and construction materials designed for a short-life duration. Post-construction the levees around the West Unconfined unit have deteriorated substantially, and water exchange with the outside Control hydrology was fluid and unrestricted. We found the West Unconfined unit to be the least efficient in water retention of the three sediment treatment units. Retaining levees around the East-and West-Confined site, however, were constructed to resist breaching during sediment loading and designed to impound sediments to a higher elevation. Soil materials used to construct the Confined unit levee systems were

Table 10 – Calculated target elevation, with respective paired transects for flood-surface elevation comparisons in three treatment units. Also calculated mean elevations for the seven permanent Sample Station treatments. Elevations are in meters and relative to NAVD88.

	Treatment Unit	Mean-Low		Mean		Mean-High	
Transect Elevation Data	**East Confined**						
	Calculated Elevation	0.205		0.331		0.451	
	Paired Transect # and Elevation	Trans # - 05	0.267	Trans # - 06	0.326	Trans # - 10	0.463
	West Unconfined						
	Calculated Elevation	0.129		0.257		0.349	
	Paired Transect # and Elevation	Trans # - 07	0.134	Trans # - 03	0.254	Trans # -13	0.313
	East Unconfined						
	Calculated Elevation	0.036		0.18		0.297	
	Paired Transect # and Elevation	Trans # - 02	0.125	Trans # - 05	0.153	Trans # - 01	0.29
Sample Site Data		**Mean**					
	Healthy Reference	0.264					
	Degraded Reference	0.149					
	Confined High	0.513					
	Confined Medium	0.309					
	Unconfined Medium	0.328					
	Unconfined Low	0.240					
	Unconfined Very Low	0.176					

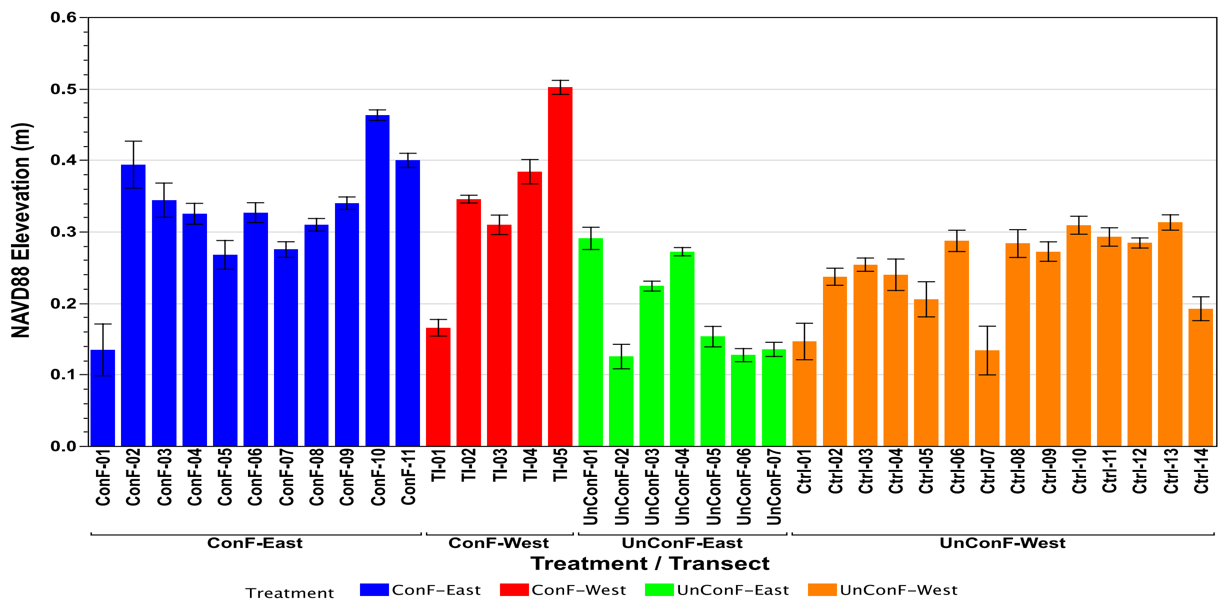


Figure 22 – Mean elevation of thirty-seven elevation transect lines by treatment.

selected for strength and to provide a longer period of sediment stability. Retaining levees around the Confined units were intentionally breached at several locations post-construction, but remained intact. Water exchange is uneven with water moving unrestricted over the tops of levees during high water events, but somewhat restricted through small breach points otherwise. We found the East- and West-Confined units second in water retention and water exchange efficiency. Placement of the East Unconfined unit was selected to take advantage of an existing levee system. However, unlike its West counterpart unit, the East Unconfined retaining levees are substantially elevated and permanent. As noted earlier, the only water exchange between the East Unconfined unit and the outside (Control) hydrology is through several small low-water weirs. Consequently, the East Unconfined unit is the most efficient in retaining water and least efficient in exchanging water, either from rainfall or from high water events.

When comparing the duration of flood events between treatments, the rank order of sediment treatment units changes from that of flooding frequency. The West Unconfined unit, with the greatest capacity to exchange water and the least capacity to retain water, consistently had the shortest mean flooding period of 13.7 hours per flood event. The East Confined unit, with an intermediate capacity to exchange and retain water ranked second with a mean of 19.7 hours per flood event, and the heavily impounded East Unconfined site ranked last with a mean of 21.9 hours of flooding per flood event. Sediment surface flooding at higher elevations is generally only impacted when water from one hydrologic system merges with another, forming one larger common system and overtopping retaining levees or low-water weirs. Flooding frequency and flood duration then become a function of how quickly water can recede from the impacted treatment unit. Units with relatively intact retaining levees and slow exchange capacity hold water longer at higher elevations, thus increasing the number of flood events at higher elevations and prolonging periods of inundation at lower elevations. As an example, compare total flood duration between the East Unconfined (7,398 hrs) and the Deteriorating Reference marshes (3,464 hrs) at their mean elevation. Although the East Unconfined unit is 0.031 m (20.8%) higher in elevation than the Degraded Reference marsh, the East Unconfined unit (with a very efficient impoundment levee system) held water 114.2% longer than the lower elevation and more open Deteriorating Reference site.

The last measurement in assessing hydrology-sediment relationships was to determine the overall flood duration within treatments; that is, for a specific elevation, what were the total hours and percent total time (over 24-months) that treatment units were flooded. We found that treatment sediment elevation and flood duration are not completely related. In ascending order of total time flooded, we found that the East Confined (0.331 m) had the highest sediment surface elevation and the least amount of cumulative flooding with 6.7% or 1,161.5 hours. The Healthy Reference marsh (0.264 m) was second in both surface elevation and in cumulative flooding, with 7.7% or 1,325.0 hours. The West Unconfined (0.257 m) treatment unit was third in sediment elevation and third in total flooding with 8.4% or 1,450.5 hours. The Degraded Reference marsh (0.149 m) had the lowest surface elevation of the five units, but was second-to-last in cumulative flooding with 20.0% or 3,463.5 hours. The East Unconfined (0.180 m) treatment unit was second-to-last in surface elevation, but last in cumulative flooding with 42.9% or 7,398 hours.

Relative to the permanent sampling stations, flooding frequency was least within the High Confined treatment site (elevation 0.513 m) with four flooding events and greatest in the Unconfined Very Low (elevation 0.76 m) and Degraded Reference (elevation 0.149 m) with 259 and 219 flooding events, respectively. The Confined Medium (elevation 0.309 m), Unconfined Medium (elevation 0.328 m), and Unconfined Low (elevation 0.024 m) sample station treatments had intermediate flooding frequencies of 68, 52, and 131, respectively, over the 24-month monitoring period. Total time flooded for the permanent sampling stations had similar trends as flooding frequency with the lowest values found for the Confined High treatment at 1% or 136.5 hours and higher values found in the Unconfined Very Low treatment with 24.3% or 4,190 hours and Degraded Reference marsh with 20% or 3,453.5 hours total time flooded. Other restoration marshes were intermediate in total time flooded with the Unconfined Medium and Unconfined Low sample station treatments having 3.8% (647 hours) and 10.5 % (1,783.8 hours), respectively. The Healthy Reference marsh flooded 7.7% over the 24-month monitoring period or 1,325 hours. A complete list of elevation, flooding statistics and number of observations for the three sediment treatment units and the seven sample station treatments are listed in Table 11 and Table 12, respectively. Figure 23 compares sediment surface elevations and total percent time flooded for the three treatment units, the Healthy and Degraded Reference marshes, and Figure 24 compares the surface elevation and total percent time flooded for the seven permanent Sample Station treatments.

Table 11 – Summary table of target elevations, flooding frequency, and flood duration for three Treatment Units and two Sample Station treatments. [1]*Elevations* for Treatment Units were calculated from transect data and site elevations for Sample Sites; [2]*Flooding Frequency* is the number of flood events that occurred at the respective elevation; [3]*Average Flood Duration* is the mean length of time (hrs) per flood event; [4]*Cumulative Flood Time* (hrs) is the total length of time flooded; and [5] *Total Time Flooded* is Cumulative Flood Time expressed as a percent of the total potential time. [6]*n pts* are the number of measurements used for calculating flooding statistics. The total monitoring time was approximately 24-months or 17,251 cumulative hours.

		Transect Line Elevation			Sample Station Levels	
	Flooding Events	**E-CT** (n=1670)	**W-UCT** (n=1387)	**E-UCT** (n=1236)	**Healthy** (n=5)	**Degraded** (n=5)
Mean-Low	[1]Elevation (m, NAVD88)	0.205	0.129	0.037		
	[2]Flooding Frequency (No.)	91	250	378		
	[3]Ave Flood Duration (hrs/flood event)	21.9	15.5	28.0		
	[4]Cumulative Flood Time (hrs)	1,996.0	3,886.0	10,583.0		
	[5]Total Time Flooded (%)	11.6%	22.5%	61.4%		
	[6]n pts => target elevation)	3992	7,772	21,168		
Mean or Mid	[1]Elevation (m, NAVD88)	0.331	0.257	0.180	0.264	0.149
	[2]Flooding Frequency (No.)	64	104	398	95	219
	[3]Ave Flood Duration (hrs/flood event)	18.1	13.9	18.6	13.9	15.8
	[4]Cumulative Flood Time (hrs)	1,161.5	1,450.5	7,398.0	1325.0	3,453.5
	[5]Total Time Flooded (%)	6.7%	8.4%	42.9%	7.7%	20.0%
	[6]n pts => target elevation)	2,323	2,901	14,797	2,650	6,907
Mean-High	[1]Elevation (m, NAVD88)	0.451	0.349	0.298		
	[2]Flooding Frequency (No.)	9	65	65		
	[3]Ave Flood Duration (hrs/flood event)	19.2	11.6	19.2		
	[4]Cumulative Flood Time (hrs)	173.0	751.5	1,184.0		
	[5]Total Time Flooded (%)	1.0%	4.4%	6.9%		
	[6]n pts => target elevation	346	1,503	2,368		

Table 12 – Summary table of calculated mean elevations for the seven Sample Station treatments. [1]Elevations were calculated from site data; [2]*Flooding Frequency* is the number of flood events that occurred at a respective elevation; [3]*Average Flood Duration* is the mean length of time (hrs) per flood event; [4]*Cumulative Flood Time* (hrs) is the total length of time flooded; and [5]*Total Time Flooded* is the Cumulative Flood Time expressed as a percent of the total potential time. [6]*n pts* are the number of measurements used for calculating flooding statistics. The total monitoring time was approximately 24-months or 17,251 cumulative hours.

	Sample Station Treatment-Levels						
Flooding Events	**Confined High** (n=5)	**Confined Medium** (n=5)	**Unconfined Medium** (n=5)	**Unconfined Low** (n=5)	**Unconfined Very Low** (n=5)	**Degraded Reference** (n=5)	**Healthy Reference** (n=5)
[1]Elevation (m, NAVD88)	0.513	0.309	0.328	0.24	0.176	0.149	0.264
[2]Flooding Frequency (No.)	4	68	52	130	259	219	95
[3]Ave Flood Duration (hrs/flood event)	34.1	21.1	12.5	13.7	16.3	15.8	13.9
[4]Cumulative Flood Time (hrs)	136.5	1,435.5	597.0	1783.8	4,190.0	3,453.5	1,325.0
[5]Total Time Flooded (%)	1.0	9.3	3.75	10.35	24.25	20.0	7.7
[6]n pts => target elevation	273	2,871	1294	3569	8381	6,907	2,650

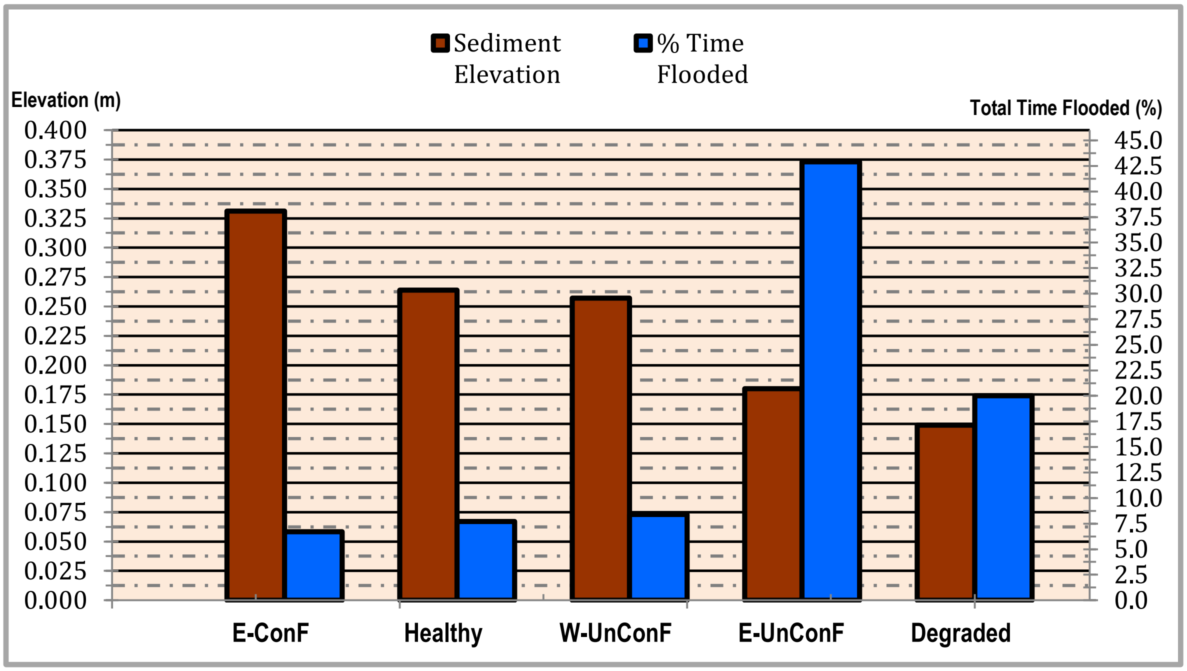


Figure 23 – Sediment surface elevations and total percent time flooded for three Sediment Treatment Units and the Healthy and Degraded Reference marshes.

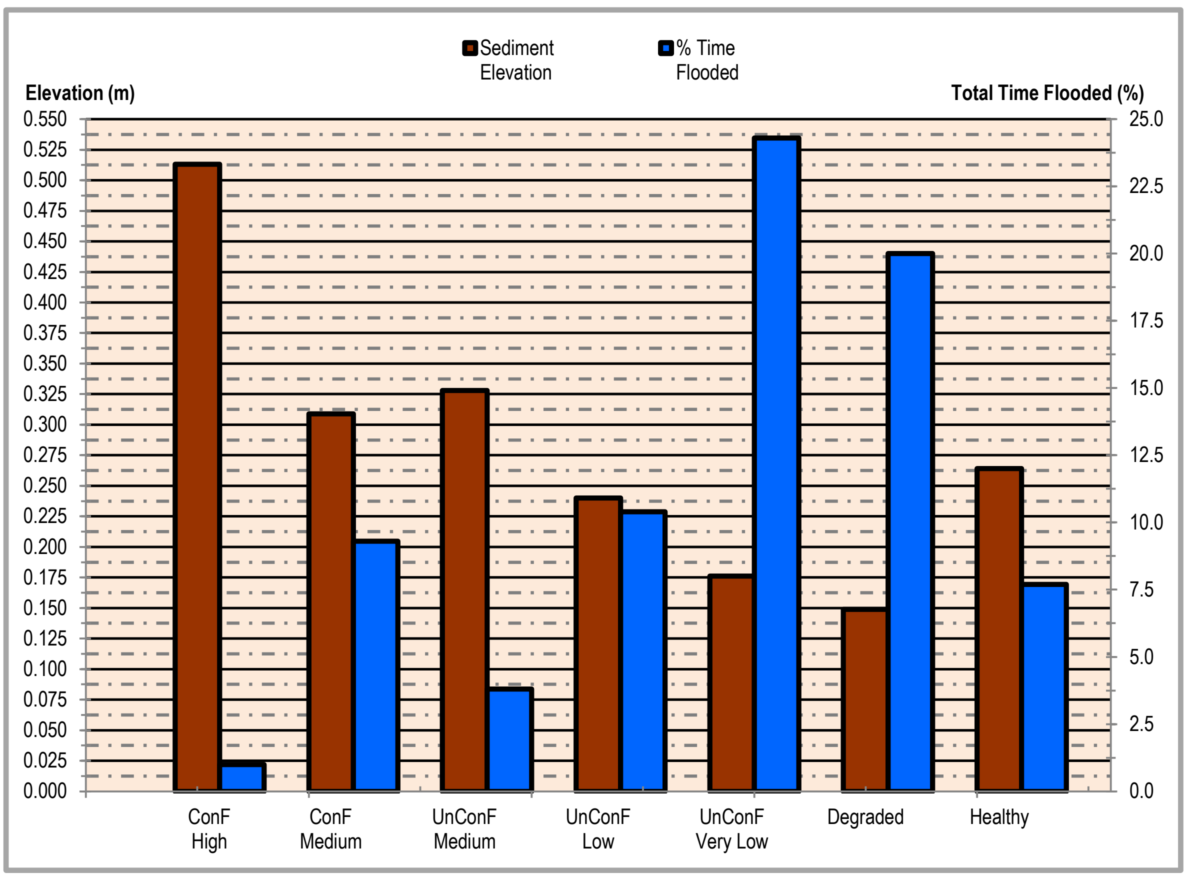


Figure 24 – Sediment surface elevations and total percent time flooded for the Sample Station treatments, which include the Degraded and Healthy Reference marshes.

Ground Truth Assessment of Marsh Structure and Function

Environmental Conditions

Soil Surface Elevation at Sampling Stations

The 2011 surface elevations of the restoration treatments and reference marshes varied significantly ($P < 0.0001$, Figure 25). The elevation of the Healthy Reference marsh was 0.23 m NAVD88, which was significantly higher than for the Degraded Reference marsh (0.15 m NAVD88). Sediment-addition produced marsh elevations statistically greater than for the Degraded Reference marsh, excepting the Unconfined Very Low treatment (Figure 25). However, two restoration treatments, the Confined High and Unconfined Medium, had elevations significantly greater than the Healthy Reference marsh (Figure 25). In fact, the Confined High treatment yielded an elevation approximately twice as high as the Healthy Reference marsh. In contrast, the elevations of the Confined Medium and Unconfined Low treatments were statistically equivalent to the Healthy Reference marsh (Figure 25).

Temporal Changes in Marsh Elevations at Sampling Stations

Confined Fill-Treatments. Average elevation at, or adjacent to, each confined fill sampling station significantly varied with time ($P<0.0001$), but were unaffected by treatment ($P=0.29$) or their interaction ($P=0.41$). Marsh elevation before construction (Fall 2008) was -0.32 ± 0.08 m NAVD88 (n=10). Twenty-eight days after fill placement (Fall 2009), marsh elevation significantly increased to 0.88 ± 0.04 m and thereafter significantly decreased to 0.41 ± 0.04 m by Fall 2011 and 0.30 ± 0.05 m by Spring 2014 (n=10); elevations for the latter two dates did not statistically differ (square root [value + 2] transformed). Marsh elevations for Confined High and Confined Medium treatments (Figure 26) did not significantly differ. Although initial elevations after fill placement exceeded the target elevation of 0.762 m (+2.5 feet), elevations decreased over time in both fill-treatments to average elevations at or just above the Healthy Reference marsh (Figure 26). However, the Confined Medium fill-treatment reached the elevation of the Healthy Marsh sooner than the Confined High fill-treatment, which after ca. 3.5 years was higher, on average, than the Confined Medium fill-treatment (Figure 26).

Unconfined Fill-Treatments. Station elevations were surveyed in the fall of 2011 and the fall of 2012. Elevations significantly varied among treatments ($P=0.004$) with the Medium Elevation treatment [0.33 ± 0.02 m] greater than the Very Low Elevation treatment [0.19 ± 0.03 m]); the Low Elevation treatment (0.25 ± 0.02 m) was intermediate. No statistically significant time effect ($P=0.60$) or interaction of time and treatment ($P=0.92$) was found as indicated in Figure 27. We propose that the rapid consolidation and compaction from the initial fill, as quantified for the confined fill-treatment (Figure 26), had stabilized by the fall of 2012 in the unconfined fill-treatment. However, elevations will certainly decrease more over time, albeit at a much slower rate, due to further compaction, loading, and regional subsidence.

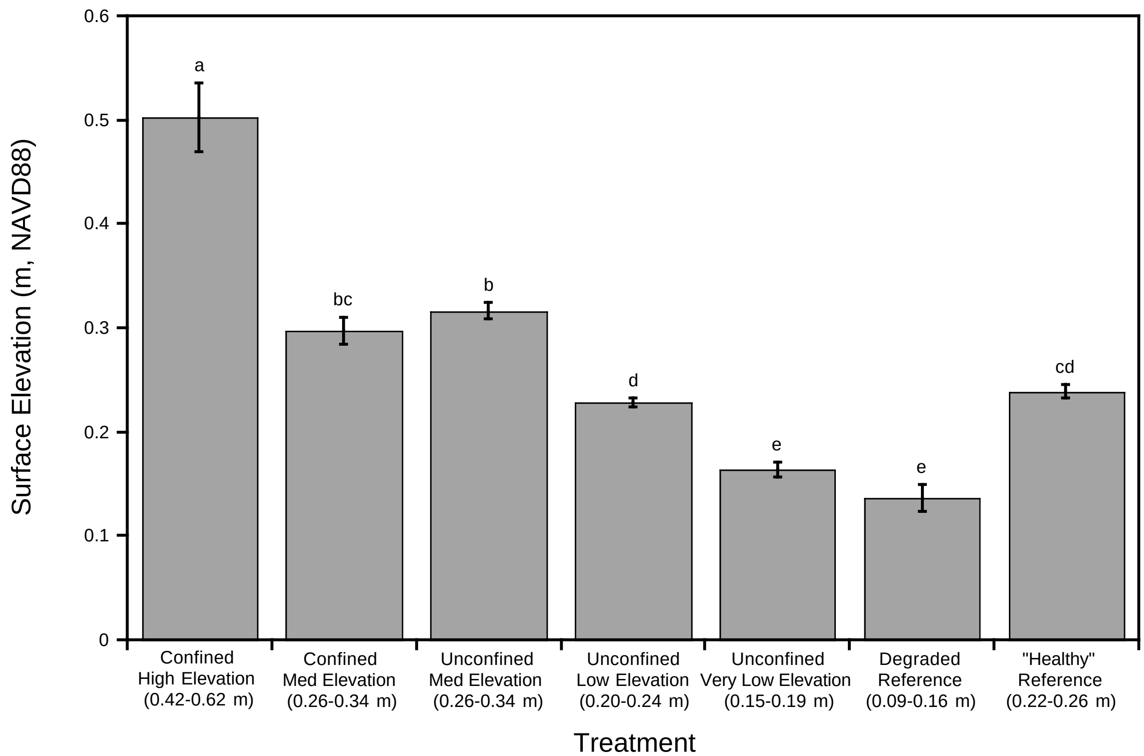


Figure 25- Treatment effects on surface elevation at sampling stations surveyed during summer/fall 2011. Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. Means with different letters identify significant differences.

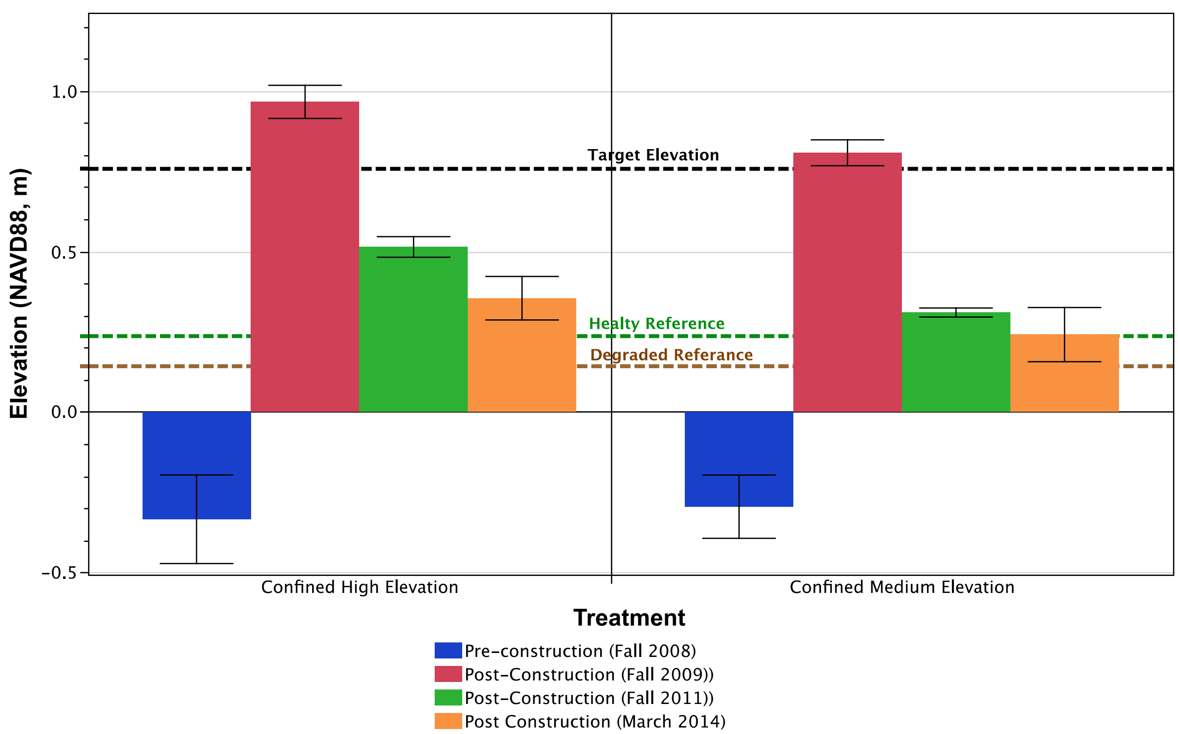


Figure 26 - Comparison of surface elevations between Confined High and Confined Medium treatments at sampling stations over time. Values presented as the mean ± 1 SE (error bars) (n=5) for each date within a treatment. Although the interaction between treatment and time was not statistically significant, the 2-way interaction graph is presented for descriptive purposes. Significant time effects are provided in the text.

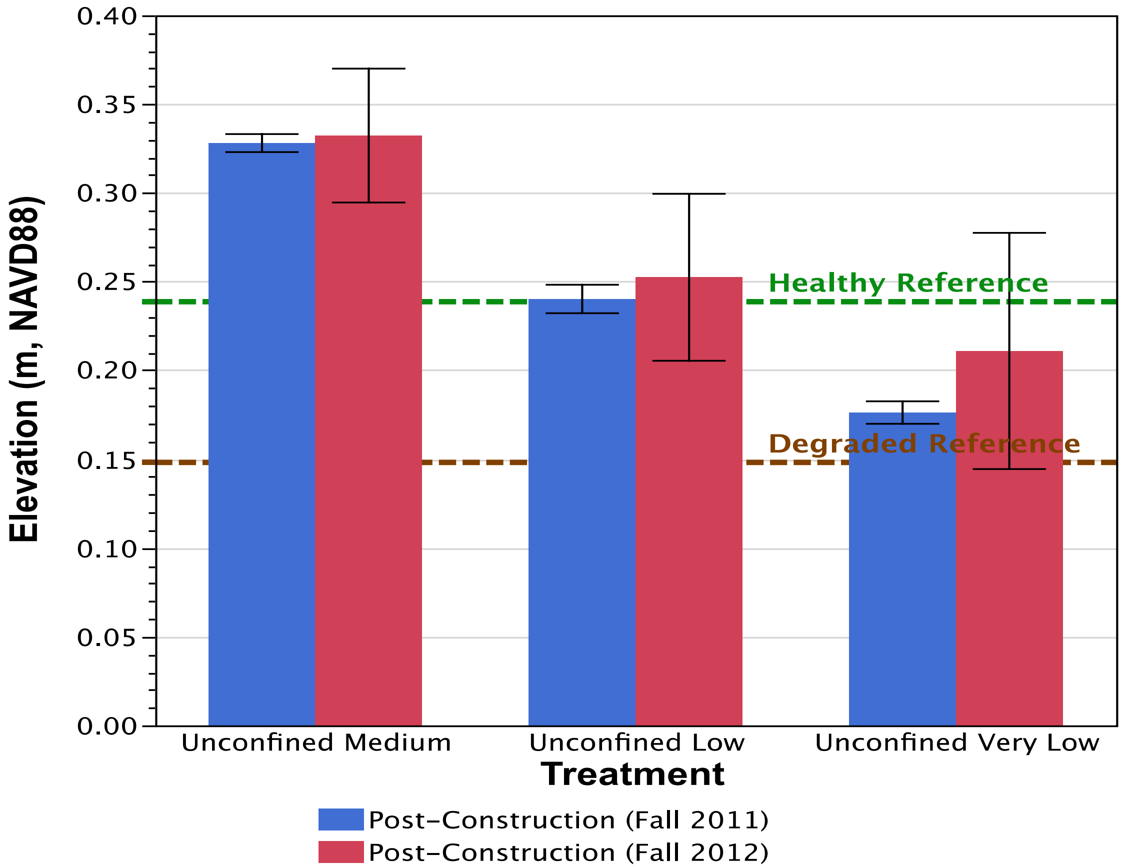


Figure 27 - Comparison of marsh surface elevations of unconfined treatments (Medium, Low, and Very Low) between Fall 2011 and Fall 2012 at sampling stations. Values presented as the mean ± 1 SE (error bars) (n=5) for each date within a treatment. Although the interaction between treatment and time was not statistically significant, the 2-way interaction graph is presented for descriptive purposes. Significant treatment effects are provided in the text.

<u>Temporal Changes in Marsh Elevation based on Transect Survey Points</u>

In addition to elevations surveyed at each of the 35 sampling stations, CPRA-contracted engineering firms surveyed multiple transects in the confined fill-areas (Figure 28) in 2008 (preconstruction) and 2009 and 2014 (both post construction) and in the unconfined fill-areas (Figure 29) in 2012 (also see Figures 20 and 21). Before sediment addition, much of the project-area marsh was in a state of deterioration, characterized by fragmented vegetation and numerous ponds (see report section *2008 Spatial Assessments*, p. 17). Marsh elevations in the confined project area in 2008 ranged from a minimum of -1.04 m to a maximum of 0.77 m, averaging -0.15 m (Table 13). In contrast, post-construction elevations in 2009 ranged from a minimum of 0.49 m to a maximum of 1.37 m, averaging 0.92 m (Table 13). Four years later in 2014, post-construction elevations ranged from a minimum of -1.11 m to a maximum of 1.49 m, averaging 0.34 m (Table 13). The change in elevation over time in the confined treatment-areas is visually apparent in Figure 28, as the survey point colors, which represent different elevations, change over time. The temporal change in average elevation is quantified in Figure 30, in which elevations west and east of the Harvey Cutoff are differentiated.

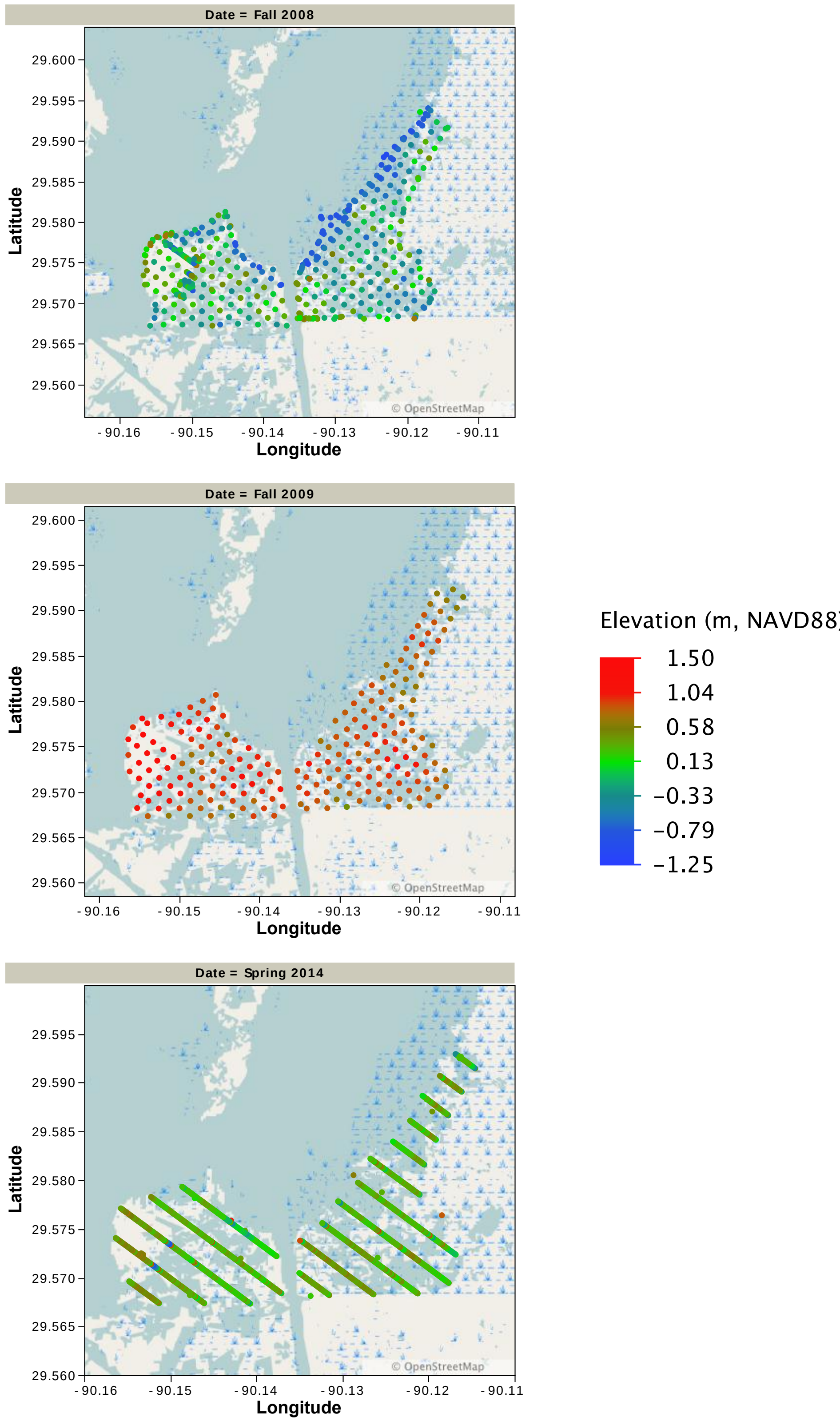


Figure 28 - Elevations (m, NAVD88) of transect survey points in 2008 (pre-construction) and 2009 and 2014 (post-construction) within the confined fill-areas. See Table 13 for summary statistics.

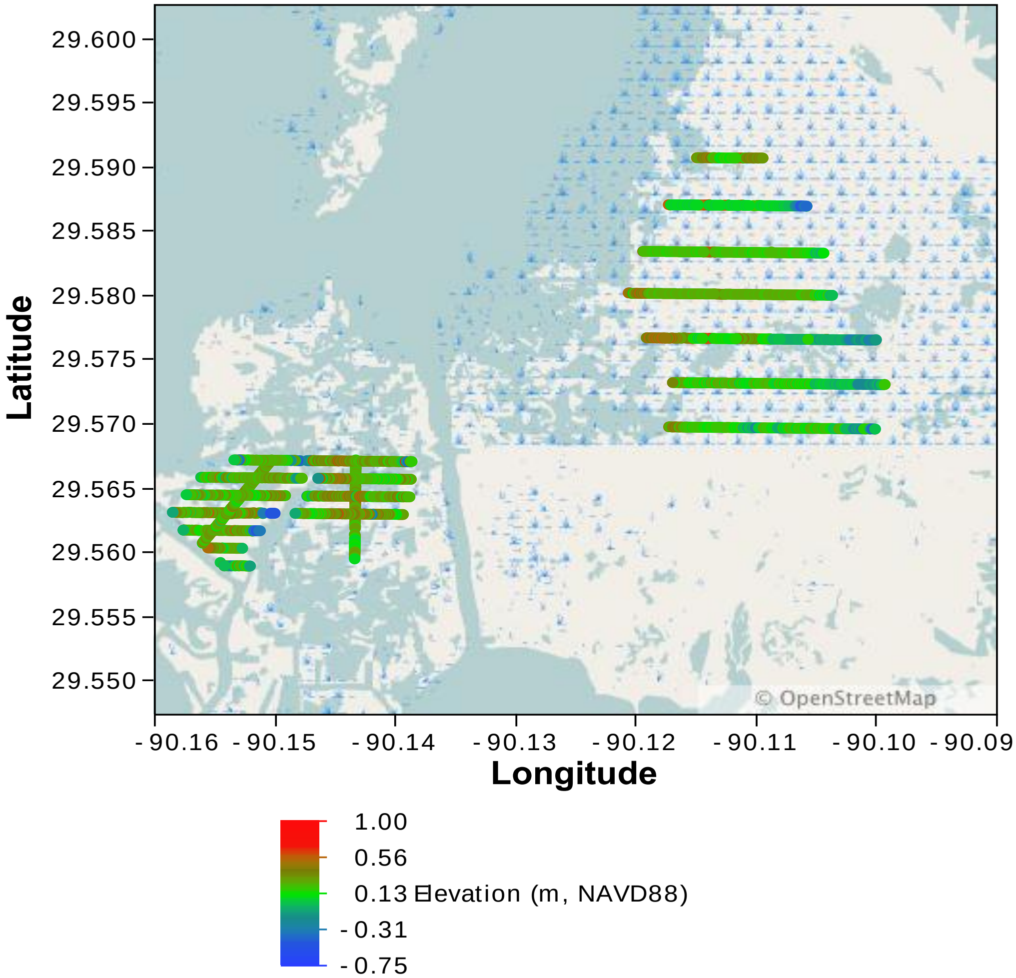


Figure 29 - Elevations (m, NAVD88) of transect survey points in 2012 within the unconfined fill-areas.

Table 13- Elevation summary statistics based on transect survey points throughout confined and unconfined treatment-areas. Elevation units are meters relative to NAVD88.

	Confined			Unconfined
	Pre-construction	Post-construction		Post-construction
	2008	2009	2014	2012
N	482	219	3282	2624
Mean	-0.15 a	0.92 b	0.34 c	0.22 d
SE	0.02	0.01	0.004	0.003
95% C.I.	-0.11 to -0.19	0.90 to 0.94	0.32 to 0.34	0.21 to 0.22
Maximum	0.77	1.37	1.49	0.95
Minimum	-1.04	0.49	-1.11	-0.60
Range	1.81	0.88	2.6	1.55

Different letters after mean elevations indicate significant differences.
N = sample size; SE = standard error; C.I. = confidence interval

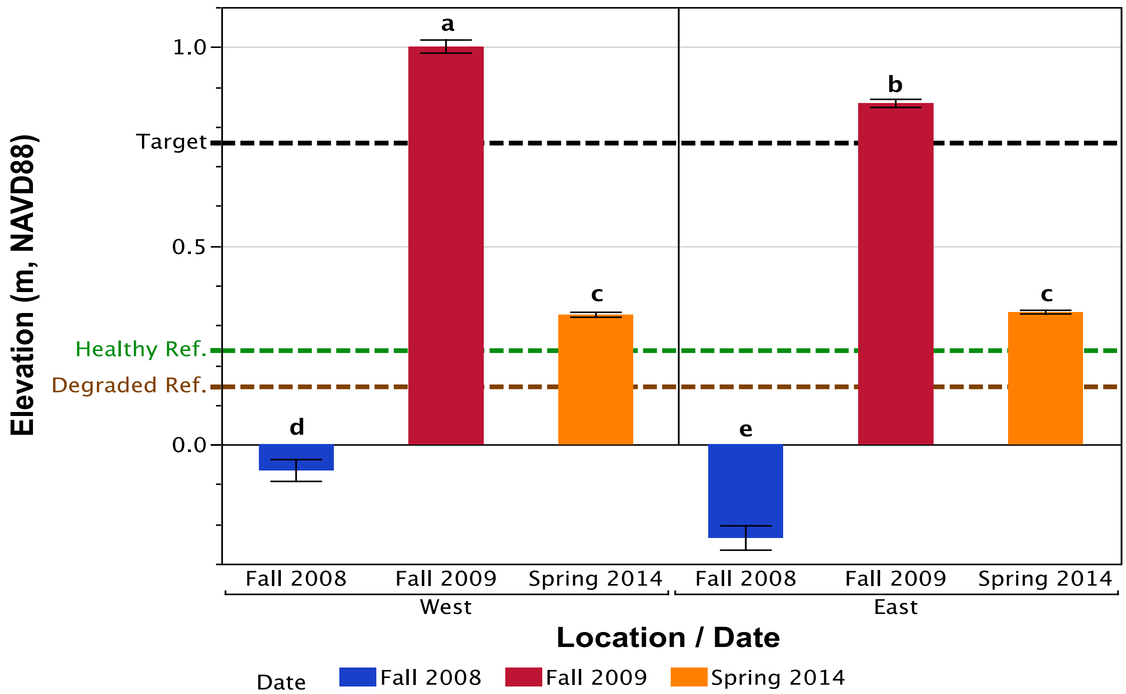


Figure 30 - Confined fill-treatment elevations (m, NAVD88) surveyed in 2008 (pre-construction) and 2009 and 2014 (post-construction) both east and west of the Harvey Cutoff for surveyed transect points shown in Figure 1.5. Means with different letters identify significant differences based on a one-way ANOVA ($P<0.0001$) with six treatment-levels identified on the x-axis (from left to right along the x-axis, n = 238, 95, 1603, 244, 124, and 1679).

On average, sediment-addition increased pre-construction elevations in confined areas by more than a factor of 10. Thereafter, elevation decreased 64% by 2014 (Figure 30). Temporal trends in marsh elevation were similar regardless of location (east versus west of the Harvey Cutoff), although 2008 and 2009 elevations were significantly lower to the east (Figure 30).

The unconfined portion of the project area was surveyed in detail by CPRA-contracted surveyors only once, in 2012. Elevations ranged from a minimum of -2.28 m to a maximum of 1.26 m, averaging 0.22 m (Table 13). The spatial variability in elevation is shown in Figure 29. On average, elevation was statistically higher in the unconfined fill-areas west of Harvey Cutoff compared the eastern portion of the project area (Figure 31). In contrast, no difference in average elevation was identified between the eastern and western portions of the confined treatment-area (Figure 31).

When we combine into one graphic the most recent elevation data for both the confined (2014) and unconfined (2012) portions of the project area, the spatial distribution in marsh elevation is clear (Figure 32). The highest elevations occur in the confined treatment areas, especially on the western side of the confined area west of the Harvey Cutoff and in the southwestern cell east of the Harvey Cutoff (Figure 32). These spatial trends in elevation are quantified in Figure 31 and discussed above.

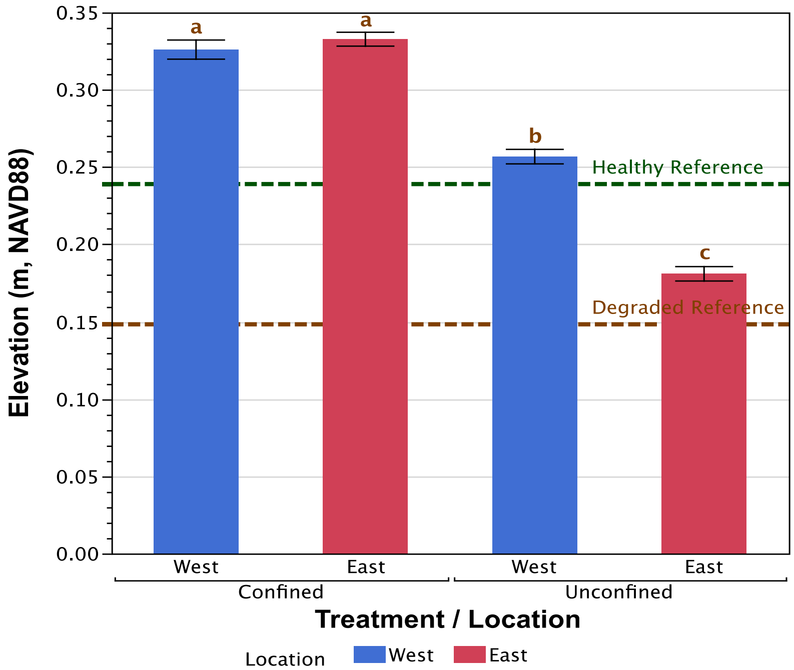


Figure 31 -Comparison of average elevations from transect survey points in confined and unconfined treatments both east and west of the Harvey Cutoff. Means with different letters identify significant differences based on a one-way ANOVA ($P<0.0001$) with four treatment-levels identified on the x-axis (from left to right along the x-axis, n = 1603, 1679. 1237, 1387).

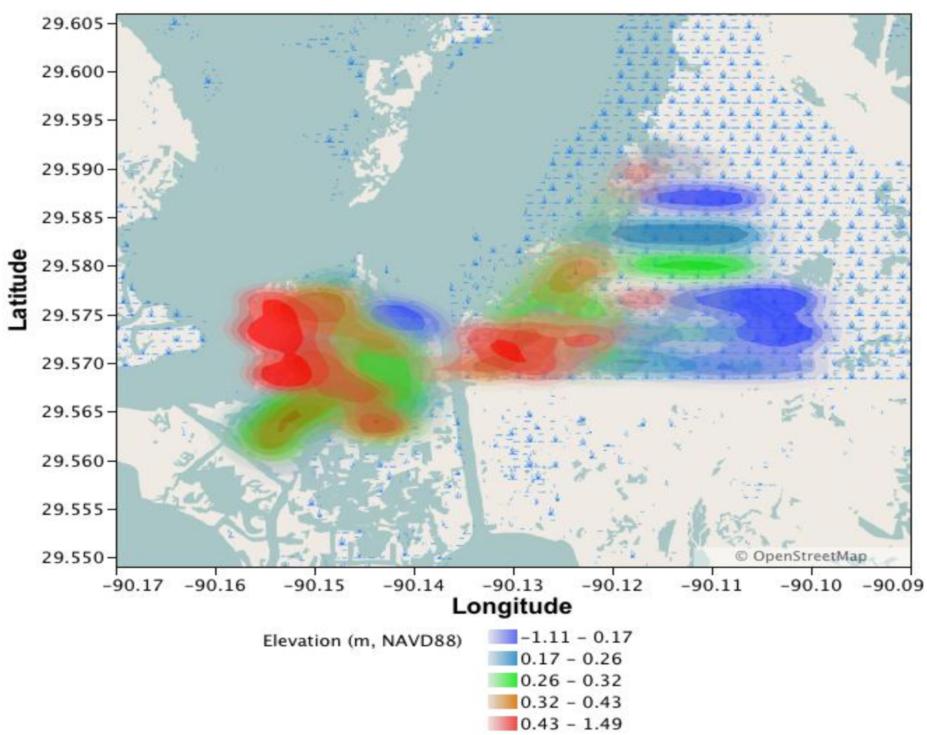


Figure 32 - Bubble plot of transect point elevations (m, NAVD88) surveyed throughout the project area encompassing confined (2014) and unconfined (2012) fill-areas (elevation bubbles were calculated and plotted in JMP Pro 12.1).

Soil Physico-Chemical Status

The components of soil texture varied with restoration treatment. Percent silt and clay were significantly higher for the Confined treatments (High and Medium) than for the two Reference marshes (Degraded and Healthy), while the unconfined treatments tended to be intermediate (Figure 33). Sand content significantly varied with restoration treatment (P=0.08); the Degraded Reference marsh tended to have higher sand than the others treatments (Figure 33). Although the effect of restoration treatment on sand and clay content varied by year (significant treatment x year interaction, Table 14), differences among means were generally small and Tukey's multiple comparison tests identified few significant differences. However, the significant treatment x year interaction for clay and sand identified some general trends (Appendix Figure A-1 and A-2). Clay content tended to decrease from 2011 to 2013 in all restoration treatments, except the Unconfined Very Low, but increased over time in the two reference marshes (Appendix Figure A-1). Sand content had the opposite response, generally increasing with time in the restoration treatments (except the Unconfined Very Low), but decreasing over time in the reference marshes (Appendix A-2). Silt content, although varying significantly by year (decreasing in 2013), did not have a year by treatment interaction (Table 14).

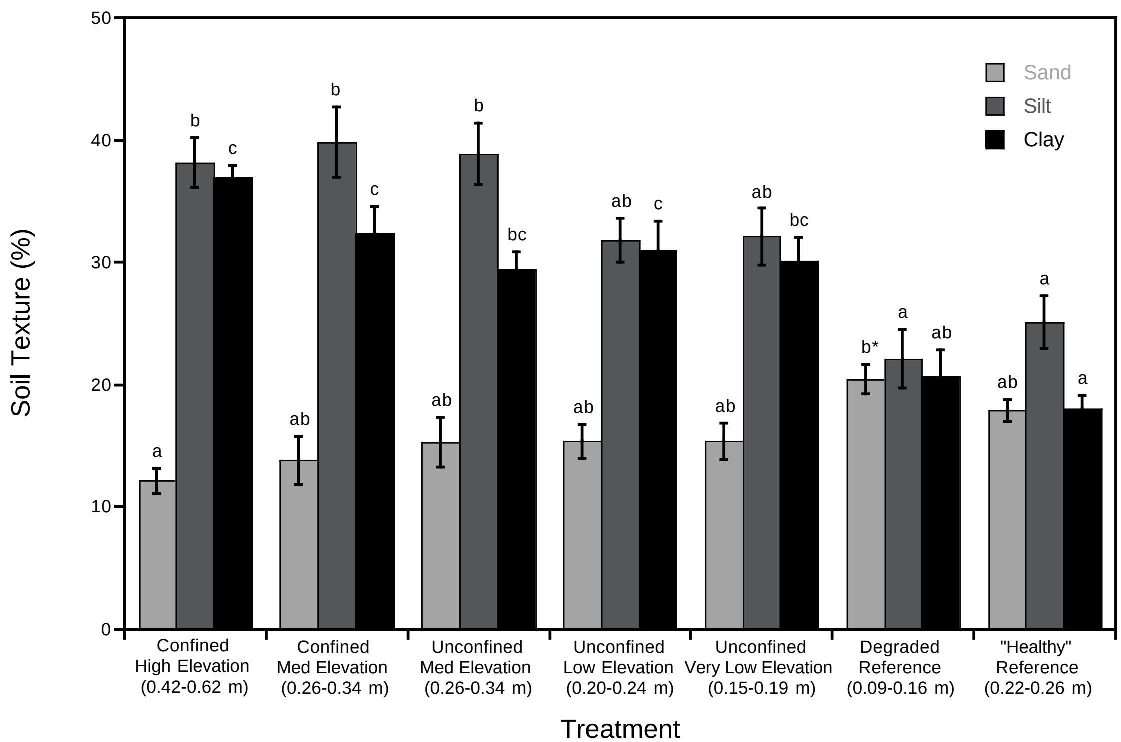


Figure 33 - Treatment effects on soil texture (% sand, silt, and clay) averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Note that the main effect of treatment on % sand was significant at p=0.08. Within a texture class, means with different letters identify significant differences within a response variable.

Table 14 -. ANOVA summary statistics. (Bold indicates p≤0.05; underline indicates p≤0.10)

	ANOVA *P*-values		
Variable	Treatment	Year	Treatment*Year
Aboveground Biomass			
Live	0.55	**0.0004**	0.92
Dead	**0.006**	**0.002**	**0.03**
Total	0.11	**0.0001**	0.37
Belowground Biomass			
Live	0.06	0.57	**0.02**
Dead	**< 0.0001**	**0.04**	0.69
Total	**< 0.0001**	0.36	0.20
Total Biomass (Above- and Belowground)			
Live	0.07	**0.001**	0.39
Dead	**< 0.0001**	0.81	0.09
Total	**< 0.0001**	0.06	0.45
Aboveground Production (Regrowth)			
Live	0.15	**< 0.0001**	0.68
Dead	0.42	0.51	0.99
Total	**0.05**	**< 0.0001**	0.76
Belowground Production (Ingrowth)[1]			
Live	0.74	N/A	N/A
Dead	0.89	N/A	N/A
Total	0.78	N/A	N/A
Plant Community Composition			
Canopy Height	0.12	**0.002**	0.40
Species Richness[2]	**0.001**	**0.03**	0.10
Top 10 Species Biomass[3]			
Factor 1 (SYSU, SYDI, TY)[*]	**0.03**	N/A	N/A
Factor 2 (PAVA, -SPPA)[*]	**0.0008**	N/A	N/A
Factor 3 (SCRO, ECCR)[*]	0.58	N/A	N/A
Decomposition			
Aboveground[4]	**0.05**	N/A	N/A
Belowground[4]	**0.03**	N/A	N/A
Cotton Strips[5]	0.09	**< 0.0001**	0.23

[*]SYSU=*Symphyotrichum subulatum;* SYDI=*Symphyotrichum divaricatum;* TY=*Typha* sp.; PAVA=*Paspalum vaginatum*; SPPA=*Spartina patens*; SCRO=*Schoenoplectus robustus;* ECCR=*Echinochloa crusgalli*.

Table 14 - ANOVA summary statistics (cont.)			
	ANOVA *P*-values		
	Treatment	Year	Treatment*Year
Soil Accretion	**0.04**	0.20	0.26
Soil Physical Properties			
Bulk Density	**< 0.0001**	**0.02**	0.22
Moisture	**< 0.0001**	**0.004**	0.78
Organic Matter	**< 0.0001**	**< 0.0001**	0.61
Mineral Matter	**< 0.0001**	**< 0.0001**	0.70
Sand	0.08	0.10	**0.002**
Silt	**0.0001**	**< 0.0001**	0.90
Clay	**< 0.0001**	0.09	**0.002**
Soil Chemical Properties[6]			
Factor 1 (Mg, Na, Salts, K, Ca, S, Cu)	**0.005**	N/A	N/A
Factor 2 (Eh, P, NO_3-N, Ca, -Conductivity)	**< 0.0001**	N/A	N/A
Factor 3 (Fe, -pH)	0.31	N/A	N/A

Notes:
[I]Incomplete analysis due to missing ingrowth bags
[2]Combined species richness from biomass and cover
[3]Biomass of the top 10 species was time-averaged prior to factor analysis and subsequent ANOVA.
[4]Incomplete analysis due to missing litterbags.
[5]Cotton strip data was depth-averaged (0-24 cm) prior to analysis.
[6]Soil chemical properties were time-averaged prior to factor analysis and subsequent ANOVA.

Soil mineral matter and organic content significantly varied with restoration treatment (Table 14). Organic matter was lowest at high elevations (Confined High and Medium and Unconfined Medium) and highest in the reference marshes; Unconfined Low and Very Low treatments were intermediate (Figure 34). Mineral matter had the opposite trend (Figure 34). Mineral and organic content significantly varied with sampling year (decreasing somewhat in 2013 for mineral matter and increasing in 2013 for organic matter, Appendix A-3). The effect of year on these variables did not differ by restoration treatment (non-significant treatment x year interaction, Table 14). Soil bulk density, which followed a similar trend as soil mineral content, was highest in the restoration treatments (~0.3 to 0.5 g cm^{-3}) and lowest in the reference marshes (~0.1 g cm^{-3}) (Figure 35). Soil bulk density was significantly lower in 2013 compared to 2011 (Appendix A-4), as evidenced by soil mineral content (Appendix A-3). The interaction between treatment and year was not significant (Table 14). Soil moisture (Figure 36) showed a similar trend to that of soil organic matter with highest values in the reference marshes (lowest elevations) and lowest values in the Confined High and Medium treatments (highest elevations); unconfined restoration treatments with intermediate elevations had intermediate soil moistures (Figure 36). Soil moisture significantly increased from 2011/2012 to 2013 (Table 14, Appendix A-5).

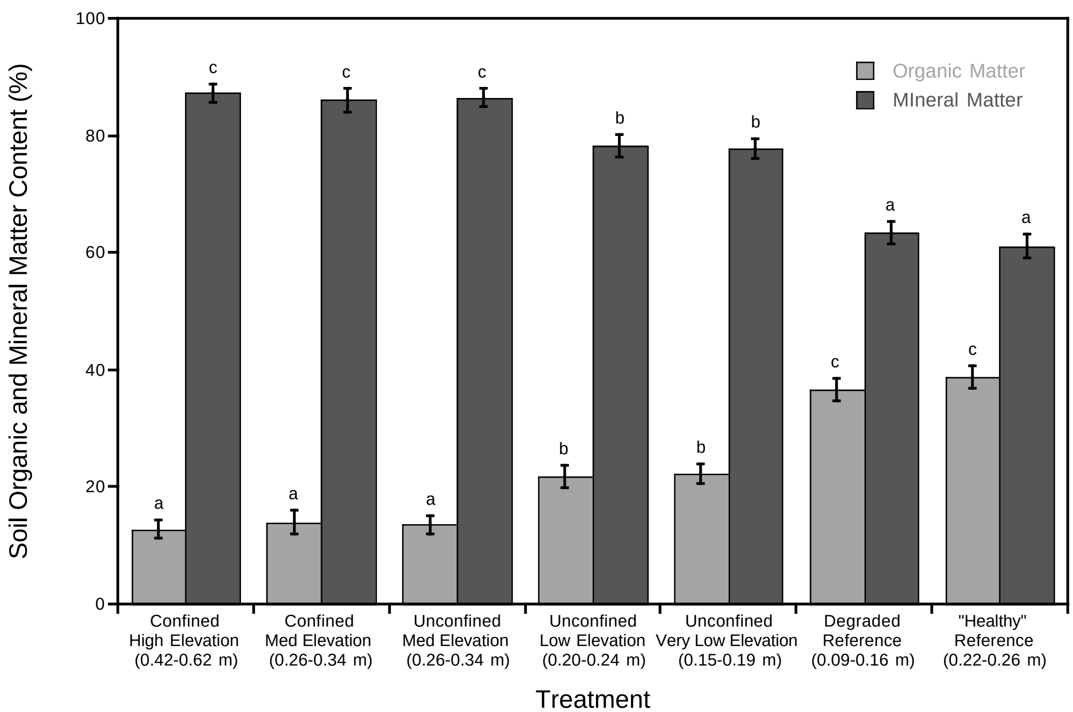


Figure 34 - Treatment effects on soil organic and mineral matter averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Means with different letters identify significant differences within a response variable.

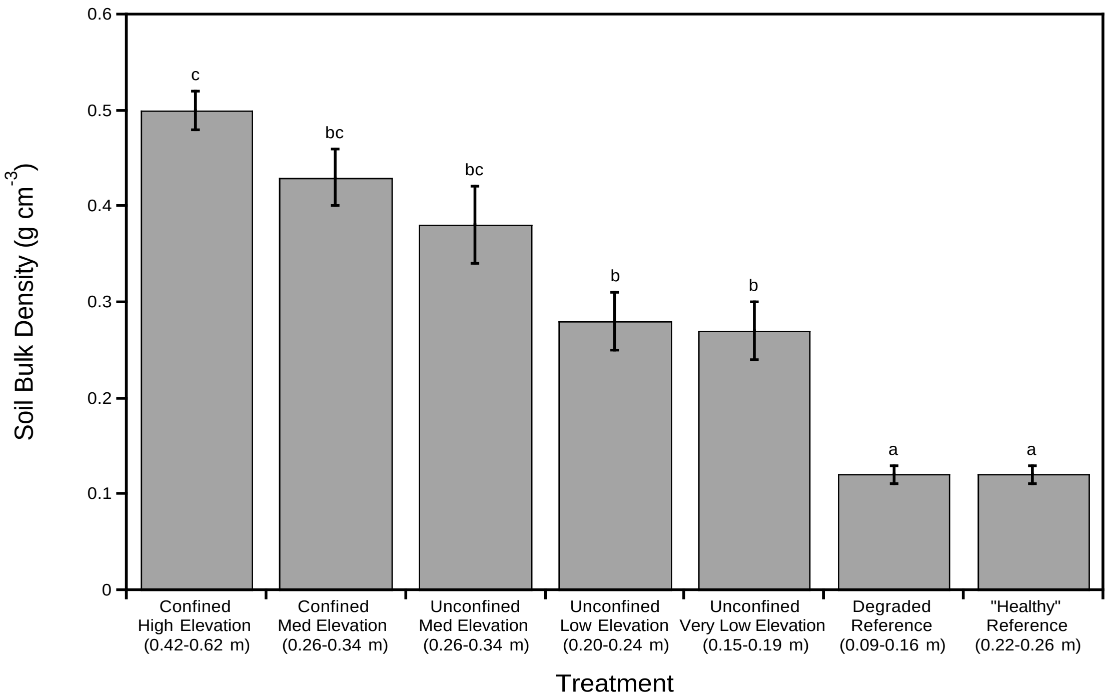


Figure 35 - Treatment effects on soil bulk density averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Means with different letters identify significant differences.

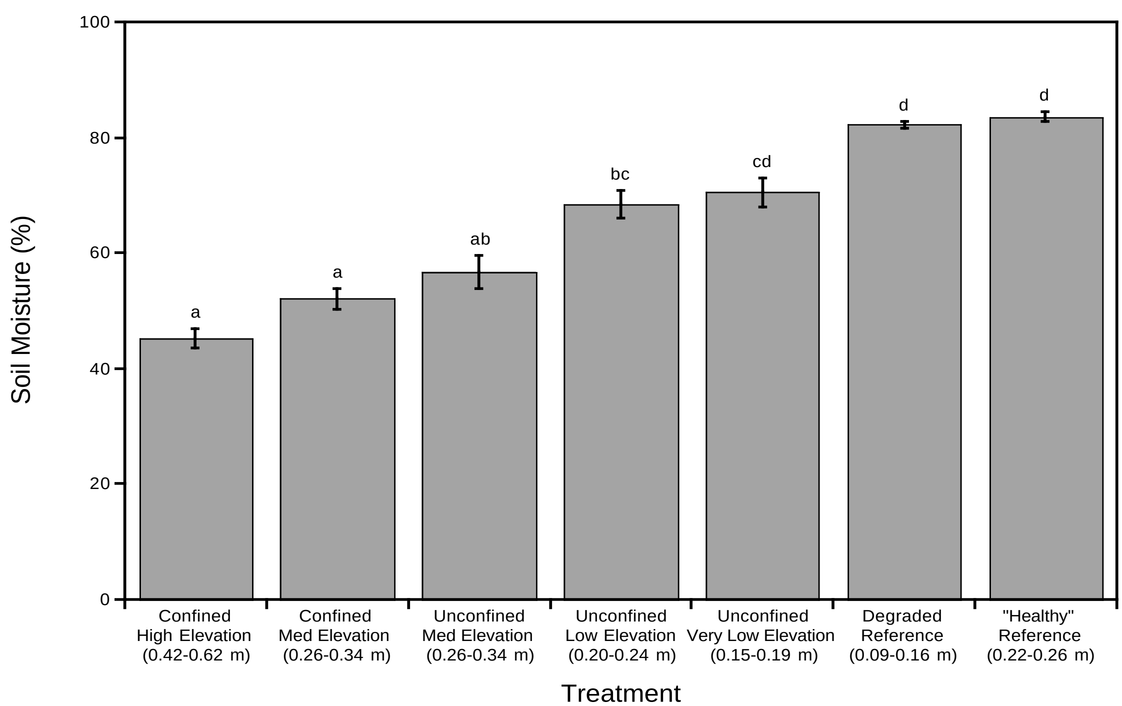


Figure 36 - Treatment effects on soil moisture averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Means with different letters identify significant differences.

The chemical status of the soils (Table 15) generally reflected the differences in soil physical characteristics and marsh elevation among the restoration and reference sites. For example, soil redox potential (Eh) was generally greater at higher elevations and lower in the reference marshes where both low elevation and high organic matter content likely generated more biochemically-reduced conditions. Surprisingly, pH tended to be similar across all treatment sites and varied from 4.8 to 5.5 on average. We expected that drier soils at higher elevations would have lower pHs because of metal-sulfide oxidation to sulfate. This process was not reflected in our data. Soil conductivity, a measure of salinity, was highest in the reference marshes and lowest in the Confined High and Medium treatments; restoration sites at intermediate elevations had intermediate conductivities. In contrast, salt content of the soil, expressed on a volumetric basis, was lowest in the reference marshes, which had almost 50% less salt per unit soil volume than the other restoration treatment-sites. The higher soil moistures (Figure 36) and lower bulk densities (Figure 35) of the reference marshes were likely responsible for this finding.

Soil available nitrogen (ammonium and nitrate) and phosphorus are the most important limiting nutrients for plant growth. As in most flooded soils, where denitrification converts nitrate to nitrogen gas, nitrate concentrations were very low, except for the highest elevation site (Confined High treatment) (Table 15). This treatment site was the most oxidized and therefore would likely have relatively high rates of nitrification (aerobic conversion of ammonium to nitrate) and relatively low rates of denitrification. Soil ammonium concentration was generally an order of magnitude, or more, greater than soil nitrate (Table 15), and as a result, soil ammonium must play the more important role in the nitrogen nutrition of the vegetation (Mendelssohn 1979). With the exception of the Unconfined Low and Very Low treatments, soil ammonium was higher in the restoration sites compared to the reference marshes (Table 15). Also, the Healthy Reference marsh tended to have more soil ammonium than the Degraded Reference marsh.

Although soil phosphorus is rarely limiting plant growth in saline wetlands, it can become limiting in dredged sediments where sand content is high. However, the dredged sediments for the Barataria Landbridge Restoration Project were relatively low in sand content based on the soil texture data (Figure 33). In fact, soil phosphorus was generally higher in the restoration sites than in the reference marshes (Table 15). Hence, we would not expect phosphorus to limit plant growth here.

Elements such as magnesium, potassium, calcium, sulfur, iron, manganese and zinc, which are all essential plant nutrients, are in high concentrations in the restoration sites relative to the reference marshes (Table 15), and are not likely limiting plant growth due to deficiencies beyond what might occur naturally in the reference marshes (Broome et al. 1975a and b). In their reduced biogeochemical states, sulfur, iron and/or manganese can become toxic to vegetation (Gambrell and Patrick 1978). However, given the relatively high elevations and Eh levels within the restoration treatment-areas (Figure 25, Table 15), it is unlikely that soluble metal concentrations would approach toxicity. Also, given the high mineral content of the soil in the restoration sites, it is probable that any toxic free sulfide, generated from the anaerobic reduction of sulfate to sulfide, would be bound to iron and manganese, reducing the availability of these potentially toxic elements (Gambrell and Patrick 1978; Figure 5 in Mendelssohn and Kuhn 2003).

Table 15 - Soil chemical variables (0-15 cm soil depth). Values presented as the mean ± 1 SE (n=15) for each treatment averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Chemical concentrations are presented on a volume basis. Probabilities in bold indicate significant treatment main effects.

Variables	Confined High Elevation (0.42-0.62 m)	Confined Med Elevation (0.26-0.34 m)	Unconfined Med Elevation (0.26-0.34)	Unconfined Low Elevation (0.20-0.24)	Unconfined Very Low Elevation (0.15-0.19)	Degraded Reference (0.09-0.16 m)	"Healthy" Reference (0.22-0.26)	Probability of Significant Treatment Effect
Factor 1 Variables								
Salts (ug/cm3)	4444 ± 706	4952 ± 407	5343 ± 286	5114 ± 998	5577 ± 936	3510 ± 271	2784 ± 275	**0.004**
Calcium (ug/cm3)	100.44 ± 12.08	111.68 ± 8.12	95.05 ± 6.62	70.50 ± 10.98	73.27 ± 9.94	36.09 ± 2.69	48.64 ± 3.07	**<0.0001**
Potassium (ug/cm3)	18.73 ± 2.44	24.11 ± 0.69	19.63 ± 1.31	22.61 ± 2.43	22.53 ± 3.72	14.02 ± 1.63	12.83 ± 1.01	**0.0009**
Magnesium (ug/cm3)	54.17 ± 7.22	68.08 ± 3.25	64.44 ± 4.47	61.23 ± 5.42	67.16 ± 7.38	38.63 ± 3.39	42.66 ± 2.89	**<0.0001**
Sodium (ug/cm3)	91.04 ± 17.52	121.73 ±19.50	134.30 ± 10.59	133.14 ± 15.15	140.95 ± 18.12	85.37 ± 5.66	75.98 ± 4.07	**0.002**
Sulfur (ug/cm3)	31.29 ± 5.95	45.78 ± 6.63	44.27 ± 4.11	27.98 ± 3.77	39.53 ± 3.84	23.82 ± 1.98	30.46 ± 2.04	**0.025**
Copper-DTPA (ug/cm3)	0.51 ± 0.15	0.74 ± 0.18	0.39 ± 0.07	0.52 ± 0.14	0.38 ± 0.06	0.18 ± 0.02	0.21 ± 0.02	**0.01**
Manganese-DTPA (ug/cm3)	7.19 ± 1.75	8.89 ± 1.72	8.26 ± 1.94	5.45 ± 3.19	4.05 ± 1.54	0.49 ± 0.09	1.65 ± 0.51	**<0.0001**
Zinc-DTPA (ug/cm3)	1.01 ± 0.20	1.09 ± 0.19	0.88 ± 0.13	0.82 ± 0.16	0.82 ±0.08	0.39 ± 0.05	0.67 ± 0.06	**0.05**
Factor 2 Variables								
Eh@2cm (mV)	319 ± 14	186 ± 37	237 ± 22	210 ± 16	166 ± 21	117 ± 23	156 ± 10	**<0.0001**
Eh@15cm (mV)	288 ± 16	193 ± 76	199 ± 17	139 ± 20	130 ± 20	75 ± 11	90 ± 12	**<0.0001**
Nitrate-N (ug/cm3)	0.21 ± 0.13	0.04 ± 0.03	0.06 ± 0.05	0.00 ± 0.00	0.01 ± 0.00	0.01 ± 0.01	0.02 ± 0.01	**0.008**
Phosphorus (ug/cm3)	0.89 ± 0.18	1.04 ± 0.13	0.64 ± 0.08	0.73 ± 0.08	0.60 ± 0.13	0.32 ± 0.07	0.29 ± 0.03	**<0.0001**
Conductivity (mS)	7200 ±776	9426 ± 1159	11934 ± 1572	14275 ± 1926	16514 ± 2169	22686 ± 2146	18380 ± 1257	**<0.0001**
Factor 3 Variables								
Iron-DTPA (ug/cm3)	75.46 ± 15.25	78.60 ± 11.75	105.50 ± 13.09	95.58 ± 9.93	81.59 ± 10.86	40.23 ± 3.40	68.41 ± 8.45	**0.0007**
Ammonium-N (ug/cm3)	3.86 ± 2.60	2.42 ± 0.73	4.10 ± 2.21	0.70 ± 0.13	1.16 ± 0.45	0.98 ± 0.14	2.04 ± 0.52	**0.04**
pH (1:1 Water)	5.1 ± 0.5	5.5 ± 0.2	4.9 ± 0.3	5.3 ± 0.2	5.2 ± 0.2	5.3 ± 0.1	4.8 ± 0.1	0.52

We used a factor analysis to reduce the 17 chemical variables into three principal factors (Table 16). Each factor is a linear combination of those chemical variables most highly correlated with that particular factor. This analysis produced three factors with eigenvalues greater than 1.0. Factor 1 had high (> 0.6) positive correlations with Mg, Na, salts, K, Ca, S and Cu, and was interpreted as a soil salts related factor. Factor 2 had high positive correlations with Eh, P, NO_3-N, and Ca and a weak negative correlation with soil conductivity, and was interpreted as a soil reduction (waterlogging) factor. Factor 3 had a high positive correlation with Fe and a high negative correlation with pH, and was interpreted as a pH factor (typically, as pH decreases, Fe availability increases).

We then analyzed the factor scores, averaged over sampling date, with ANOVA (Figure 37). The restoration treatments significantly affected both Factors 1 and 2 (Table 14), which explained 51% and 19%, respectively, of the variation in the chemical data. Factor 1, the soil-salts factor, was highest in the restoration treatments, both Confined and Unconfined, with one exception – the Confined, High treatment (Figure 37). Here, precipitation may have leached some of the salts that accumulated from evapo-transpiration. In contrast, the lowest Factor 1 scores were for the reference marshes (Figure 37), where the soil stays saturated and concentration of salts due to evaporation is low. Also, the lower volumetric-based salt levels in the reference marshes compared to the treatment sites was likely due to the difference in bulk density between treatment sites and reference marshes; i.e., the reference marshes had lower mineral matter per unit volume and hence less salt than the treatment sites with higher bulk density.

Factor 2, the soil oxidation factor, was positively related to soil Eh, i.e., the higher the soil Eh (more oxidized) the higher score for this factor. Hence, Factor 2 scores were greatest for the restoration sites (Figure 37), which were higher in elevation and less flooded, compared to the reference marshes, which were lower in elevation and more flooded. In fact, this factor decreased in an almost linear fashion with decreasing elevation (Figure 37). Factor 3 did not significantly differ among the different restoration treatments (Table 14).

Vegetation Structural Responses

Plant Species Composition

A total of 30 different species were found in the restoration treatments and reference marshes (Table 17). The overwhelmingly dominant plant species, based on biomass, in the reference marshes, both Healthy and Degraded, was *Spartina patens*. Of secondary importance were *Schoenoplectus americanus* in both reference marshes and *Distichlis spicata* in the Degraded Reference marsh (Table 17). In the restoration treatment sites, *Paspalum vaginatum* and *S. patens* were the most dominant, with *Schoenoplectus* spp., *Symphyotrichum* spp., and *Typha* of lesser dominance, but still prevalent (Table 17).

Table 16 – Factor analysis of time-averaged soil chemical variables. Indicated variables corresponding to bolded correlation coefficients (≥±0.6) define the factor.

Rotated Factor Pattern (Correlations)			
Variable	**Factor1**	**Factor2**	**Factor3**
Magnesium	**0.8976**	0.25926	-0.00192
Sodium	**0.87914**	-0.10488	-0.03093
Total Salts	**0.87675**	0.14412	-0.04404
Potassium	**0.84462**	0.24296	-0.12098
Calcium	**0.69405**	**0.62773**	0.08227
Sulfur	**0.66964**	0.04263	0.41473
Copper	**0.60903**	0.51008	-0.33334
Zinc	0.56765	0.49545	-0.17365
Eh @ 15 cm	0.25712	**0.81768**	0.29956
Eh @ 2 cm	0.05436	**0.74684**	0.28923
Phosphorus	0.38953	**0.68731**	-0.13023
Nitrate-N	-0.36192	**0.6575**	-0.24338
Manganese	0.43441	0.552	0.36373
Conductivity	-0.17446	**-0.83332**	-0.22479
Iron	0.30116	0.27091	**0.72256**
Ammonium-N	-0.09887	0.188	0.55895
pH	0.33339	0.17218	**-0.85518**
Eigenvalue	7.31	2.80	2.02
Variance Explained	51%	19%	14%

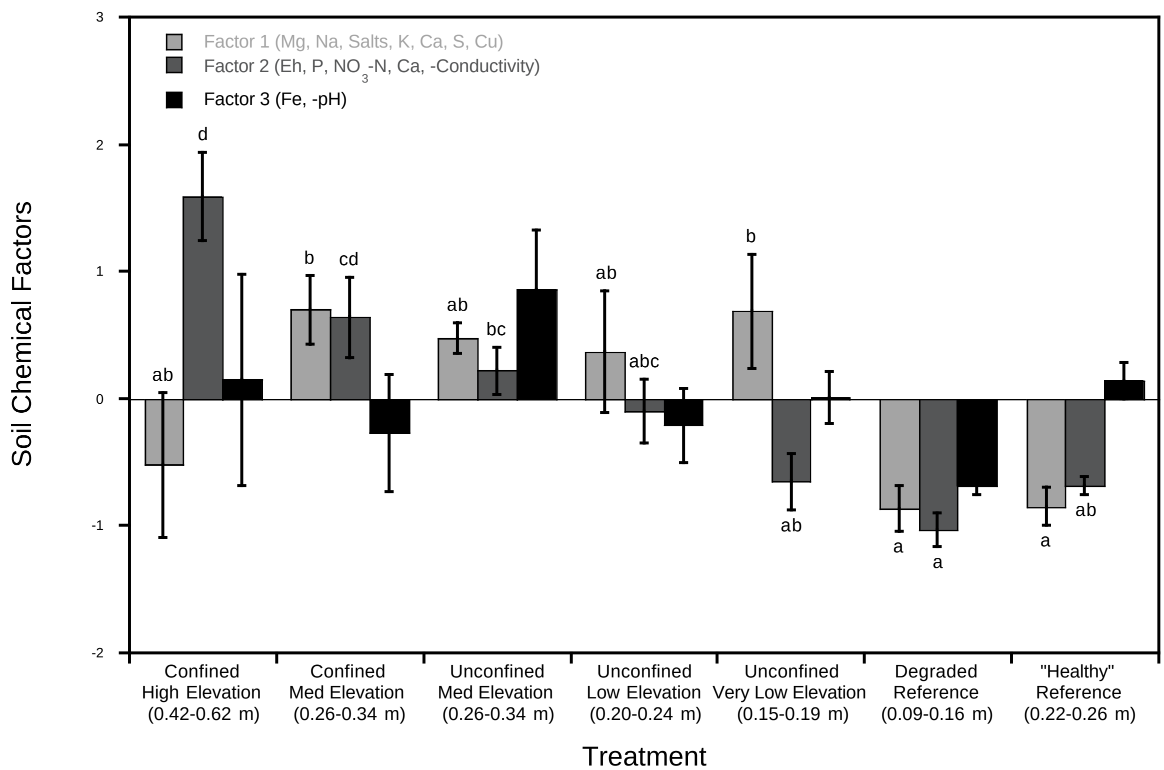


Figure 37 - Treatment effects on principal component factors of the chemical variables averaged over Fall 2011, Fall 2012, and Fall 2013. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Although treatment did not significantly affect Factor 3 (hence, the absence of letters associated with Factor 3 means), the scores for this factor are graphed for descriptive purposes. Means with different letters identify significant differences within response factors.

Table 17 - Plant community composition[1], species-specific and total biomass, canopy height, and species richness averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (n=15) for each treatment.

	Treatment						
Plant Biomass (g m^{-2})	**Confined High Elevation (0.42-0.62 m)**	**Confined Med Elevation (0.26-0.34 m)**	**Unconfined Med Elevation (0.26-0.34 m)**	**Unconfined Low Elevation (0.20-0.24 m)**	**Unconfined Very Low Elevation (0.15-0.19 m)**	**Degraded Reference (0.09-0.16 m)**	**"Healthy" Reference (0.22-0.26 m)**
Alternanthera philoxeroides				6.2± 5.9			
Amaranthus australis				1.6± 1.6			
Baccharis halimifolia	70.5 ± 52.4						1.3± 1.3
Bacopa monnieri	21.8 ± 16.5		18.5± 12.7	3.8 ± 3.8	3.3± 3.0		
Cyperus filicinus			0.1± 0.1	0.4 ± 0.4			
Cyperus odoratus		1.1± 1.1					9.0± 6.1
Distichlis spicata	1.8± 1.8		1.0± 1.0	121.6 ± 81.2	11.6± 8.3	163.0 ± 53.7	2.7± 2.7
Echinochloa crusgalli	10.4± 8.2		3.0± 2.3	7.0 ± 7.0	5.0± 4.8		
Eleocharis cellulosa					52.5± 31.2		26.2 ± 14.4
Eleocharis fallax							28.6 ± 13.9
Eleocharis parvula			0.1± 0.1				
Fimbristylis castanea							1.0 ± 1.0
Hydrocotyle bonariensis							0.1 ± 0.1
Ipomoea sagittata		0.3± 0.2		7.4 ± 3.4			24.0 ± 4.9
Juncus roemerianus			48.9± 35.3				
Lythrum lineare	0.1± 0.1			0.3 ± 0.2	0.7± 0.7		4.5 ± 3.7
Paspalum vaginatum	57.5 ± 40.0	1097 ± 395	813.9± 261.0	134.8 ± 90.7	621.8± 224.1		
Phragmites australis					0.4± 0.4		
Pluchea camphorata	7.9 ± 7.6		6.4± 6.4				
Polygonum punctatum	2.2 ± 1.7		0.3± 0.3		5.0± 5.0		0.2 ± 0.2
Sagittaria lancifolia							48.8 ± 18.9
Schoenoplectus americanus	0.8 ± 0.8	461.7 ± 145.2	67.1± 39.4	431.5 ± 98.1	291.1± 98.2	307.8 ± 68.7	176.7 ± 55.8
Schoenoplectus robustus	138.1 ± 39.2	0.1 ± 0.1	53.8± 35.0	63.6 ± 29.3	144.2± 84.7		

Table 17 (cont.)

Plant Biomass (g m^{-2})	**Treatment** **Confined High Elevation (0.42-0.62 m)**	**Confined Med Elevation (0.26-0.34 m)**	**Unconfined Med Elevation (0.26-0.34 m)**	**Unconfined Low Elevation (0.20-0.24 m)**	**Unconfined Very Low Elevation (0.15-0.19 m)**	**Degraded Reference (0.09-0.16 m)**	**"Healthy" Reference (0.22-0.26 m)**
Solidago sempervirens	24.1 ± 24.0						
Spartina alterniflora				4.6 ± 4.6			
Spartina patens	353.9 ± 235.0	191.9 ± 114.7	667.0± 227.2	737.4 ± 185.1	190.3± 97.6	1320 ± 187	1837 ± 355
Symphyotrichum divaricatum	64.7 ± 52.0		33.7± 33.7	13.6 ± 6.3	2.7± 2.1		7.1 ± 4.9
Symphyotrichum subulatum	119.9 ± 60.3		42.0± 24.1	1.9 ± 1.5	0.1± 0.1	0.8 ± 0.8	1.5 ± 1.0
Typha sp.	177.6 ± 99.8		48.8± 29.3				
Vigna luteola							1.4 ± 1.0
Total Biomass	1058 ± 206	1752 ± 287	1808± 204	1536 ± 180	1329± 178	1792 ± 177	2170 ± 332
Canopy Height (cm)	107.9 ± 12.6	79.8 ± 6.4	82.0± 5.8	78.0 ± 5.4	84.0± 6.6	96.2 ± 5.3	99.1 ± 5.7
Total Species Richness (#)	15	6	15	15	13	4	16
Average Species Richness	4.2 ± 0.5	2.2 ± 0.2	3.3± 0.6	4.7 ± 0.5	3.1± 0.3	2.7 ± 0.2	6.0 ± 0.4

[1]**The plant species listed are those found in the biomass plots at the 35 permanent stations and are not meant to be inclusive of all species in the project areas. Additional species observed but not recorded at the 35 permanent stations are presented in the *2010 Spatial Assessment* discussion, page 20 of this report.**

The Healthy Reference marsh had a total of 16 different species compared to 4 species in the Degraded Reference marsh. Average species richness and total species richness were 2x and 4x greater, respectively, in the Healthy Reference marsh compared to the Degraded Reference marsh (Table 17). For the restoration sites, *P. vaginatum*, *S. patens*, and *S. americanus* were most important, based on biomass. For example, *P. vaginatum* had the highest biomass of any species in three of the five restoration sites. The Confined High Elevation treatment was unique in containing relatively high biomass of the woody composites (family Asteraceae) *Symphyotrichum divaricatum and S. subulatum* as well as the composite shrub, *Baccharis halimifolia*. *Spartina patens* was also prevalent in the Confined High elevation site. Of the 30 total species found in the restoration and reference sites, 13 species had vegetative biomasses below 10 g m^{-2} (Table 17). Hence, many of the plant species occurring in the study area were relatively minor; only three species dominated (*S. patens*, *P. vaginatum*, and *S. americanus*) and three species co-dominated (*S. robustus*, *D. spicata*, and the *Symphyotrichum* species).

In total, the restoration sites had more species (25) than the reference marshes (16) (Table 17). Five species were unique to the reference marshes, i.e., not found in the restored sites; these were *Eleocharis fallax*, *Fimbristylis castanea*, *Hydrocotyle bonariensis*, *Vigna luteola*, and *Sagittaria lancifolia.* In contrast, 14 species were found only in the restoration sites and not in the reference marshes. The heterogeneity in elevation, and resulting differences in soil moisture (Figure 36), and the initially barren nature of the restoration sites allowed for a diverse array of species to recruit into the restoration sites, although the more intense sampling within the restoration sites (25 stations) compared to the reference marshes (10 stations) may have contributed to this trend. Regardless, the total number of species was slightly higher in the Healthy Reference marsh (16) than in four of the five restoration treatments (ranging from 13-15 species) (Table 17). The exception was in the Confined Medium treatment, which had only 6 species, possibly because of the overwhelming dominance of *P. vaginatum* in this particular restoration site. The Degraded Reference marsh had the lowest total number of species (4), and much lower than the Healthy Reference marsh (16), likely because of its lower elevation (Figure 25) and greater inundation (Figure 24), than the Healthy Reference marsh. Somewhat higher soil conductivity and total salts, in combination with lower soil surface Eh, may have also contributed to this effect (Table 15).

To determine if species associations significantly differed among restoration treatments and reference marshes, a factor analysis of species biomasses was first conducted, as previously done for the chemical data. The analysis used the 10 most dominant species, all of which occurred at a frequency of six or greater and had a summed total biomass of >100 g m^{-2} (Figure 38). Factor analysis identified three factors with eigenvalues > 1 that explained a total of 80% of the variation in the species biomass data (Table 18). Vegetation-association Factor 1 explained 40% of the variation with high positive correlations with *S. subulatus* (0.87), *S. divaricatum* (0.58), and *Typha* sp. (0.51). Vegetation-association Factor 2 explained 22% of the variation in the species biomass data with a high positive correlation with *P. vaginatum* (0.80) and a high negative correlation with *S. patens* (-0.67). Vegetation-association Factor 3 explained 18% of the variation in the species biomass with high positive correlations with *S. robustus* (0.78) and *Echinochloa crusgalli* (0.77). The restoration treatments significantly affected vegetation-association Factors 1 and 2 (Table 14). The Factor 1 vegetation-association was most prevalent in the Confined High treatment with elevations ranging from 0.42 to 0.62 m (Figure 38). In

Table 18 – Factor analysis of time-averaged 10 most dominant plant species. Indicated variables corresponding to bolded correlation coefficients (≥±0.5) define the factor.

Rotated Factor Pattern (Correlations)

Plant Species	**Factor1**	**Factor2**	**Factor3**
Symphyotrichum subulatum	**0.8680**	0.1369	0.2502
Symphyotrichum divaricatum	**0.5784**	0.0979	-0.1515
Typha sp.	**0.5093**	0.0803	0.1191
Paspalum vaginatum	-0.2952	**0.7977**	-0.0946
Distichlis spicata	-0.1106	-0.2503	-0.0359
Schoenoplectus americanus	-0.1484	-0.3719	-0.3293
Ipomoea sagittata	-0.1108	-0.4113	-0.1329
Spartina patens	-0.2323	**-0.6660**	-0.1783
Schoenoplectus robustus	0.0031	0.0763	**0.7756**
Echinochloa crusgalli	0.0940	0.1561	**0.7679**
Eigenvalue	2.25	1.25	1.01
Variance Explained	39.65%	22.07%	17.81%

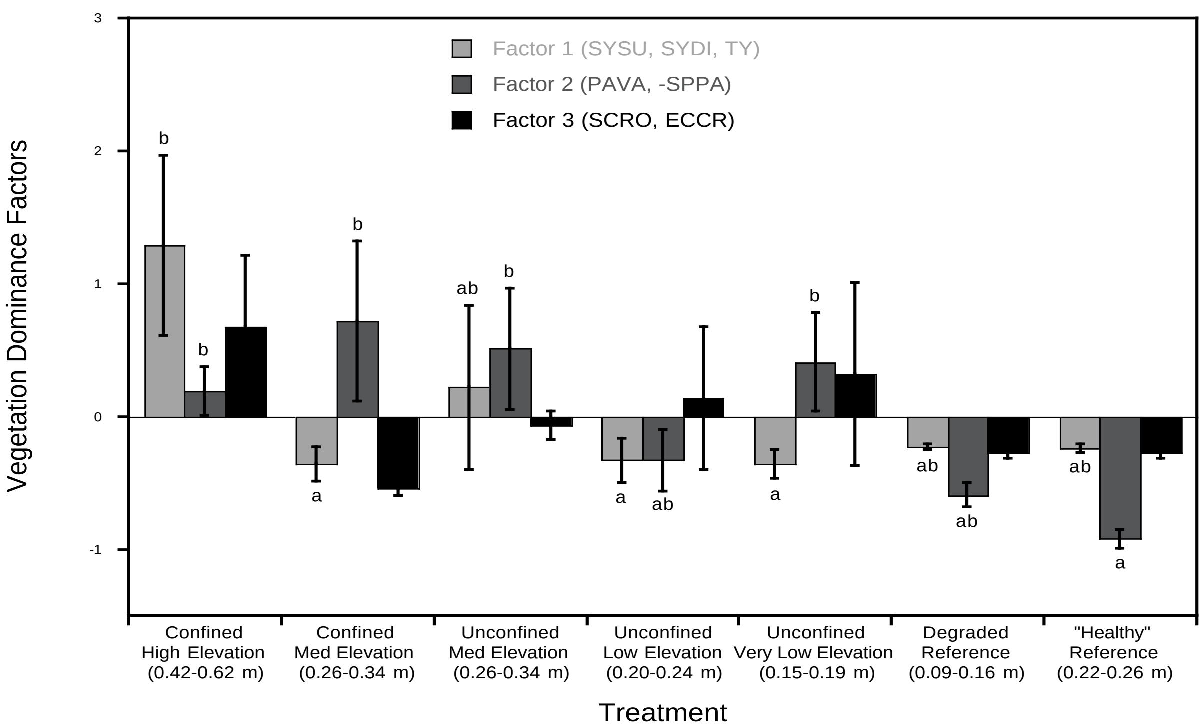


Figure 38 - Effect of treatment on factor scores for plant species dominance (based on biomass) averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (n=15) (error bars) for each treatment. Although treatment did not significantly affect Factor 3 (hence, the absence of letters associated with Factor 3 means), the scores for this factor are graphed for comparative purposes. Means with different letters identify significant differences within a factor. SYSU= *Symphyotrichum subulatum;* SYDI= *Symphyotrichum divaricatum;* TY=*Typha* sp.; PAVA=*Paspalum vaginatum*; SPPA=*Spartina patens*; SCRO= *Schoenoplectus robustus;* ECCR=*Echinochloa crusgalli*.

contrast, the remaining restoration treatments and reference marshes contained less of this vegetation-association (Figure 38). The Factor 2 vegetation-association was dominated by two species that were inversely related to each other, i.e., as *P. vaginatum* biomass increased, *S. patens* biomass decreased. *Paspalum vaginatum* was more prevalent in the restoration treatment sites, while *S. patens* was more prevalent in the reference marshes. The effect of treatment was not significant for Factor 3 (Table 14).

Average Plant Species Richness

Average species richness, the average number of species per restoration or reference site, was significantly affected by treatment (Table 14, Figure 39). Species richness was highest in the Healthy Reference marsh, the habitat-condition that the sediment-restoration attempts to mimic. Species richness in the Degraded Reference marsh was significantly lower than for the Healthy Reference marsh, but statistically equivalent to that in the restoration treatments (Figure 39). However, species richness of the Unconfined Low Elevation Treatment was statistically similar to that of the Healthy Reference Marsh, as was the Confined High Elevation treatment. The relatively high species richness of the Confined High Elevation site was due to recruitment of species not found at the lower elevation Healthy Reference marsh. Species richness significantly changed with time (Table 14) with values somewhat lower in 2012 (3.3 ± 0.3) compared to 2011 (3.9 ± 0.3) and 2013 (4.0 ± 0.4), which were not significantly different.

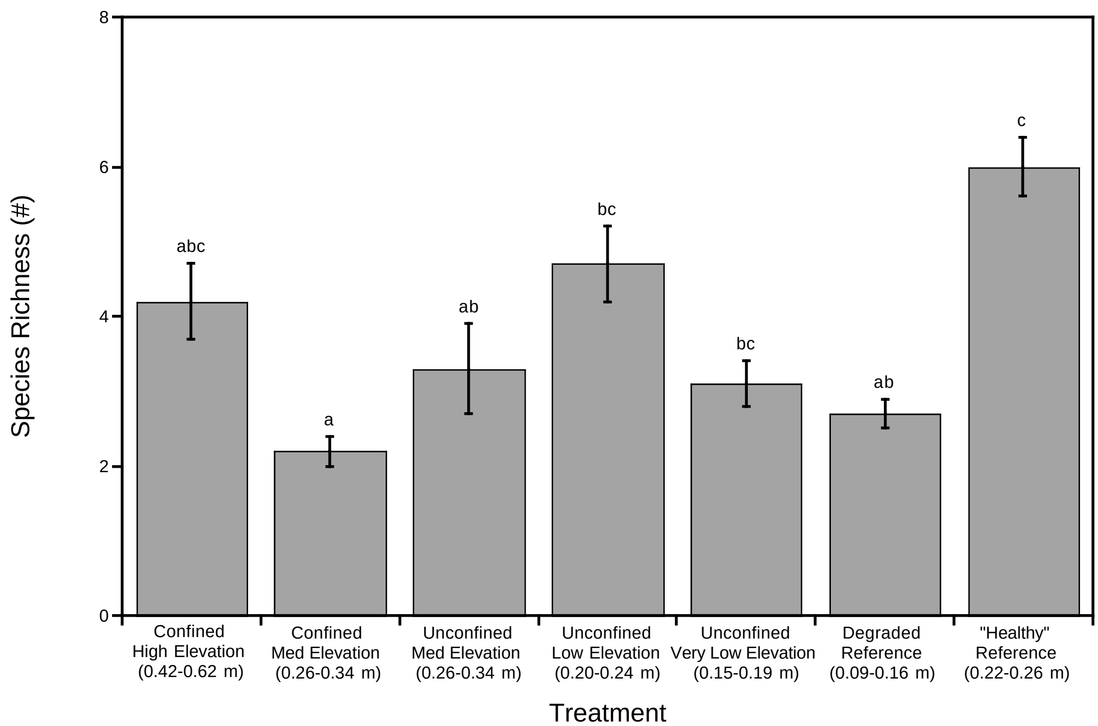


Figure 39 – Average species richness by treatment averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment (note that significant differences are at $P \leq 0.10$). Means with different letters identify significant differences.

Total (Above- and Belowground) Plant Biomass

Live + dead total biomass (Figure 40) significantly varied with restoration treatment (Table 14). The reference marshes had significantly greater total biomass than the Confined High and Unconfined Medium Elevation-treatments, and tended to be greater than the other restoration treatments, with the exception of the Unconfined Low Elevation, which was similar to that of the reference marshes. Average total biomass ranged from as low as 1664 g m^{-2} to a high of about 5035 g m^{-2} (Figure 40), and varied significantly with year (Appendix A-6). Overall, the reference marshes had the greatest total biomass (with the exception of the Unconfined Low Elevation treatment) while the highest elevation restoration site (Confined High Elevation) had the lowest total biomass, indicating poor recovery in the latter. The other restoration sites were intermediate in recovery, based on total biomass, with the exception of the Unconfined Low Elevation treatment, which was equivalent in total biomass to the reference marshes (Figure 40).

Live total biomass and dead total biomass also significantly differed with treatment (Figure 40, Table 14), although the former at P=0.07. Trends in the dead total biomass with treatment were similar to live + dead total biomass. In contrast, live total biomass did not significantly differ among treatments with the exception of the Confined High Elevation site, where live total biomass tended to be lower than for the other treatments (Figure 40). Live total biomass in 2012 was significantly lower than in 2011 and 2013 by ~28% (Appendix A-6).

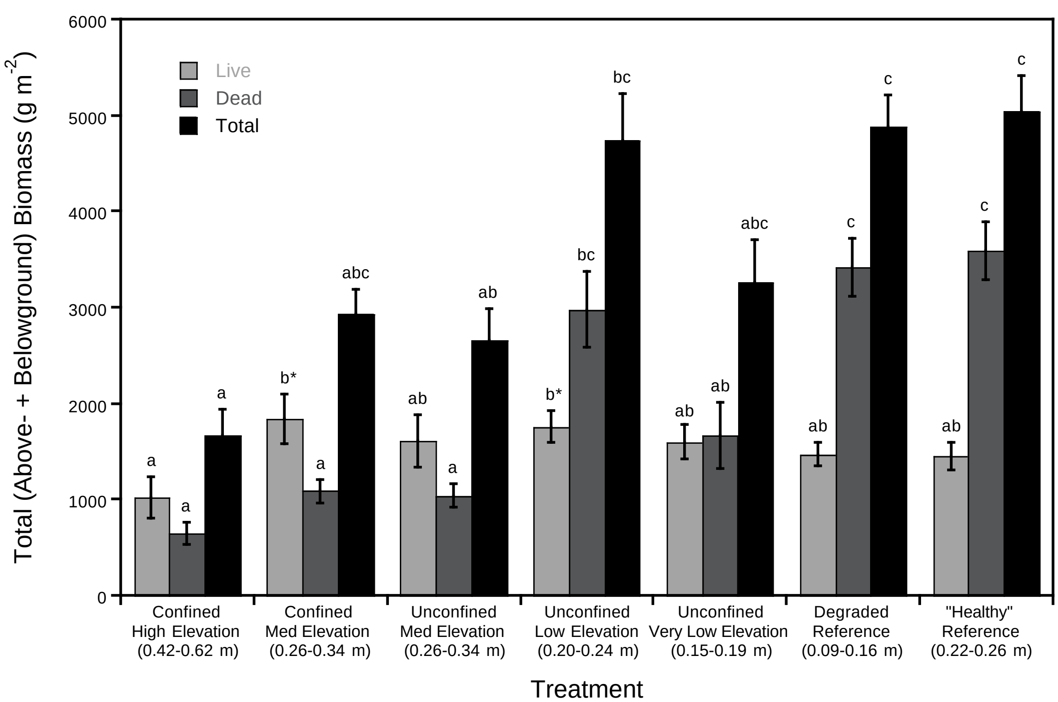


Figure 40 - Total (above- + belowground) live, dead, and live + dead (total) biomass averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Treatment effect on live total biomass was significant at P=0.07. Means with different letters identify significant differences within a response variable.

Components of Total Biomass

Aboveground Biomass. Live, dead, and total (live + dead) aboveground biomass (Figure 41) had similar trends over treatments as for total biomass (above- + belowground) (Figure 40), with the exception of the Unconfined Low Elevation site. Aboveground biomass (live, dead, and total) varied somewhat by year (Appendix A-6). However, only for dead aboveground biomass was the effect of sampling year influenced by treatment (significant treatment x year interaction, Table 14) (Figure 42). Dead biomass in the reference marshes and the Unconfined Low and Very Low treatments decreased over time, while higher elevation restoration sites (Confined High and Medium, and Unconfined Medium) were relatively constant over time (Figure 42).

Belowground Biomass. Belowground biomass accumulates in soil and provides a measure of the standing crop of belowground organic matter at any point in time. Total (live + dead) belowground biomass was much higher in the two reference marshes and the Unconfined Low and Very Low treatment sites compared to the Confined High and Medium and Unconfined Medium sites (significant treatment effect [Table 14], Figure 43). Within the restoration marshes, higher elevations (> 0.26 m NAVD88) dramatically impaired total belowground organic matter accumulation.

Live belowground biomass, which provides a relative estimate of recent belowground production, varied significantly (*P*= 0.06) with restoration treatment, but this effect depended on sampling year (significant treatment x year interaction [Table 14], Figure 43). The largest differences among treatments occurred in 2013 with the greatest live belowground biomass in the Unconfined Low sites and the lowest in the Confined High Elevation sites (Figure 44). The other restoration sites had live belowground biomass statistically similar to the Healthy Reference marsh in 2013, indicating that recovery of belowground biomass had occurred. In contrast, live belowground biomass in 2013 at the Confined High Elevation site was almost 4x less than at the Healthy Reference site, although not significantly different at *P*=0.05 (Figure 44).

Not surprisingly, dead belowground biomass significantly differed with restoration treatment (Table 14), with the reference marshes having the highest dead belowground biomass and the restoration sites, especially the three at the highest elevations, having the lowest (Figure 43). The exception was for the Unconfined Low Elevation restoration site that had dead belowground biomass similar to the reference marshes (Figure 43). Sampling year significantly affected dead belowground biomass, increasing from 2011 to 2013 (Appendix A-7), but the interaction between sampling year and treatment was not significant (Table 14).

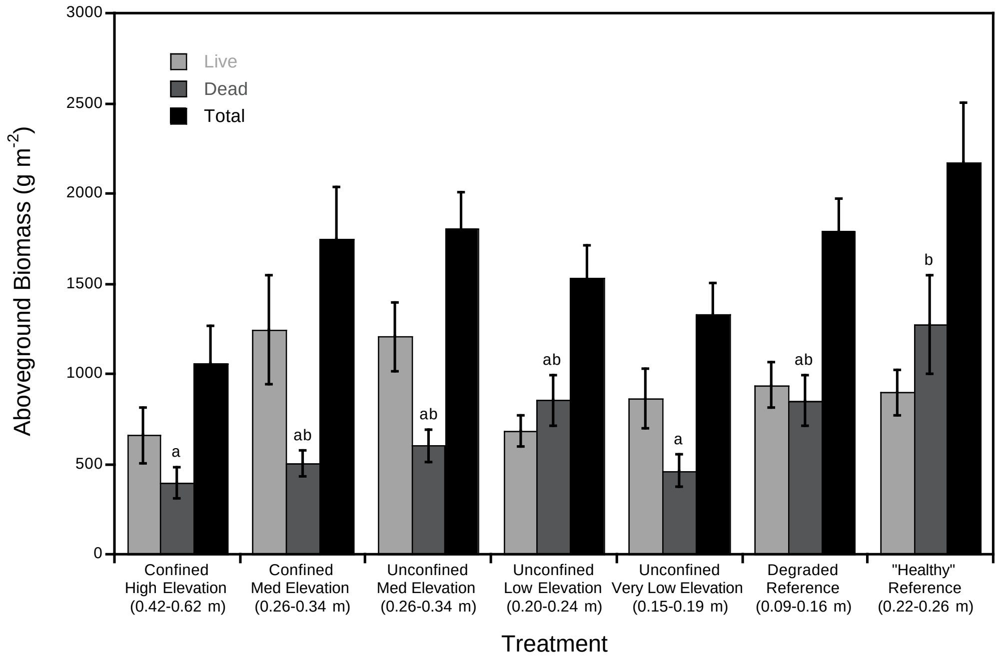


Figure 41 - Live, dead, and total aboveground biomass averaged over Fall 2011, Fall 2012, and Fall 2012 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Means with different letters identify significant differences within a response variable. Although treatment did not significantly affect live or total aboveground biomass (hence, the absence of letters indicating significant differences), means are graphed for descriptive purposes.

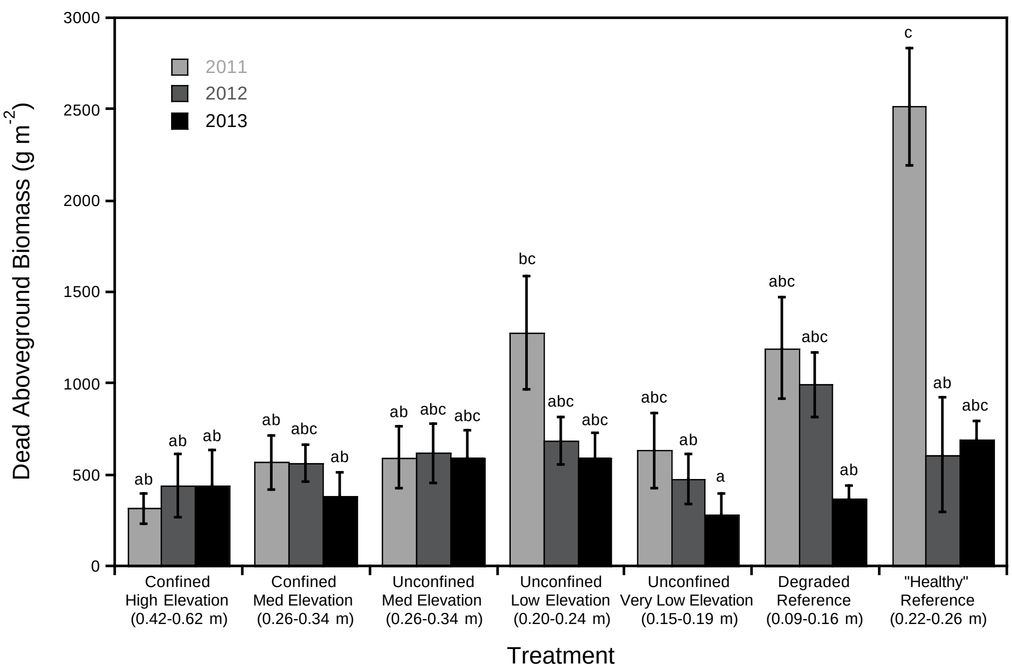


Figure 42 - Dead aboveground biomass by treatment by sampling year. Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. Means with different letters identify significant differences.

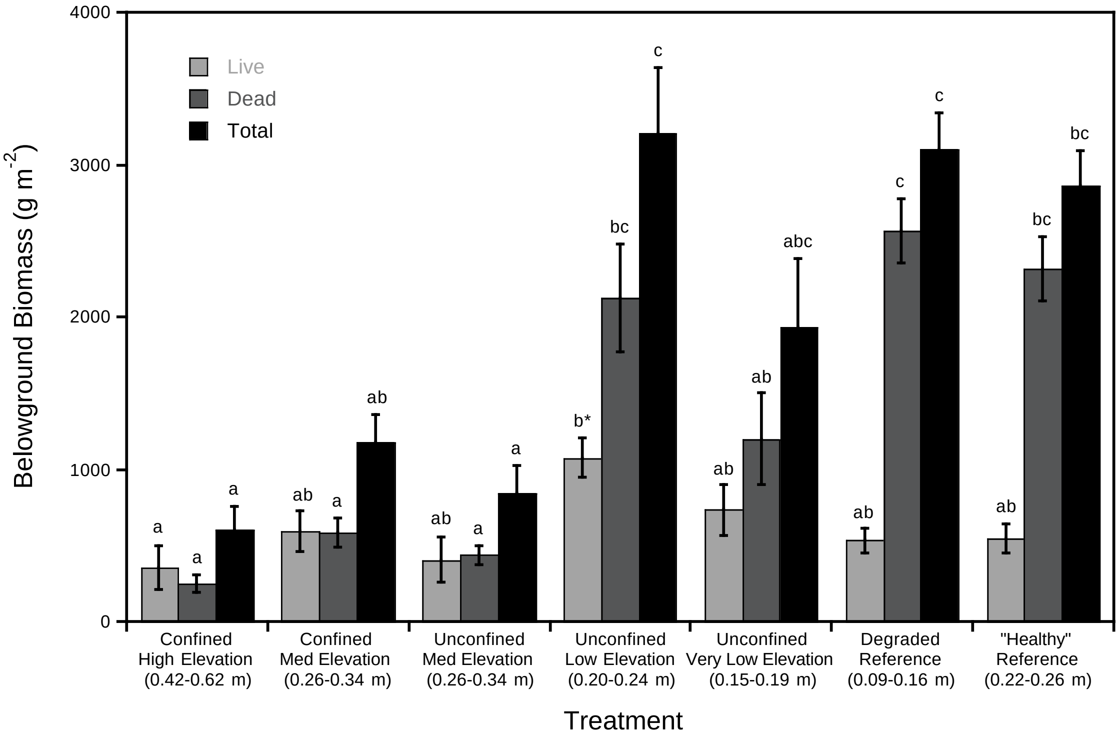


Figure 43 - Live, dead, and total belowground biomass averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events. Values presented as the mean ± 1 SE (error bars) (n=15) for each treatment. Means with different letters identify significant differences within a response variable.

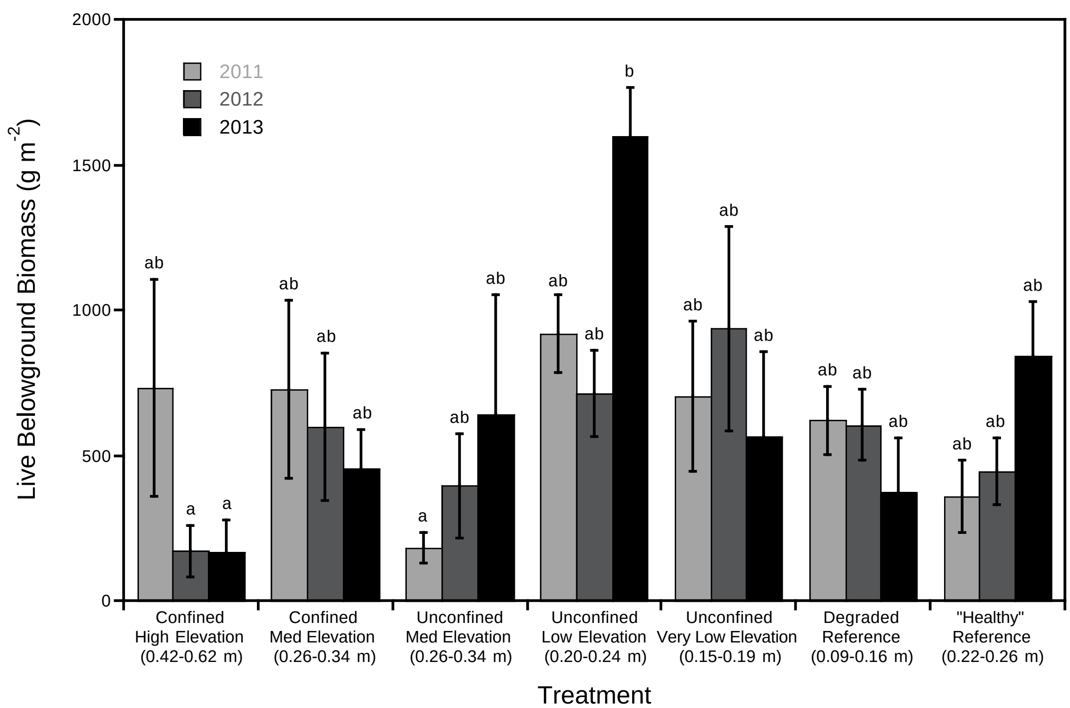


Figure 44 - Live belowground biomass by treatment by sampling year. Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. Means with different letters identify significant differences.

Relationship between Total Plant Biomass and Abiotic Drivers

As previously noted, total plant biomass (above- + belowground, live and dead) significantly varied with restoration treatment-level (Table 14). The seven restoration treatment-levels, which included the two reference marshes, varied in a number of abiotic variables. We measured 26 potentially important abiotic variables in this research. Total biomass had highly significant regressions with relatively high coefficients of determinations (r^2) for four of these variables: soil moisture – r^2=0.59, P<0.0001; soil Eh@15cm – r^2 = 0.54, P<0.0001; soil bulk density – r^2 = 0.56, P<0.0001; soil organic matter – r^2 = 0.51, P<0.0001). Moisture and organic matter were positively related to total biomass, while bulk density and Eh@15cm were negatively associated with total biomass. Three other variables, marsh elevation, total soil salts and Eh@2cm, were also significantly related, all negatively, to total biomass but explained less of the variation in the variables (r^2 = 0.38, P<0.0001; r^2 = 0.29, P<0.0008; r^2 = 0.27, P<0.0013; respectively).

To better understand the relative influence of the 26 abiotic variables on total biomass, we first performed principal components analysis to reduce the number of variables to six principal components (all with eigenvalues >1) that were independent, linear combinations of the variables. These six factors explained 87% of the variation in the abiotic data. Factor 1, which explained 28% of the variation in the abiotic data, was highly positively correlated with marsh elevation, Eh, and bulk density and highly negatively correlated with soil moisture, soil organic matter, and conductivity; we interpreted this factor as a marsh elevation-soil oxidation factor with low soil moisture and organic matter. Factor 2, which explained 20% of the abiotic variation, was highly positively correlated with the soil textural components and manganese; we interpreted this factor as representing soil texture. Factor 3, which explained 18% of the variation in the abiotic data, was highly positively correlated with total soil salts and related cations like sodium and potassium; we interpreted this factor as a soil salinity factor. Factor 4, which explained 8% of the variation in the abiotic data was positively correlated with soil pH and negatively correlated with soil ammonium; we interpreted this factor as a soil pH factor. Factors 5 and 6 only explained 6% and 5%, respectively, of the abiotic variation; Factor 5 was positively associated with nitrate and Factor 6 was positively associated with soil temperature.

We next did a stepwise multiple regression of total biomass (the dependent variable) on factor scores derived from the six factors (independent variables). We could explain 66% of the variation in total biomass with four of the six factors. The most important factor controlling total biomass was Factor 1 (marsh elevation and related soil oxidation and soil moisture), which explained 35% of the variation in total biomass. Factor 2, soil texture, was the second most important controlling factor explaining 15% of the variation in total biomass. Factor 3, soil salts, explained another 8% of the total biomass variation, and Factor 4, pH, explained the smallest, but still significant percentage, at 5%. Thus, as marsh elevation increased and soil moisture decreased and soils became more oxidized (higher redox potentials), plant total biomass decreased. Also, albeit of lesser importance, increases in soil textural components (especially silt and clay) and soil salts were related to lower total plant biomass.

Functional Ecosystem Responses

To assess if restored marshes are functioning similarly to reference marshes, we measured selected functional responses, e.g., accretion, plant decomposition, belowground productivity, etc. These responses to the restoration treatments are described below.

Accretion

The restoration treatments significantly affected sedimentation (Table 14). Accretion was highest in the Unconfined Very Low treatment and lowest in the Confined High Elevation treatment (Figure 45). Because of the marsh fire that burnt the study plots within the Healthy Reference marsh, we were unable to collect accretion data at that location. However, we would expect accretion in the Healthy Reference marsh to have been similar to that in the Unconfined Low Elevation site, which had a similar elevation to the Healthy Reference marsh (Figure 25). Accretion was inversely related to marsh elevation (r- -0.55, p= 0.002), with the least accretion in the Confined High Elevation site and most accretion in the Unconfined Very Low Elevation site (Figure 45). Although accretion was statistically similar in all restoration treatments compared to the Degraded Reference marsh, the highest elevation fill-area (Confined-High) had, on average, a third less accretion than the degraded Reference marsh and a fifth less than the Very Low Elevation Treatment (Figure 45). Accretion can be an important process contributing to the maintenance of marsh elevation as sea level rises. These results show that marsh accretion can proceed after sediment-slurry restoration as long as sediment-addition is not too great, resulting in an infrequently flooded marsh surface.

Aboveground Production

Live, dead and total (live + dead) aboveground production in 2012 and 2013 were determined by measuring the regrowth of plots that were clipped to ground level in each preceding year. Only total aboveground production had a significant response to the restoration treatments (Table 14). Total aboveground production peaked in the Unconfined Medium and Low Elevation sites and was lowest in the Confined Medium Elevation site (Figure 46). The Unconfined Low Elevation site had significantly greater aboveground production than the Confined Medium Elevation site (Figure 46). Live and total aboveground production significantly increased from 2012 to 2013 (Appendix A-8). Aboveground production data were not available for the Healthy Reference marsh because of the 2012 fire. In general, these results suggest that unconfined sediment restoration that yields moderate surface elevations (0.20 to 0.34 m NAVD88) promotes the highest aboveground production, albeit differences were not always statistically significant. This elevation range allowed a hydroperiod (Figure 24) in the study area that was conducive to plant growth.

Belowground Production

No significant treatment effect was found for belowground production (Table 14). Variation in the belowground production was large and missing data at the Confined High Elevation site and the Healthy Reference marsh precluded identifying significant effects (Figure 47). Regardless, average values (Figure 47) generally support conclusions reached for aboveground production.

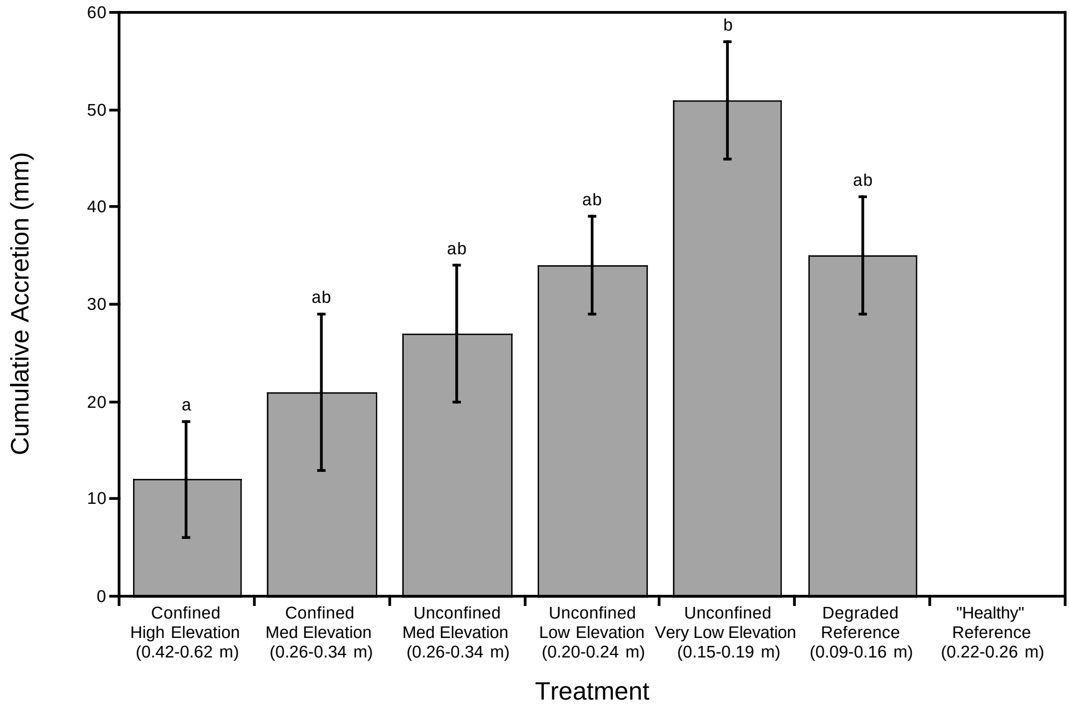


Figure 45 - Treatment effects on cumulative accretion averaged over 2012 and 2013. Values presented as the mean ± 1 SE (error bars) (n=10) for each treatment. The Healthy Reference marsh was affected by a marsh fire in 2012. Means with different letters identify significant differences.

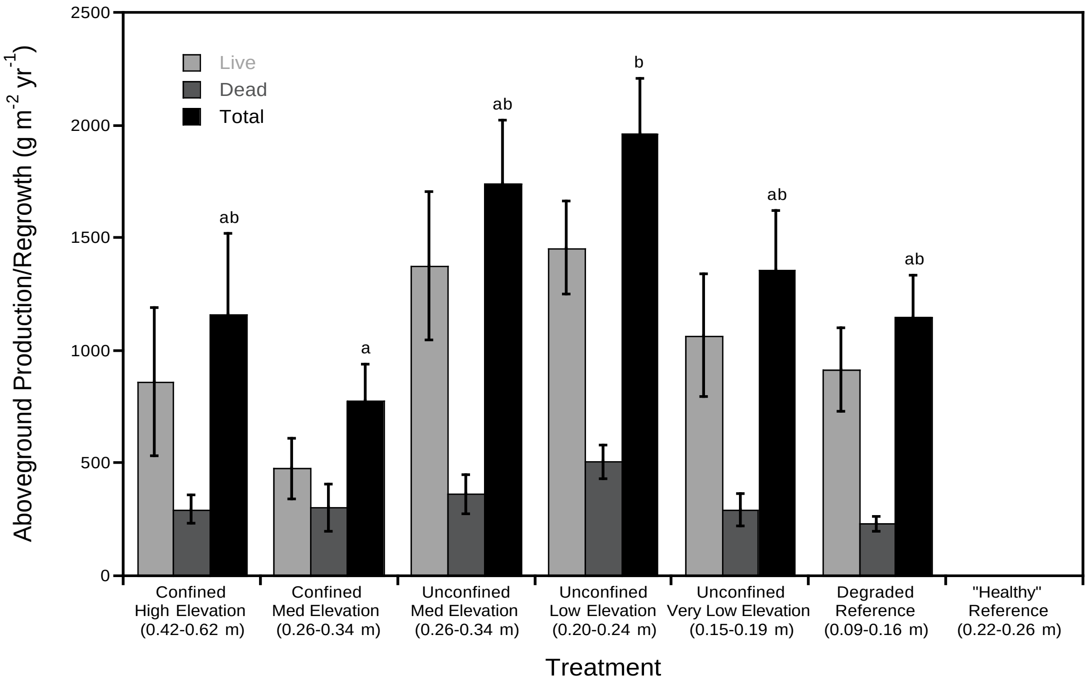


Figure 46 - Aboveground production averaged over Fall 2012 and Fall 2013 sampling events. Values are means ± 1 SE (error bars) (n-10) for each treatment. The Healthy Reference marsh burnt in 2012. Means with different letters identify significant differences within a response variable. Although treatment did not significantly affect live or dead aboveground production (hence, the absence of letters indicating significant differences), means are graphed for descriptive purposes.

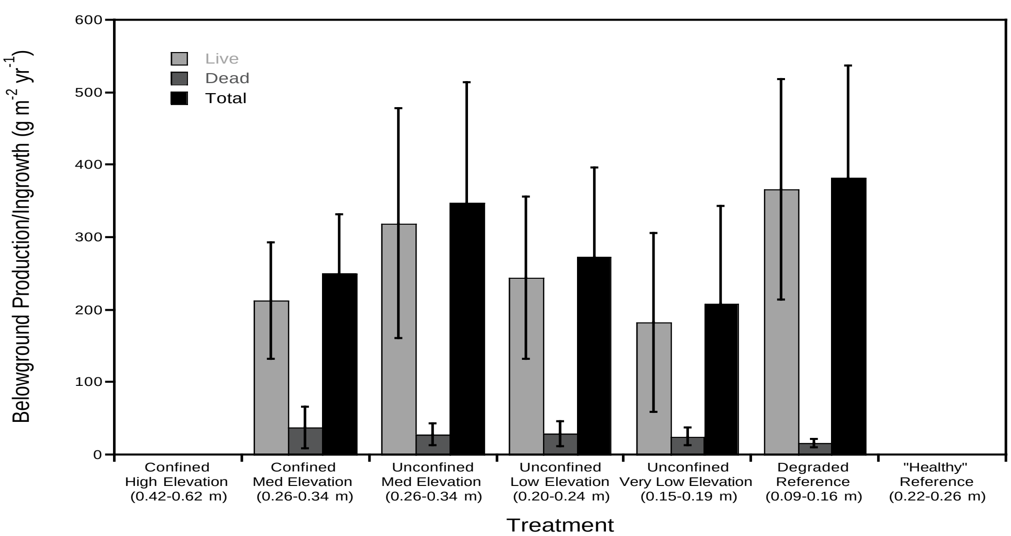


Figure 47 - Belowground production measured over a one-year period from October 1, 2011 to September 26, 2012. Values are means ± 1 SE (error bars) (n=5) for each treatment. The Healthy Reference marsh burnt in 2012, and the Confined High Elevation site experienced hurricane and animal disturbance.

Cellulose Decomposition

The restoration treatments had only a marginally significant effect (*P*=0.09) on cellulose degradation, as measured with cotton strips (Figure 48), which varied on average from 3.8 to 5.2 percent cotton tensile strength loss per day (% CTSL d^{-1}). The Confined Medium Elevation site had significantly higher cellulose degradation than the Healthy Reference marsh. All other treatment sites had intermediate cellulose degradation rates. Cellulose degradation differed greatly over time (Table 14) with 2013 having the highest degradation rates (Figure 48, insert).

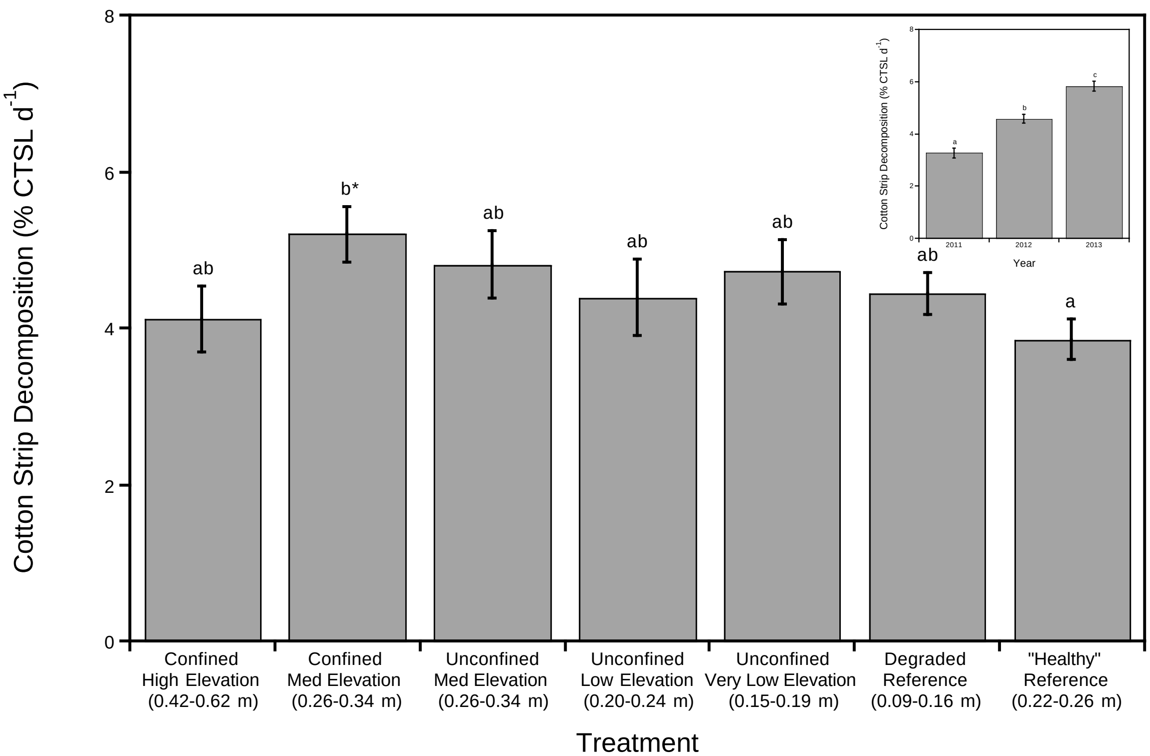


Figure 48 - Cotton tensile-strength loss (CTSL) as percent per day, averaged over Fall 2011, Fall 2012, and Fall 2013 sampling events and three soil depths. Values presented as the mean ± 1 SE (error bars) (n=45) for each treatment. Means with different letters identify significant differences. b* indicates multiple comparisons significant at P<0.10.

Plant Litter Decomposition

The decomposition of aboveground leaves and stems placed on the soil surface for a 1-year period was significantly affected by the restoration treatments (Table 14). The Confined Medium Elevation had the lowest aboveground decomposition (highest % mass remaining), which was significantly greater than that of the Unconfined Low Elevation site and Degraded Reference marsh (Figure 49). Aboveground litter decomposition was measured in 2012 only; consequently, no year or treatment x year analyses could be performed (Table 14).

Belowground plant material placed in the soil was also significantly affected by the restoration treatments (Table 14). As observed for aboveground decomposition, belowground decomposition was slowest in the Confined Medium Elevation site, which was significantly slower than for the belowground material placed in the Unconfined Very Low Elevation site (Figure 49). All other restoration sites had belowground decomposition rates intermediate between these extremes. As for aboveground litter decomposition, no year or treatment x year interaction analyses could be performed on belowground decomposition measured in 2012 only (Table 14).

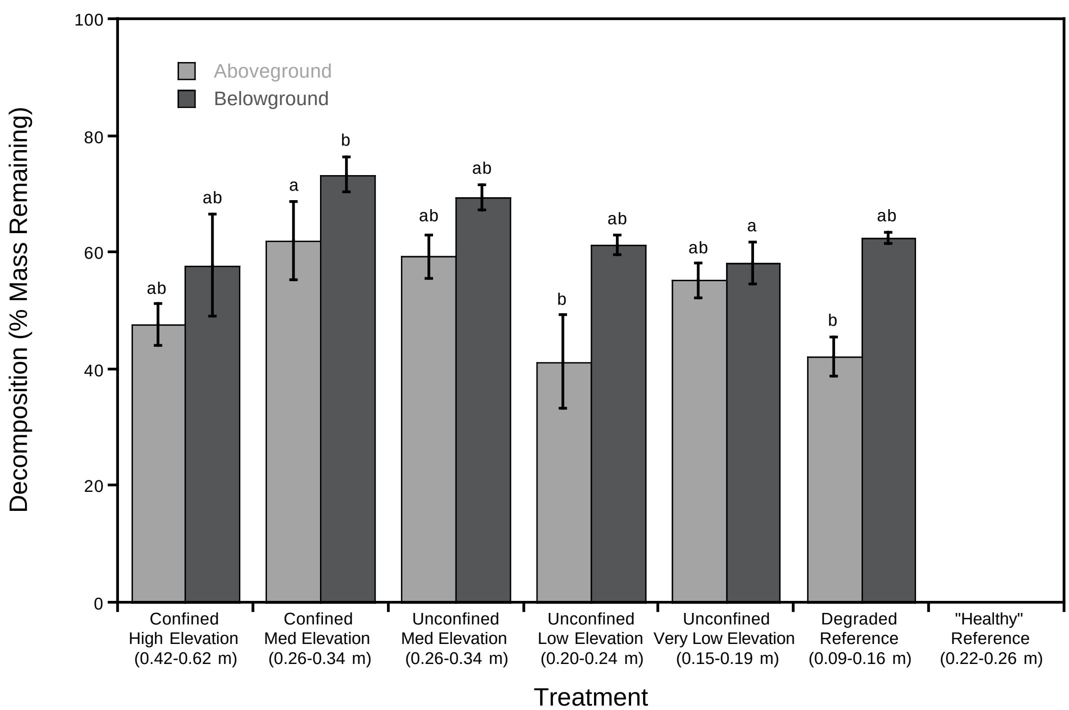


Figure 49 - Decomposition of aboveground and belowground plant organic matter over a 1-year period. Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. The Healthy Reference marsh burnt in 2012. Means with different letters identify significant differences within a response variable.

DISCUSSION

The overall goal of this research was to determine if marsh restoration with hydraulically dredged sediment-slurries could restore, or at least improve, the ecological structure and function of degraded brackish marshes within the Barataria Landbridge project area. The restoration-effects of sediment-slurry application were compared to two types of nearby reference marshes – a Healthy Reference marsh, which appeared visually intact but was characterized by hummocky vegetation intermixed with numerous gullies, and a Degraded Reference marsh, which was in a highly fragmented state of deterioration and appeared similar to project marshes before sediment application. Restoration success was based on two criteria: (1) the ability to reach ecological equivalency with the Healthy Reference marsh and (2) the ability to reach a condition indicative of ecological improvement compared to the Degraded Reference marsh. The latter criterion is more likely attainable in the short-term, but in itself is not a satisfactory measure of restoration success. In contrast, the former criterion matches the ultimate goal of ecological restoration, i.e., restoration to a healthy reference condition, but is often difficult to achieve.

Geospatial Assessments

The Barataria Landbridge geospatial analysis was completed using four base maps; 2008, 2010, 2011, and 2012. The 2008 pre-construction aerial was an orthophoto-quad flown by the U.S. Geological Survey, the remaining three post-construction base maps were flown by Aero-Data Corporation, LLC under a personal service contract for this study. Images used in post-construction analyses are high-resolution scans (2032 dpi) and have a 0.076-meter pixel ground resolution. All of the base maps were geo-registered to the North American Datum of 1983 (NAD83) and to identifiable ground control positions with coordinates acquired from ground survey using ERDAS Imagine 8.7. Maintaining consistent mapping and landform terminology was critical to accurately assessing temporal and spatial changes throughout the four-year study period. A list of terminology, such as map acronyms, surface feature descriptions, and definitions was created early in study period and used throughout the study.

The Barataria Landbridge study area consists of 1,975 acres of marsh separated by the Harvey Cutoff canal. The land area west of the Harvey Cutoff is locally known as *Temple Island* and the land area east of the Harvey Cutoff is lease property, managed by the *Little Lake Hunting Club*. Temple Island is the smaller of the two study areas with 255.1 hectares and the Little Lake Club has 544.5 hectares. The study area was further divided into nine treatment cells; six Confined sediment cells, two Unconfined sediment cells, and one Control cell, which also served as the Degraded Reference marsh. A boundary control overlay was constructed using the outer retention levees and salient surface features to circumscribe the project area.

In 2008, marsh conditions in the Barataria Landbridge study area were typical of a rapidly subsiding and deteriorating deltaic marsh. Vegetation was generally weak and consisted of small fragmented communities of predominantly *Spartina patens* (marshhay cordgrass). Vegetation was generally shallow rooted in soft unconsolidated muck and often up-rooted in mass during storm events. Soil banks along major water bodies were severely eroded and had collapsed where vegetation had been under-cut by waves, tides, and storm surges. The marsh was dominated by numerous shallow-water open ponds. In 2008 (pre-construction conditions), the study area consisted of 498.5 ha (62.4%) water and 300.7 ha (37.6%) vegetation, or slightly less than a 2:1 water-to-vegetation ratio. Vegetation was spatially disproportionate within the study area, with marsh areas near large water bodies containing less vegetation and more open water than the interior marsh. Treatment cells near Bayou Rigolettes contained an average of 73% water and 27% vegetation; or slightly greater than a 2.5:1 water-to-vegetation ratio. During the same-time period, interior marshes contained about 20% more vegetation and were near a balanced 1:1 ratio of vegetation to water.

At the end of 2010 (1st year post-construction), landscape changes within the study area were significant with an overall 78% increase in vegetative cover and a 47% reduction in open water from 2008. Vegetation within the higher elevated Confined cells consisted almost entirely of annual species. Plant community composition appeared to be driven by soil moisture, with numerous micro-environments supporting a number of diverse plant communities. Although there was a high degree of species richness within the Confined cells, no one species appeared to be dominant.

Vegetative recovery was relatively uniform within like-treatment cells, with the largest increases occurring in the more elevated Confined sediment cells and less in the Unconfined sediment cells. Most notable were changes in the number of individual plant units and in plant unit size. Within the Confined sediment cells, the number of individual plant units decreased from 96 to 18 and increased in size from 0.24 to 3.6 hectares. Within the Unconfined sediment cells, the number of individual plant units increased from 213 to 3,990 in the East Unconfined cell and from 130 to 4,679 in the West Unconfined cell, with an average plant community size of 0.016 hectares. Although these were small plant units, the large number was a good indicator that there was a big population of germinating plant propagules in both Unconfined cells. The Control cell ended 2010 with 39% vegetative cover, about a 20% loss and the only cell in 2010 to lose vegetative cover.

Between 2010 and 2011 there were only minor habitat shifts within the study area as sediments continued to de-water and cells became increasingly drier. The most noticeable changes occurred within the East Confined cells, where previously shallow, open-water flats dried out leaving large open areas of unvegetated soils. In 2010, there were no measureable unvegetated dry soil areas; however, by the end of the 2011 growing season, there were 150.7 hectares of unvegetated soils within the study area. By the end of 2011, the Barataria Landbridge marsh consisted of 69% vegetative cover, 12% open water, and 19% unvegetated bare soils. Changes in habitat between 2010 and 2011 were a 4% increase in vegetative cover, a 65% decrease in open water, and a 100% increase in unvegetated bare soils.

In August 2012, Hurricane Isaac caused extensive flooding throughout the Barataria Landbridge marsh causing moderate damage to vegetation. In October 2012, a two-day field investigation was conducted about six-weeks post-Isaac. Based on field assessments, it was determined that within the standing perennial plant communities, damage was limited to the above-ground portions of the plants and was primarily mechanical damage; that is, stems were either broken off somewhere above the crown or were still attached to the crown and lodged over. Damage to the below ground portion of the plant was minimal, root and rhizomes were firm, of applicable color, and intact. There was no odor in the immediate root zone to suggest any deteriorating tissue. Wrack-damaged plants in many cases were simply covered with detrital material and were yellow or light brown from lack of sunlight. Many had no mechanically damage, but were simply lodged over.

At the end of the 2012 growing season and post-Hurricane Isaac, the study area consisted of 64% (51.4 ha) vegetative cover, 11% (89.4 ha) open water, and 25% (198.7 ha) unvegetated bare soils. The overall changes in habitat were modest with the exception of the unvegetated areas. Vegetative cover decreased by 7% (41.3 ha), water areas increased by 4% (3.6 ha), and the unvegetated bare soil areas increased by 32% (48.2 ha). From 2008 to 2012, the Landbridge marsh increased in vegetative cover by 71%, decreased in surface water areas by 81.8%, and added 198.7 ha (24.9%) of elevated unvegetated soils. In comparison, the un-treated Control marsh lost 41.8% of its 2008 vegetative cover, increased its open water area by 29.9%, and added 3.8 hectares (5.1%) of elevated, but unvegetated soils.

Hydrology and Salinity Assessment

Hydrologic conditions within the study area varied slightly across the three treatment areas. Water levels within the Control marsh and within the Confined sediment cells were statistically the same, suggesting that the remaining low retention levees had little effect on water movement when compared to the impounded East Unconfined marsh. The mean water level within the East Unconfined sediment cell was substantially greater having a 0.126 m differential than either the Confined or Control cells. Based on the 24-month sonde dataset, we found a consistent synchronous ebb and flow pattern between the Control marsh and the East Confined cell, with only a slight mean elevation differential (0.004 m) between the two treatments. We also found that water levels within the Control and East Confined marshes were primarily driven by tides. The East Unconfined sediment cell is an impounded marsh formed by a series of protection levees and five fixed crest weirs. Hydrology within the East Unconfined treatment unit is primarily dependent on overflow from high water events initiating from the Little Lake Club Canal and from captured water collected from rainfall events. An analysis of the water data showed that water levels within the Control marsh exceeded the East Unconfined fixed crest weirs (creating a flooding event) 296 times over the 24-month monitoring period. In addition, water levels within the East Unconfined treatment unit never dropped to- or-below 0.0 NAVD88 for the entire monitoring period. However, the Control and East Confined units recorded water levels that were at-or-below 0.0 NAVD88 for 40.9% and 38.2% of the monitoring period, respectively.

Water salinities within the three study treatment units were significantly different from each other. The East Confined treatment unit had the greatest mean salinity of 5.2 ppt, the East Unconfined sediment cell had the lowest mean salinity of 3.6 ppt, and the Control marsh had an intermediate mean salinity of 3.8 ppt; the n-value for all treatment salinity means was 34,503. Salinity variation was greatest within the two more open free-flow marsh units, both of which are influenced by the same hydrologic regime. The East Confined treatment unit had a slightly greater variability (range 12.2 ppt) than the Control marsh (range 11.0 ppt). The East Unconfined treatment unit, which is completely impounded and sheltered from outside diurnal hydrology 72.8% of the time, had the least amount of salinity variation, with a range of only 4.02 ppt.

Based on the relatively low mean salinity levels and the post-construction floristic composition, marshes within the project area would be classified as intermediate to brackish marsh. Salinities (even at spike levels) are low enough not to pose any major salt related stress to prevent the establishment or recovery of intermediate to brackish vegetation. However, the sonde-salinities are for open water and may not exactly reflect soil salinities, which are often higher than open water salinities and are more likely related to plant response.

Surface Elevation and Hydrology

The primary objective of the Surface Elevation and Hydrology task was to assess the relationships between marsh elevation and hydrology for the four sediment treatment units and the seven Sample Station treatments. We asked the question: How did treatment sites differ in flooding frequency, flood event duration, total hours flooded and percent of total time flooded?

A flood event was defined as a rising and falling water cycle that begins with a water level below a reference elevation, rises to- or exceeds- the reference elevation, and ends when water falls below the initial elevation; one full cycle (regardless of the flood duration) would constitute one flood event, or frequency.

Thirty-seven elevation transects were surveyed between 2011 and 2012 by T. Baker Smith, LLC, an engineering firm contracted by the Louisiana Coastal Protection and Restoration Authority. The number of transects within each treatment varied, but the total number of elevation points that make up the surface elevation dataset was significantly large and contained approximately 5,900 elevations. We found a significant difference in sediment elevation between the Confined and Unconfined treatments, a significant difference between the East and West Unconfined treatments, but no significant difference in mean elevation between the East and West Confined treatment units. Mean elevations within the East- and West-Confined sites were highest at 0.331 m and 0.327 m, respectively. The East Unconfined unit contained the lowest mean elevation at 0.180 m, with the West Unconfined at an intermediate mean elevation of 0.257 meters.

When assessing critical flooding events, such as frequency and duration of flooding, we found an interaction between treatment hydrology, elevation, and containment (levees). Over the 24-month sampling period, the higher elevated East Confined site experienced 91 flooding events, the intermediately elevated West Unconfined site 250 events; the East Unconfined site, with the lowest mean sediment elevation, had the greatest number of flooding events at 378. Flooding frequency in the Healthy Reference marsh, with a slightly higher (+0.05 m) elevation than the East Confined mean-low elevation, had 95 flooding events, a commensurable increase in flooding events given the slight differences in elevations.

When comparing the duration of flood-events between treatments, the West Unconfined unit, with the greatest capacity to exchange water and the least capacity to retain water, consistently had the shortest mean flooding period of 13.7 hours. The East Confined unit, with an intermediate capacity to exchange and retain water ranked second with a mean of 19.7 flooding hours, and the heavily impounded East Unconfined site ranked last with a mean of 21.9 hours. Flooding frequency in the Healthy Reference marsh was 95, and flooding frequency within the Degraded Reference marsh was 219 flooding events over 24-months. Relative to the permanent sampling stations, flooding frequency was least within the High Confined treatment site (elevation 0.513 m) with 4 flooding events, and greatest in the Unconfined Very Low (elevation 0.76 m) and Degraded Reference (elevation 0.149 m) with 259 and 219 flooding events, respectively. The Confined Medium (elevation 0.309 m), Unconfined Medium (elevation 0.328 m), and Unconfined Low (elevation 0.024 m) sample station treatments had intermediate flooding frequencies of 68, 52, and 130, respectively, over the 24-months monitoring period.

The last measurement in assessing hydrology-sediment relationships was to determine the overall flood duration within treatments; that is, for a specific elevation what is the total hours and percent total time (over 24-months) that treatment units were flooded. In ascending order of total time flooded, the East Confined had the highest sediment surface elevation and the least amount of cumulative flooding with 6.7% or 1,161.5 hours. The Healthy Reference marsh was second in both surface elevation and in cumulative flooding with 7.7% or 1,325.0 hours. The West Unconfined treatment unit was third in sediment elevation and third in total flooding with 8.4% or 1,450.5 hours. The Degraded Reference marsh had the lowest surface elevation of the five units, but second-to-last in cumulative flooding with 20.0% or 3,463.5 hours. The East Unconfined treatment unit was second-to-last in surface elevation, but last in cumulative flooding with 42.9% or 7,398 hours. A complete list of elevation and flooding statistics are listed in Table 11.

Marsh Elevation and the Physico-chemical Environment

The use of both confined and unconfined sediment-slurry application created a suite of marsh elevations and resulting hydrologic regimes that affected soil physico-chemical condition, vegetation composition, vegetation structure, and ecological function. Average sample station elevations, surveyed in summer-fall 2011, within the restored marshes ranged from a high of 0.51 m to a low of 0.18 m NAVD88, with an array of intermediate elevations (Figure 25), The minimum sampling station elevation was 0.16 m and the maximum was 0.62 m NAVD88.

Confined and unconfined sediment-placement initially yielded mixed results. Two years after the confined sediment placement (summer-fall 2011), elevations in some marsh areas (i.e., Confined High Elevation treatment) were twice as high as elevations of the Healthy Reference marsh (0.51 m versus 0.24 m, Figure 25). In contrast, confined deposition to medium elevations (i.e., Confined Medium Elevation treatment) produced elevations statistically equivalent to the Healthy Reference marsh and significantly greater than the Degraded Reference marsh (Figure 25). Relative to the unconfined sediment application method, we also found that two years after deposition, elevations were both significantly greater than (Unconfined Medium Elevation – 0.30 m) and lesser than (Unconfined Very Low Elevation – 0.16 m) the Healthy Reference marsh (0.24 m) (Figure 25). Confined placement had a greater tendency to overfill relative to the Healthy Reference marsh, while unconfined placement had a greater tendency to underfill compared to the Healthy Reference marsh. In the final analysis, however, both placement methodologies can be successful as long as elevations are closely monitored to prevent over- or underfilling. Frequent movement of the discharge pipe may be necessary to create a more uniform elevation.

More than four years after deposition (Spring 2014), elevations at confined marsh sampling stations (+30 cm NAVD88), averaged over High and Medium Elevation treatments, were about 6 cm higher than the Healthy Reference marsh (+24 cm NAVD88) and 15 cm higher than the

Degraded Reference marsh (+15 cm NAVD88). Elevation points from the transect surveys (Figure 28), which surveyed many more points than just the sampling stations, place the confined sediment elevation 3 cm higher (33 cm NAVD88) at +9 cm and +18 cm relative to the Healthy and the Degraded Reference marsh, respectively (Figure 30), regardless of location relative to the Harvey Cutoff. Unconfined treatment elevations at the Unconfined Medium and Unconfined Low sampling stations three years after sediment deposition were about 9 and 1 cm, respectively, greater than the Healthy" Reference marsh, while the Unconfined Very Low site was 3 cm lower (Figure 27). All unconfined elevation treatments were higher than for the Degraded Reference marsh; average elevations for the Very Low, Low, and Medium treatments were 4, 10 and 18 cm, respectively, greater than for the Degraded Reference marsh (Figure 27). When we used the elevation points from the transect surveys within the unconfined portions of the project area (Figure 29), marsh elevations in the western portion were within 2 cm of the Healthy Reference marsh and 11 cm higher than the Degraded Reference marsh (Figure 31), while marsh elevations in the eastern portion were 6 cm lower than the Healthy Reference marsh and 3 cm higher than the Degraded Reference marsh (Figure 31). Differences in resulting elevations after sediment deposition (Figure 32) created different hydrologic regimes (see the *Hydrology Assessments* section, p. 37, of this report for additional discussion on treatment site hydrology).

The extent to which soil texture of the restored marshes had reached equivalency to that of the Healthy Reference marsh or improved upon the Degraded Reference marsh depended upon the particular textural component and restoration treatment. The restored marshes had higher silt and clay content compared to the reference marshes, especially at higher elevations (Figure 33). As a result, mineral matter and soil bulk density (Figures 34 and 35, respectively) were higher in the restored marshes than in the reference marshes. The Unconfined Low and Very Low Elevation sites tended to have lower mineral matter and bulk density than the higher elevation confined sites. Soil organic matter content of the restored sites never reached equivalency with the reference marshes, although the lower elevation unconfined sites had greater organic content than higher elevation confined and unconfined treatments (Figure 34). Thus, we predict that organic matter content of the lower elevation restored sites will reach equivalency with the reference marshes over time and before the higher elevation restored sites do.

Because the restored marshes were for the most part at higher elevations than the reference marshes, soil moisture content was considerably lower in the former, especially for the Confined High Elevation treatment site (Figure 36). Soil Eh reflected these differences in soil moisture, with higher Eh, more oxidized soil, in the restoration sites compared to the reference marshes, especially for the Confined High Elevation treatment (Table 15). Consequently, the restored marshes had not yet reached equivalency with the reference marshes with respect to soil moisture and Eh. Because soil Eh controls, as well as responds to, numerous biogeochemical processes, and because soil organic matter, which was relatively low in the restored marshes, provides a carbon source for many of these important biogeochemical processes, the restored marshes, especially those at higher elevations, had likely not yet reached biogeochemical functional equivalency with reference marshes. However, as previously stated in the Sampling Design section of the Materials and Methods, the Healthy Reference marsh, although appearing intact, was difficult to traverse due to the hummocky vegetation, interspersed among small gullies. This condition was apparent in high-resolution aerial imagery (Google Earth, imagery date 1-25-2015), and is indicative of excessive inundation (Mendelssohn and McKee 1988, Slocum et al.

2005). Hence, the "healthy" designation for this reference marsh is only valid in a relative sense when compared to the Degraded Reference marsh, which was characterized by extensive pond formation and marsh fragmentation. Acknowledging the above, sediment addition that resulted in more oxidized and lower moisture soils compared to the Healthy Reference marsh was likely a beneficial effect of sediment application, except at the highest sediment addition (Confined High Elevation treatment).

Soil ammonium-nitrogen in the higher elevation restored marshes was equivalent to or greater than that in the reference marshes (Table 15). Nitrate-nitrogen concentration was small in comparison to ammonium, but nevertheless tended to be greater at the highest elevations (Table 15). Phosphorus concentrations were consistently greater for sediment-restored marshes compared to reference marshes (Table 15). These trends most probably reflect the high mineral content in the restored marshes; ammonium and available phosphorus (i.e., phosphate) are both sorbed to the clay fraction of soils. Where soil mineral matter and bulk density were low, both in lower elevation restored sites and in reference marshes, inorganic nitrogen and phosphorus were generally low. However, concentrations of inorganic nitrogen and phosphorus were relatively high throughout the study area, and were likely not primary factors influencing plant growth in themselves.

Total salts, which were presented on a soil volume basis (Table 15), provide a measure of the salt content per volume of soil, and is an indicator of the amount of salts to which plant roots are exposed within that unit volume of soil. However, the higher mineral matter at restored sites may have reduced salt availability due to sorption to clay exchange sites. Total salts were higher for the restored marsh sites than the reference marshes (Table 15), likely due to more salt-associated mineral matter in the restored marshes, i.e., higher bulk density (Figure 35). In contrast, soil conductivity, the generally accepted approach for assessing plant response to soil salinity, exhibited a different trend among the treatment sites with lower levels in the restored marshes compared to the reference marshes (Table 15). Soil conductivity, as determined on a 1-part soil to 2-part water extract, provides a measure of conductivity per unit mass of soil after extraction in deionized water. If we use total salts as the indicator of plant exposure, we must conclude that equivalency with the Healthy Reference marsh had not yet occurred and that there had not been an improvement compared to the Degraded Reference marsh. However, from the standpoint of soil conductivity, sediment-restoration generated lower conductivities than the Healthy Reference marsh, which is generally considered beneficial even though these conductivities were not equivalent to reference conditions. Also, lower soil conductivities relative to the Degraded Reference would generally be considered an improvement in soil quality.

In wetlands, where soils are often waterlogged, certain elements can become toxic because of their greater solubility under low Eh conditions (e.g., Fe and Mn) or because of transformations under reducing conditions to more toxic forms (e.g., toxic sulfide produced from non-toxic sulfate). However, given the relatively high Eh and bulk density of the restored marsh soils and the relatively low concentrations of extractable metals, toxicity of any of the aforementioned elements was unlikely under existing conditions. For Fe, sediment application resulted in equivalency with the Healthy Reference marsh but an increase above that in the Degraded Reference marsh (Table 15). In contrast, Mn concentrations in the sediment-restored marshes more than tripled compared to the Healthy Reference marsh and increased by a factor of 10 compared to the Degraded Reference marsh (Table 15).

When all the soil chemical data were merged through a factor analysis, we found that restored sites were distinguished from reference marshes based on two multi-environmental factors. Although restoration sites tended to have higher Factor 1 scores, representing total salt content (Na and other major cations per unit soil volume), than Healthy and Degraded Reference marshes (Figure 37), this was statistically significant for only two of the five restoration treatments (Confined Medium Elevation and Unconfined Very Low Elevation). Thus from a statistical standpoint, ecological equivalency for Factor 1 was achieved for three of the five restoration treatments. Significant changes in Factor 1 scores compared to the Degraded Reference marsh only occurred in two of the five restoration treatments (Figure 37). Factor 2 was interpreted as a soil oxidation-conductivity factor with higher factor scores indicating more oxidized soils with lower conductivity (Table 16). Sediment application increased Factor 2 scores in a near linear fashion, resulting in the Confined High and Medium Elevation sites having significantly higher Factor 2 scores than the Healthy Reference marsh. Compared to the Degraded Reference marsh, Factor 2 scores indicated more oxidized (less saturated) and lower conductivity soils for the three highest elevation restoration sites (Figure 37). Hence, Factor 2 scores indicated ecological equivalency with the Healthy Reference marsh for the three lowest elevation restoration sites; the two highest elevations sites (Confined High and Medium) had significantly higher Factor 2 scores. Relative to the Degraded Reference marsh, an improvement in ecological condition (higher Factor 2 scores) occurred at the three highest elevation sites (Figure 37). Factor 3, which was related to pH and soil Fe, was equivalent for all treatment and reference marshes (Figure 37). In summary, the ability of sediment-slurry restoration to restore ecological equivalency of soil chemical variables to reference conditions depended on the particular restoration treatment, which differed in added sediment and resulting marsh elevation.

Plant Species Composition and Structure

Although 30 different plant species were identified, only three dominated: *Spartina patens*, *Paspalum vaginatum*, and *Schoenoplectus americanus*. Three additional species were of secondary importance: *Schoenoplectus robustus*, *Distichlis spicata*, and *Symphyotrichum* spp. (Table 17).

From the standpoint of total number of species, the restoration sites, except the Confined Medium Elevation, had generally similar values to the Healthy Reference marsh (Table 17). The Degraded Reference marsh had the lowest number of total species. Hence, sediment-slurry application increased the number of total species greatly by creating new substrate that was colonized by a diverse array of species and an environment with lower conductivity and higher soil Eh. However, the Confined High Elevation treatment, even though having a high total species number, contained species not found in either the Healthy or Degraded Reference marshes (Table 17). These species are more typical of infrequently flooded wetlands or low-elevation spoil banks. Hence, the Confined High Elevation treatment was too high in elevation to yield a species composition similar to the Healthy Reference marsh; species compositional equivalency was not attained here. The remaining restoration treatments, although better approaching equivalency with respect to species biomass with the Healthy Reference marsh, also differed in some dramatic ways. For example, *P. vaginatum* was the dominant species within the restoration treatments, yet this species was absent from the Healthy Reference marsh. Also, although *S. patens* was prevalent in the restoration treatments, its biomass was approximately half of that found for the Healthy Reference marsh (Table 17). It is apparent that more time is needed before the restoration sites reach species compositional equivalency with the Healthy Reference marsh. However, the restoration sites do have a much more diverse species composition than the Degraded Reference marsh and numerous species in common with the Healthy Reference Marsh (Table 17).

The factor analysis of the species biomasses identified three species associations, two of which were significantly affected by the restoration treatments (Figure 38). Species association #1 (Factor 1) included plant species that characterized the Confined High Elevation treatment, which had the highest elevation of any of the restoration sites. This treatment was relatively distinct from the others in both characteristic species and somewhat extreme levels of some soil physico-chemical variables. However, statistical equivalency with the Healthy Reference marsh was attained for all treatments, even though the Confined High Elevation treatment tended to differ (Figure 38). Additionally, there was no statistically significant difference compared to the Degraded Reference marsh. Species association #2 (Factor 2) was associated with *P. vaginatum*, which was the most dominant species in the restoration sites, and the relative paucity of *S. patens*, which was more dominant in the Healthy Reference marsh compared to the restoration sites. *Sagittaria lancifolia*, although not one of the 10 most dominant species (but among the top 15), was found only in the Healthy Reference marsh. Consequently, the presence of this species in the restoration sites will likely be a good indicator that species compositional equivalency has begun. With the exception of the Unconfined Low Elevation site, none of the restoration sites reached statistical equivalency with the Healthy Reference marsh, and there was no change compared to the Degraded Reference marsh (Figure 38). Species association #3 (Factor 3), which included *Schoenoplectus robustus* and *Echinochloa crusgalli,* showed equivalency with reference marshes.

Average species richness was greatest in the Healthy Reference marsh, and significantly greater than the Degraded Reference marsh (Figure 39). Sediment-slurry addition restored species richness at the restoration treatment sites to levels statistically equivalent to the Healthy Reference marsh, with the exception of the Confined and Unconfined Medium elevation treatments. However, the species richness of the restored sites was not statistically greater than for the Degraded Reference marsh (Figure 39); hence, based on the average species richness, sediment application did not yield an improvement beyond that in the Degraded Reference marsh. Nevertheless, species composition within the restored sites was very different from that of the Degraded Reference marsh (Table 17). Hence, it would appear that species richness in the restored marshes is beginning to proceed toward that of the Healthy Reference marsh and away from the Degraded Reference marsh, but more time is required before species richness equivalency can be concluded.

The Unconfined Low Elevation treatment was the only restoration site to quantitatively approach the total biomass (live plus dead above- and belowground) of the Healthy Reference Marsh. Statistical equivalency also occurred for the Confined Medium and Unconfined Very Low treatments, albeit at much lower biomasses (Figure 40). All other restoration treatments resulted in substantially lower total biomass, sometimes significantly lower, as for the Confined High and Unconfined Medium Elevation sites. Relative to the Degraded Reference marsh, total biomass was substantially lower for higher elevation restoration treatments; only the Unconfined Low Elevation treatment had total biomass quantitatively equal to the Degraded Reference marsh. Total live biomass in the restoration marshes was statistically equivalent to the Healthy Reference marsh, which did not differ from the Degraded Reference marsh (Figure 40). Dead total biomass in the restoration marshes had not reached equivalency with either reference marsh, with the exception of the Unconfined Low treatment (Figure 40).

For live and total (live+dead) aboveground biomass, restored marshes were statistically equivalent with reference marshes. Restoration treatment only significantly affected dead biomass (Figure 41); Confined Medium, Unconfined Medium, and Unconfined Low restoration sites were equivalent to the Healthy Reference marsh, while Confined High and Unconfined Very Low restoration sites had significantly lower dead biomass than the Healthy Reference marsh (Figure 41). Nonetheless, the trend across treatments for aboveground biomass (live + dead) indicates that moderate elevation treatment sites (Confined Medium, Unconfined Medium, and Unconfined Low) approached that of the Healthy Reference marsh, suggesting aboveground biomass recovery. Aboveground biomass (live + dead) for moderate elevation treatments was also similar to that of the Degraded Reference marsh (Figure 41).

Sediment-slurry addition in the Unconfined Low and Very Low Elevation treatments restored belowground (live + dead) biomass to that of the Healthy Reference marsh (Figure 43). In contrast, high elevations, whether in confined or unconfined treatments, impaired belowground biomass accumulation. Interestingly, the Degraded Reference marsh had total belowground biomass equivalent to the Healthy Reference marsh (Figure 43), suggesting that the hummocky nature of plant growth in both locations produced similar belowground plant growth patterns. Relative to the Degraded Reference marsh, total belowground biomass was reduced at the higher elevation restoration sites (Confined High, Confined Medium, and Unconfined Medium) and maintained at the lower elevation sites (Unconfined Low and Very Low treatments). The Unconfined Low Elevation site also supported the most live belowground biomass, particularly in 2013 (Figure 44). By 2013, live belowground biomass in the restored sites had recovered to that of the Healthy Reference, although the highest elevation restoration site (Confined High) tended to yield the lowest live belowground biomass (Figure 44).

The most important environmental controls on total above- plus belowground biomass were associated with differential marsh elevation. A stepwise multiple regression of total biomass on factor scores derived from principal components analysis of 26 abiotic variables found that 66% of the variation in total biomass could be explained by these variables. The most important variables controlling total plant biomass were differential marsh elevation and associated soil oxidation and soil moisture; higher elevations reduced soil moisture, increased soil oxidation, and negatively affected total biomass. To a lesser extent, higher percentages of soil textural components (especially silt and clay) and higher total soil salts were also related to lower total biomass.

Ecological Functions

Sediment-slurry application, although increasing marsh elevation, did not impair subsequent sedimentation, although sediment accretion was lowest, albeit non-significantly, for the Confined High Elevation treatment (Figure 45). Because accretion plots in the Healthy Reference marsh were burnt during the 2012 fire, a mean accretion rate for this reference condition is missing. However, we assume that accretion rates in the Healthy Reference marsh were similar to those in Unconfined Low restoration site because of their similar marsh elevations (Figure 25). If so, we would conclude that accretion had likely reached equivalency with all restoration sites, although the Confined High Elevation site tended to be lowest. Relative to the Degraded Reference marsh, marsh accretion was maintained but not improved (Figure 45). Thus, marsh accretion to aid in counterbalancing relative sea-level rise (eustacy and isostacy) can occur after sediment-slurry restoration, as long as the resulting marsh elevations are not so high as to prevent regular tidal flooding and resulting sedimentation.

Aboveground production (live+dead) was greatest at moderate elevations within the Unconfined Medium and Low Elevation treatments and lowest in the Confined Medium Elevation site (Figure 46). Although not statistically significant, it would appear that aboveground production in these moderate elevation restoration sites has surpassed that in the Degraded Reference marsh (data not available for the Healthy Reference Marsh because of the 2012 fire), suggesting an improvement in aboveground production relative to the Degraded Reference marsh. Live aboveground production (Figure 46) had similar trends across restoration treatments, although the overall treatment effect was not significant (Table 14). Dead aboveground production for the restoration marshes did not significantly differ from that of the Degraded Reference marsh (Figure 46). Belowground production (Figure 47) generally supports the findings from aboveground production, but missing data and high variability prevent strong conclusions.

Cellulose degradation in restored marshes was equivalent to that in the Healthy Reference marsh, with the exception of the Confined Medium Elevation site that had higher cellulose decomposition rates (Figure 48). Cellulose degradation in the restoration marshes was also equivalent to that in the Degraded Reference marsh. Cellulose had highest decomposition rates in the Confined Med Elevation site and lowest in the Healthy Reference marsh (Figure 48). Aboveground plant litter decomposition in the restoration marshes was equivalent to the Degraded Reference, except for the Confined Medium Elevation site, which was slower (Figure 49). For belowground plant litter decomposition, no significant differences were identified between the Degraded Reference marsh and the restoration marshes (Figure 49) (Healthy Reference Marsh was lost due to the 2012 fire). These data suggest that there was little effect of sediment restoration on the decomposition of relatively labile organic matter like cellulose, although the Confined Medium Elevation sites tended to have greater cellulose decomposition than the Healthy Reference marsh. Less labile aboveground plant litter tended to decompose more slowly at higher elevation treatments; this was statistically significant in the comparison of the Confined Medium Elevation treatment with the Degraded Reference marsh (Figure 49). Belowground litter showed less of a difference in decomposition rates among treatments; however, the Confined Medium Elevation site had slower decomposition rates than the Unconfined Very Low site. Establishing a stronger relationship between organic matter degradation and sediment-slurry application would benefit from further investigation.

SYNTHESIS AND CONCLUSIONS

The overall goal of this research was to determine if marsh restoration with hydraulically dredged sediment-slurries could restore, or at least improve, the ecological structure and function of degraded brackish marshes within the Barataria Landbridge project area. To accomplish this goal, we assessed the effects of sediment-slurry application on temporal changes in vegetative cover, hydrology relative to marsh elevation, soil quality, vegetation establishment and growth, and several important ecological functions within deteriorating interior brackish marshes. These response variables in sediment-slurry restored marshes were compared to two types of nearby reference marshes – a Healthy Reference marsh, which appeared visually intact but characterized by hummocky vegetation intermixed with numerous gullies, and a Degraded Reference marsh, which was in a highly fragmented state of deterioration and appeared similar to project marshes before sediment application. Restoration success was based on two criteria: (1) the ability to reach ecological equivalency with the Healthy Reference marsh and (2) the ability to reach a condition indicative of ecological improvement compared to the Degraded Reference marsh. The primary reference condition for the assessment of temporal variation in vegetative cover and hydrology-marsh elevation relationships was the Degraded Reference marsh.

We addressed three major research questions:

1. What are the effects of sediment-enhanced elevation on vegetative recovery over a range of microhabitats within a deteriorating marsh?
2. What are the physico-chemical factors that control successful vegetative establishment and growth, succession, and resultant ecosystem functions?
3. At what surface elevation does maximum plant recovery occur, and are wetland structure and function equivalent to or surpassing that of un-amended reference wetlands?

What are the effects of sediment-enhanced elevation on vegetative recovery over a range of microhabitats within a deteriorating marsh?

Surface features in the Barataria Landbridge marsh in 2008, before sediment-slurry deposition, were typical of rapidly subsiding and deteriorating deltaic marshes. Vegetation within the Barataria Landbridge marsh consisted of weak and fragmented plant communities of *Spartina patens* (marshhay cordgrass). Marsh shorelines were severely eroded, collapsing, and numerous shallow open-water ponds dominated the marsh as a whole. In 2008, habitat types were limited and consisted of 498.5 ha (1,231.8 ac) of water (62.4%) and 300.7 ha (743.1 ac) of vegetation (37.6%), or slightly less than a 2:1 water-to-vegetation ratio. In the first year post-construction (2010), the same marsh area consists of 67% vegetation (532.6 ha; 1,316 ac) and 33% of open water (266.8 ha; 659.2 ac), or a vegetation-to-water ratio of 2:1. There was an overall 77.5% increase in vegetative cover and a 47.3% reduction in open water between 2008 and 2010. By the end of 2011, the Barataria Landbridge marsh consisted of 69% vegetative cover, 12% open water, and 19% unvegetated bare soils. Changes in habitat between 2010 and 2011 were a 4% increase in vegetative cover, a 65% loss in open water, and a 100% increase in unvegetated bare soils, the latter being most prevalent in the East Confined cells. At the end of the 2012 growing season and post-Hurricane Isaac, the Barataria Landbridge project area consisted of 64.0% vegetative cover (51.4 ha; 1,263.6 ac), 24.9% unvegetated bare soils (198.7 ha; 490.9 ac), and 11.2% of open water marsh (89.4 ha; 220.9 ac). The overall changes in habitat were modest with the exception of the unvegetated areas. Vegetative cover decreased by 7% (41.3 ha), water areas increased by 4% (3.6 ha), and the unvegetated bare soil areas increased by 32% (48.2 ha). From 2008 to 2012, the Landbridge marsh increased in vegetative cover by 71%, decreased in surface water areas by 81.8%, and added 198.7 ha (24.9%) of elevated unvegetated soils. The increase in vegetative cover was associated with an increase in total and average species richness compared to the Degraded Reference marsh, as determined from the 35 permanent sampling stations. However, the dominant plant species in the Healthy Reference marsh were dissimilar to those in the higher elevation restoration sites, although species composition was more diverse relative to the Degraded Reference marsh. Overall, the reference marshes had the greatest total (above- plus belowground, live and dead) biomass (with the exception of the Unconfined Low Elevation treatment), while the highest elevation restoration site (Confined High Elevation) had the lowest total biomass, indicating poor recovery in the latter. The other restoration sites were intermediate in recovery, based on total biomass, with the exception of the Unconfined Low Elevation treatment, which was equivalent in total biomass to the reference marshes. Average total biomass ranged from a low of 1664 g m^{-2} to a high of about 5035 g m^{-2}. In contrast, aboveground biomass (live+dead) did not differ between restoration treatments and reference marshes, although the highest (Confined High) and lowest (Unconfined Very Low) restoration treatments tended to have lower aboveground biomass then restored marshes at intermediate elevations. Belowground biomass decreased markedly with increasing elevation and was significantly lower than for the Healthy Reference marsh. Within the restoration areas, higher elevations (> 0.26 m NAVD88) dramatically impaired total belowground organic matter accumulation.

What are the physico-chemical factors that control successful vegetative establishment and growth, succession, and resultant ecosystem functions?
We found that sediment-slurry application to deteriorated brackish marshes altered soil physico-chemistry, plant recruitment, and ecosystem function, but the intensity and direction (positive or negative) of effects depended on the marsh elevation achieved and the particular response variable. For abiotic variables, 24 of the 26 measured variables, including soil temperature, had significant treatment effects, i.e., varied with marsh elevation. Among the restored marshes, soil bulk density, mineral matter content, soil oxidation (Eh), total salts, individual salt-related elements, and nutrient content (ammonium, nitrate, phosphorus, and manganese) increased with higher sediment-slurry derived elevations, while organic matter, soil moisture, and soil conductivity decreased. For the most part, these abiotic variables, especially those related to differential elevation and soil moisture, drove the ecological responses.

Many of the ecological response variables significantly differed across treatment levels (15 of 23 at $P<0.10$). Again, elevation and resulting hydrology were likely the most important controlling factors. In fact, aerial imagery analysis indicated that plant community composition was driven by soil moisture, with numerous micro-environments supporting a number of diverse plant communities. Flooding frequency and flood durations within the treatment units and sample stations were a function of sediment elevation and watershed efficiency, with the higher and better-drained marshes experiencing the least number of flood events. Flooding frequency within the elevated East Confined treatment unit (with the highest mean sediment elevation) was lowest at 91; flooding frequency within the East Unconfined site (with the lowest mean sediment elevation) was greatest at 378 events. Flooding frequency in the Healthy reference marsh was 95, and flooding frequency within the Degraded reference marsh was 219. Relative to the permanent sampling stations, flooding frequency was least in the High Confined treatment at 4 and most in the Unconfined Very Low treatment and Degraded Reference marsh (259 and 219, respectively).

The Confined Medium, Unconfined Medium, and Unconfined Low treatments had intermediate flooding frequencies (68, 52, and 130, respectively). Also, we generally found length of flooding to be negatively correlated with sediment elevation in relatively open unconfined marshes and confounded in marshes that are impounded with restricted water exchange. Total time flooded increased in the following order: East Confined unit, Healthy Reference marsh, West Unconfined unit, Degraded Reference marsh, and East Unconfined unit. For the permanent sampling stations, total time flooded had similar trends as flooding frequency with lowest values for the Confined High treatment at 1% and highest for the Unconfined Very Low and Degraded Reference marsh (24.25% and 20%, respectively). The other restoration marshes were intermediate in flooding frequency (3.75 – 10.35 %) and similar to that of the Healthy Reference marsh (7.7%). Sampling at permanent ground stations supported the importance of marsh elevation and resultant hydrology as critical abiotic drivers of plant growth response. At higher marsh elevations, soils were drier and more oxidized, and plant total biomass was lower. In addition, higher percentages of silt and clay as well as higher concentrations of soil salts corresponded to lower total biomass. Fifty-eight percent of the variation in total biomass was explained by marsh elevation-soil moisture, soil texture, and soil salts. Within the restoration marshes, total (above- and belowground) biomass, total dead biomass, total belowground biomass, and accretion were all lower at the higher elevations, e.g., the Confined High Elevation treatment; also, aboveground production and aboveground litter decomposition tended to be lower at the higher elevations. Certain species, like *Symphyotrichum subulatum* and *S. divaricatum*, were more prevalent at the highest elevation treatment-levels.

At what surface elevation does maximum plant recovery occur, and are wetland structure and function equivalent to or surpassing that of un-amended wetlands?

Considerable differences were still apparent between the physico-chemical conditions of the restoration sites relative to the Healthy Reference marsh, although final marsh elevations in the restored marshes were at, or approaching, statistical equivalency. Furthermore, there was limited improvement in physico-chemical variables relative to the Degraded Reference marsh; variables that did improve were marsh elevation, soil ammonium and phosphorus, as well as Fe, and potential stressors like conductivity. In fact, when soil chemical variables were combined in a factor analysis, we found equivalency to the reference marshes for some of the restored marshes, but certainly not all. Elevation and resulting hydrology were primary drivers of the majority of abiotic responses.

Total and average species richness in the sediment-restored marshes were equivalent to the Healthy Reference marsh, with the exception of the Confined Medium treatment (total species richness) and Confined Medium and Unconfined Medium treatments (average species richness). Also, the dominant plant species in the Healthy Reference marsh were dissimilar to those in the higher elevation restoration sites, although species composition was more diverse relative to the Degraded Reference marsh. Total biomass (above- and belowground, live and dead) in the lower elevation restored marshes was equivalent to the Healthy Reference marsh and significantly higher than for the Confined High Elevation site. In contrast, aboveground biomass (live+dead) did not differ among restoration treatments and reference marshes suggesting equivalency, although the highest (Confined High) and lowest (Unconfined Very Low) restoration treatments tended to have lower aboveground biomass then restored marshes at intermediate elevations.

Belowground biomass decreased markedly with increasing elevation and was not equivalent to the Healthy Reference marsh.

Functional responses such as accretion, primary production, and organic matter decomposition also responded to sediment-slurry restoration. Accretion rates in the restoration marshes were equal to those in the Degraded Reference marsh, although the Confined High treatment tended to be lower than the reference and was significantly lower than the Unconfined Very Low treatment. Aboveground production within the restoration marshes tended to equal that in the Degraded Reference marsh; the trend among treatments in belowground production was similar to that for aboveground. Although differences in organic matter decomposition occurred among treatments, these differences were not quantitatively large. Cellulose degradation in the restored marshes was equivalent to the Healthy Reference marsh, except the Confined Medium Elevation treatment, as well as being equivalent to the Degraded Reference marsh. Plant litter decomposition was either equivalent to the Degraded Reference marsh (for belowground litter) or near equivalent (for aboveground litter). Unfortunately, comparison of these measures between the Healthy Reference and restoration marshes was not possible due to fire.

In summary, we conclude that the restoration marshes have not yet reached ecological equivalency with reference marshes for many of the variables and processes measured in this research. However, the restored marshes are now very different from the Degraded Reference marsh, which is highly ponded and fragmented, for a number of response variables. Although it is difficult to identify a specific elevation that best promotes functional equivalency because different ecological responses had different optimum elevations, marsh elevations between 0.24 m and 0.31 m NAVD88, resulting in percent time flooded of approximately 4 to 10%, respectively, helped to promote either equivalency with the Healthy Reference marsh (7.7% time flooded) or improvement compared to the Degraded Reference marsh (20% time flooded). Thus, sediment-slurry application was successful in converting degraded fragmented marshes into marshes with contiguous vegetative cover and high species richness, now more closely resembling the Healthy Reference marsh than the Degraded Reference marsh. Nonetheless, more time for functional development of the restoration marshes is necessary before ecological equivalency with the surrounding natural marshes is completely achieved, at least relative to the response variables measured in this research.

REFERENCES

American Public Health Association (APHA), 2005. Standard methods for the analysis of water and wastewater. Twenty-first edition. APHA, Washington, DC, USA.

Barowka and Bonura Engineers and Consultants, L.L.C. 2010. Dedicated Dredging on the Barataria Basin Landbridge Project Completion Report. Project No. BA-36. Prepared for the State of Louisiana, Office of Coastal Protection and Restoration, April 2010.

Blake, G.R. and K.H. Hartge, 1986. Bulk Density. In: A. Klute (Ed.), Methods of soil analysis, Part 1. Physical and mineralogical methods, Second Edition. American Society of Agronomy – Soil Science Society of America, Madison, WI, pp. 363-375.

Boesch, D.F. 1982. Modification in Louisiana, causes, consequences, and options. Conference on Coastal Erosion and Wetlands Proceedings. U. S. Fish and Wildlife Service Biological Service Program FWS/OBS-83/59. 259 pp.

Broome, S. W., W. W. Woodhouse, and E. D. Seneca. 1975a. The relationship of mineral nutrients to growth of *Spartina alterniflora* in North Carolina: I. Nutrient status of plants and soils in natural stands. Soil Sci. Soc. Amer. Proc. 39: 295-301.

Broome, S. W., W. W. Woodhouse, and E. D. Seneca. 1975b. The relationship of mineral nutrients to growth of *Spartina alterniflora* in North Carolina: II. The effects of N, P, and Fe fertilizers. Soil Sci. Soc. Amer. Proc. 39:301-307.

Christensen, B.T. and P.A. Malmros, 1982. Loss-on-ignition and carbon content in a beech forest soil profile. Holartic Ecology, 5: 376-380.

Couvillion, B.R., J.A Barras, G.D. Steyer, W. Sleavin, M. Fischer, H. Beck, N. Traham, B. Griffin, and D. Heckman. 2011. Land area change in coastal Louisiana from 1932 to 2010. U.S. Geological Survey Scientific Investigations Map 3164, scale1:265,000, 1 sheet, 12 p. pamphlet.

DeLaune, R.D., S.R. Pezeshki, J.H. Pardue, J.H. Whitcomb, and W.H. Patrick, Jr. 1990. Some influences of sediment addition to a deteriorating salt marsh in the Mississippi Deltaic Plain: a pilot study. Journal of Coastal Research 6:181-188.

Gambrell, R. P. and W. H. Patrick, Jr. 1978. Chemical and microbiological properties of anaerobic soils and sediments. Pages 375-425 *in* D. D. Hook and R. M. M. Crawford, editors. Plant Life in Anaerobic Environments. Ann Arbor Science, Ann Arbor, MI.

Gardner, W.H., 1986. Water Content. In: A. Klute (Ed), Methods of soil analysis, Part 1. Physical and mineralogical methods, Second Edition. American Society of Agronomy – Soil Science Society of America, Madison, WI, pp. 493-544.

Gee, G.W. and J.W. Bauder, 1986. Particle-size analysis. In: A. Klute (Ed), Methods of soil analysis, Part 1. Physical and mineralogical methods, Second Edition. American Society of Agronomy – Soil Science Society of America, Madison, WI, pp. 383-411.

Graham, S. A. and I. A. Mendelssohn. 2013. Functional assessment of differential sediment slurry applications in a deteriorating brackish marsh. Ecological Engineering 51:264-274.

Harrison, A. F., P. M. Latter, and D. W. H. Walton (eds.). 1988. Cotton strip assay: an index of decomposition in soils. (ITE Symposium no. 24). Grange-over-Sands: Institute of Terrestrial Ecology.

Kaiser, H. F. 1958. The varimax criterion for analytic rotation in factor analysis. Psychometrika 23:187-200.

Lindsay, W.L. and W.A. Norvell, 1978. Development of a DTPA soil test for zinc, iron, manganese, and copper. Soil Science Society of America Proceedings, 42: 421-428.

Maltby, E. 1987. Use of cotton strip assay in wetland and upland environments - an international perspective. In: Harrison, A.F., P.M. Latter, D.W. H. Walton (eds.) Cotton strip assay: an index of decomposition in soils. (ITE Symposium no. 24). Grange-over-Sands: Institute of Terrestrial Ecology.

Mehlich, A. 1984. Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15:1409–1416.

Mendelssohn, I. A. 1979. Nitrogen metabolism in the height forms of *Spartina alterniflora* in North Carolina. Ecology 60: 574-584.

Mendelssohn, I. A. and N. L. Kuhn. 2003. Sediment subsidy: effects on soil-plant responses in a rapidly submerging coastal salt marsh. Ecological Engineering 21:115-128.

Mendelssohn, I. A. and K. L. McKee. 1988. *Spartina alterniflora* die-back in Louisiana: Time course investigation of soil waterlogging effects. J. Ecol. 76:509-521.

Mendelssohn, I. A., R. E. Turner, and K. L. McKee. 1983. Louisiana's eroding coast: management alternatives. J. Limnol. Soc. 5th Africa 9:63-85.

Mulvaney, R.L., 1996. Nitrogen – inorganic forms. In: D.L. Sparks (Ed), Methods of soil analysis, Part 3. Chemical methods. American Society of Agronomy – Soil Science Society of America, Madison, WI, pp. 1123-1184.

Reed, D.J., Editor. 1995. Status and historical trends of hydrologic modification, reduction in sediment availability, and habitat loss in the Barataria and Terrebonne estuarine system. BTNEP Publication No. 20. Thibodaux, La. Barataria-Terrebonne National Estuary Program. 338 pp. plus appendices.

Rhoades, J.D. 1982. Soluble Salts. In: A.L. Page (ed.) Methods of soil analysis, Part 2 Chemical and microbiological properties, 2nd edition. Agronomy 9: 149-157.

Schrift, A., I. A. Mendelssohn and M. Materne. 2008. Salt marsh restoration with sediment-slurry amendments following a drought-induced large-scale disturbance. Wetlands 28:1071-1085.

Slocum, M. G. and I.A. Mendelssohn. 2008. Use of experimental disturbances to assess resilience along a known stress gradient. Ecological Indicators 8:181-190.

Slocum, M. G., I. A. Mendelssohn, and N. L. Kuhn. 2005. Effects of sediment-slurry enrichment on salt marsh rehabilitation: plant and soil responses over seven years. Estuaries 28:519-528.

Stagg, C. L. and I. A. Mendelssohn. 2010. Restoring ecological function to a submerged salt marsh. Restoration Ecology 18:10-17.

Wilsey, B.J., K.L. McKee and I.A. Mendelssohn. 1992. Effects of increased elevation and macro- and micronutrient additions on *Spartina alterniflora* transplant success in salt-marsh dieback areas in Louisiana. Environmental Management 16: 505-511.

APPENDIX

Appendix Table A-1. Locations and elevations of sampling stations including conversion calculations from NAVD88-Geoid09 to NAVD88-Geoid12A.

Station ID*	Northing (LA State Plane, m)	Easting (LA State Plane, m)	Latitute	Longitude	Elevation (NAVD88, Geoid 09 (m)	Geoid Height (N) for Geoid12A (m)	Geoid Height (N) for Geoid09 (m)	ΔN(12A-09) (m)	GEOCON11 estimated Δ in ellipsoid height (2011-2007)** (m)	Elevation (NAVD88, Geoid 12A)*** (m)	Elevation (NAVD88, Geoid 12A) (ft)
1-C-H	121605.922	1117787.198	29.591436	90.117435	0.513	-24.974	-24.934	-0.04	-0.02	0.533	1.750
1-C-M	121590.579	1117821.519	29.591348	90.117119	0.297	-24.974	-24.933	-0.041	-0.02	0.318	1.043
1-DEG	118888.38	1117153	29.566958	90.124268	0.129	-24.905	-24.872	-0.033	-0.02	0.142	0.466
1-REF	123154.989	1118485.037	29.605223	90.110338	0.224	-25.012	-24.967	-0.045	-0.02	0.249	0.816
1-U-M	120930.715	1117719.113	29.585512	90.118495	0.294	-24.957	-24.918	-0.039	-0.02	0.313	1.028
1-U-VL	120495.056	1118901.121	29.581307	90.106049	0.165	-24.939	-24.898	-0.041	-0.02	0.186	0.611
1-U-L	120427.254	1118746.514	29.58071	90.107618	0.230	-24.938	-24.897	-0.041	-0.02	0.251	0.822
2-C-H	120331.118	1116782.712	29.579972	90.128141	0.483	-24.944	-24.911	-0.033	-0.02	0.496	1.626
2-C-M	120290.674	1116798.806	29.579625	90.127828	0.279	-24.943	-24.909	-0.034	-0.02	0.293	0.962
2-DEG	118602.172	1117117.223	29.564347	90.124994	0.147	-24.897	-24.866	-0.031	-0.02	0.158	0.518
2-REF	123076.281	1118562.314	29.604622	90.109409	0.255	-25.01	-24.964	-0.046	-0.02	0.281	0.922
2-U-M	119970.479	1117705.225	29.576334	90.118489	0.314	-24.93	-24.893	-0.037	-0.02	0.331	1.085
2-U-VL	119932.65	1118922.687	29.57627	90.105933	0.150	-24.924	-24.884	-0.04	-0.02	0.170	0.558
2-U-L	120057.046	1118623.753	29.577417	90.108972	0.240	-24.929	-24.889	-0.04	-0.02	0.260	0.852
3-C-H	119586.689	1116442.926	29.573398	90.131539	0.476	-24.926	-24.895	-0.031	-0.02	0.487	1.597
3-C-M	119420.102	1117827.03	29.571718	90.117216	0.331	-24.916	-24.88	-0.036	-0.02	0.347	1.140
3-DEG	118405.112	1117149.78	29.562765	90.124573	0.090	-24.892	-24.861	-0.031	-0.02	0.101	0.331
3-REF	122917.35	1118653.969	29.603078	90.108364	0.248	-25.005	-24.959	-0.046	-0.02	0.274	0.899
3-U-M	119357.687	1117944.349	29.571191	90.116444	0.308	-24.914	-24.878	-0.036	-0.02	0.324	1.063
3-U-VL	119122.194	1118650.636	29.568955	90.108829	0.158	-24.904	-24.866	-0.038	-0.02	0.176	0.577
3-U-L	119520.941	1118729.454	29.572568	90.107961	0.227	-24.914	-24.875	-0.039	-0.02	0.246	0.807
4-C-H	119445.976	1115760.877	29.572223	90.138523	0.419	-24.925	-24.897	-0.028	-0.02	0.427	1.401
4-C-M	118934.451	1114968.805	29.567657	90.146689	0.320	-24.916	-24.892	-0.024	-0.02	0.324	1.062
4-DEG	118367.424	1117345.946	29.562375	90.122372	0.164	-24.891	-24.858	-0.033	-0.02	0.177	0.581
4-REF	122859.642	1118700.228	29.60258	90.107989	0.226	-25.003	-24.958	-0.045	-0.02	0.251	0.822
4-U-M	118837.961	1115147.584	29.566762	90.144972	0.322	-24.912	-24.888	-0.024	-0.02	0.326	1.071
4-U-VL	118306.823	1115159.27	29.561945	90.144956	0.154	-24.898	-24.875	-0.023	-0.02	0.157	0.514
4-U-L	118344.583	1115173.101	29.562296	90.144785	0.229	-24.899	-24.876	-0.023	-0.02	0.232	0.760
5-C-H	118931.324	1114368.778	29.567912	90.152766	0.620	-24.919	-24.897	-0.022	-0.02	0.622	2.042
5-C-M	118925.02	1114908.074	29.567589	90.147399	0.261	-24.916	-24.892	-0.024	-0.02	0.265	0.868
5-DEG	118844.208	1117798.691	29.56676	90.117587	0.150	-24.902	-24.866	-0.036	-0.02	0.166	0.545
5-REF	122802.993	1118758.98	29.602026	90.107408	0.241	-25.001	-24.956	-0.045	-0.02	0.266	0.872
5-U-M	118836.763	1114464.775	29.566802	90.152009	0.342	-24.916	-24.894	-0.022	-0.02	0.344	1.130
5-U-VL	118002.974	1114253.124	29.559289	90.154256	0.192	-24.895	-24.875	-0.02	-0.02	0.192	0.630
5-U-L	118316.594	1113944.7	29.562194	90.157483	0.214	-24.905	-24.886	-0.019	-0.02	0.213	0.699

* C=Confined; U+Unconfined; H=High, M=Medium; L=Low; VL=Very Low; DEG=Degraded Reference; REF=Healthy Reference

** Average change in ellipsoid height at multiple points within project area (n=219, range: -0.18 to 0.21 m, as estimated with GEOCON 11, http://www.ngs.noaa.gov/GEOCON11/)

*** Elevation corrected for Geoid 12A and ellipsoil change = Geoid09 elvation + [(Δ ellisposid height) - (N Goid12A - N Geoid 09)]

Numerical calculation for station 1-C-H is as follows: 0.513 m +[(-0.02) - (-24.974-(-24.934)] = 0.533 m

Accuracy is approximately ± 0.03 m, based on standard errors for ellipsoid height and geoid height conversions.

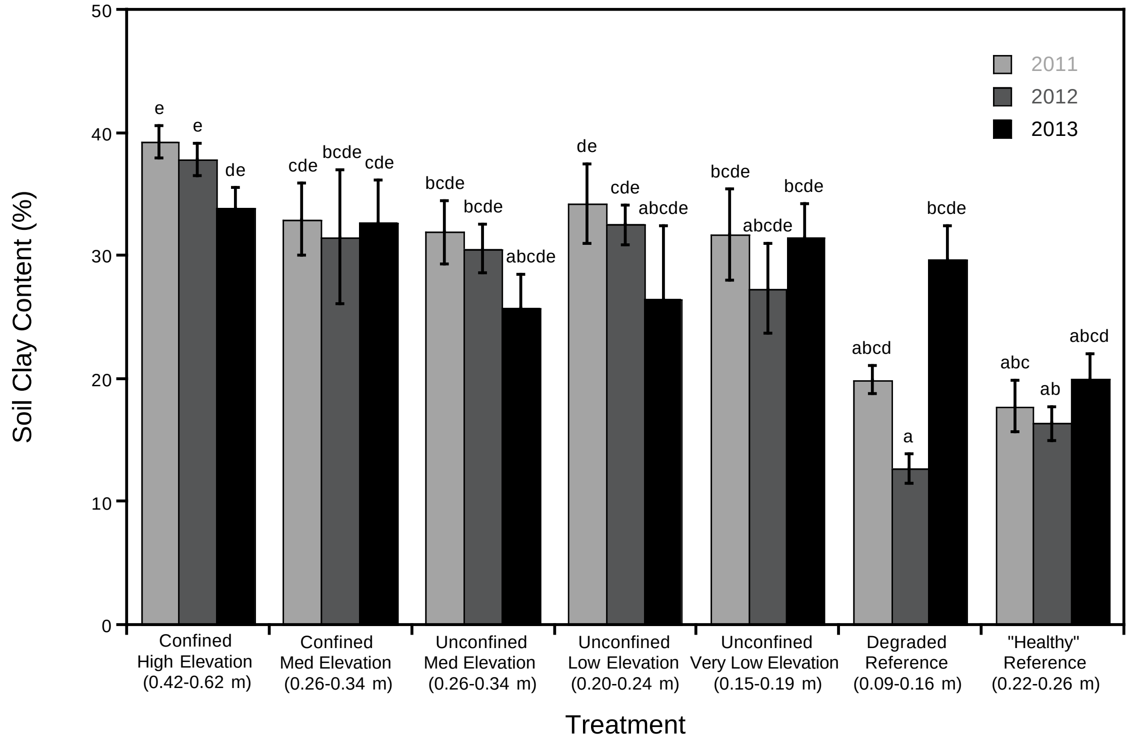


Figure A-1 - Treatment effects on soil clay content by sampling year (Fall 2011, Fall 2012, and Fall 2013). Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. Means with different letters identify significant differences within response variables.

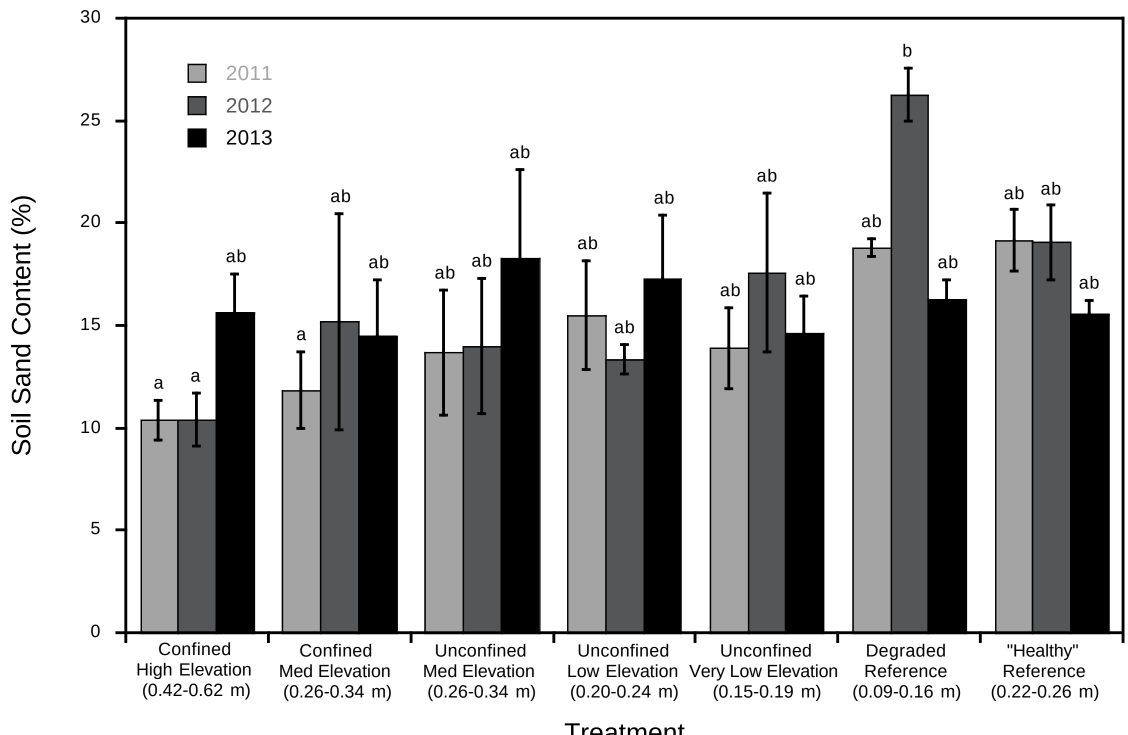


Figure A-2 - Treatment effects on soil sand content by sampling year (Fall 2011, Fall 2012, and Fall 2013). Values presented as the mean ± 1 SE (error bars) (n=5) for each treatment. Means with different letters identify significant differences within response variables.

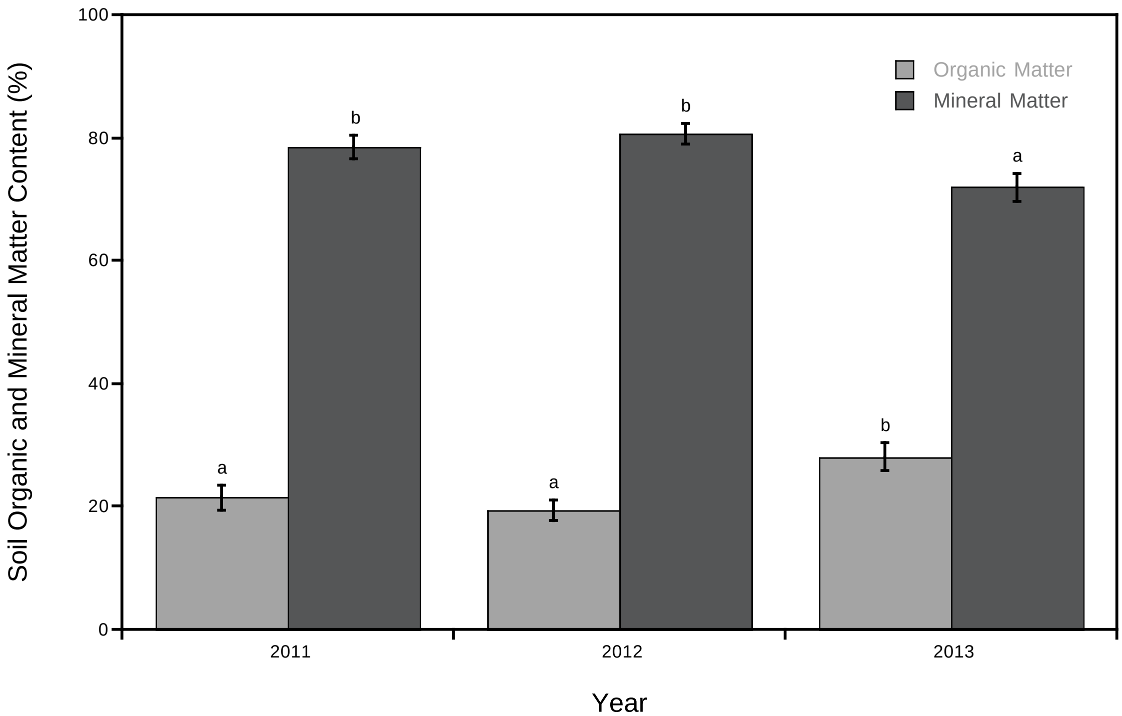


Figure A-3 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on soil organic and mineral content. Values presented as the mean ± 1 SE (error bars) (n=35) for each treatment. Means without letters in common are significantly different within response variables.

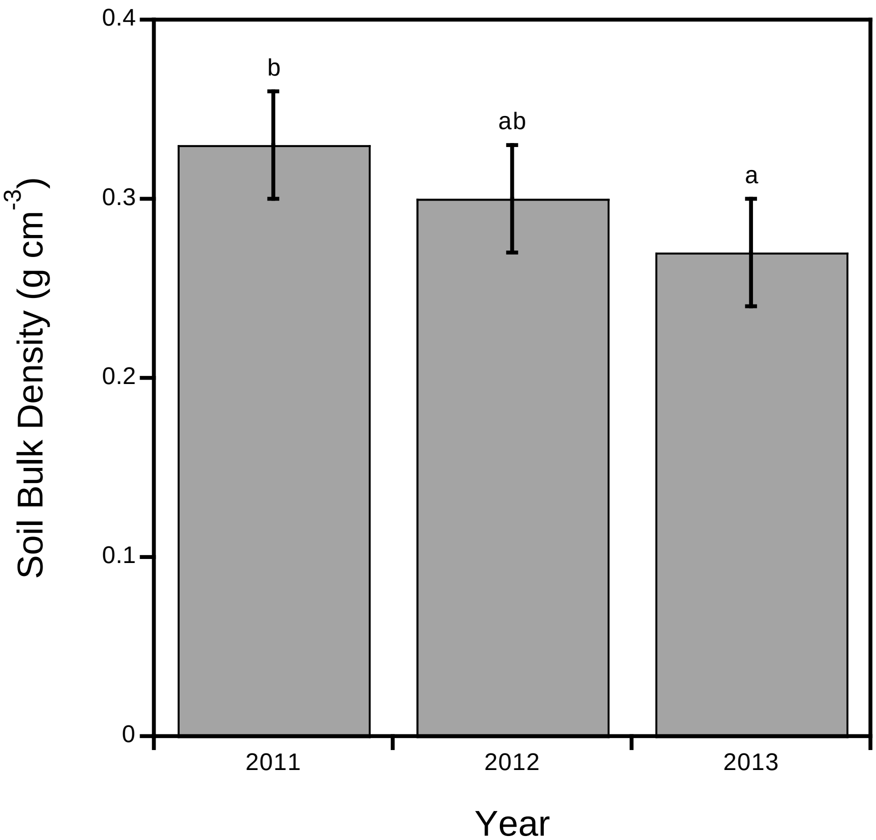


Figure A-4 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on soil bulk density. Values presented as the mean ± 1 SE (error bars) (n=35) for each treatment. Means without letters in common are significantly different.

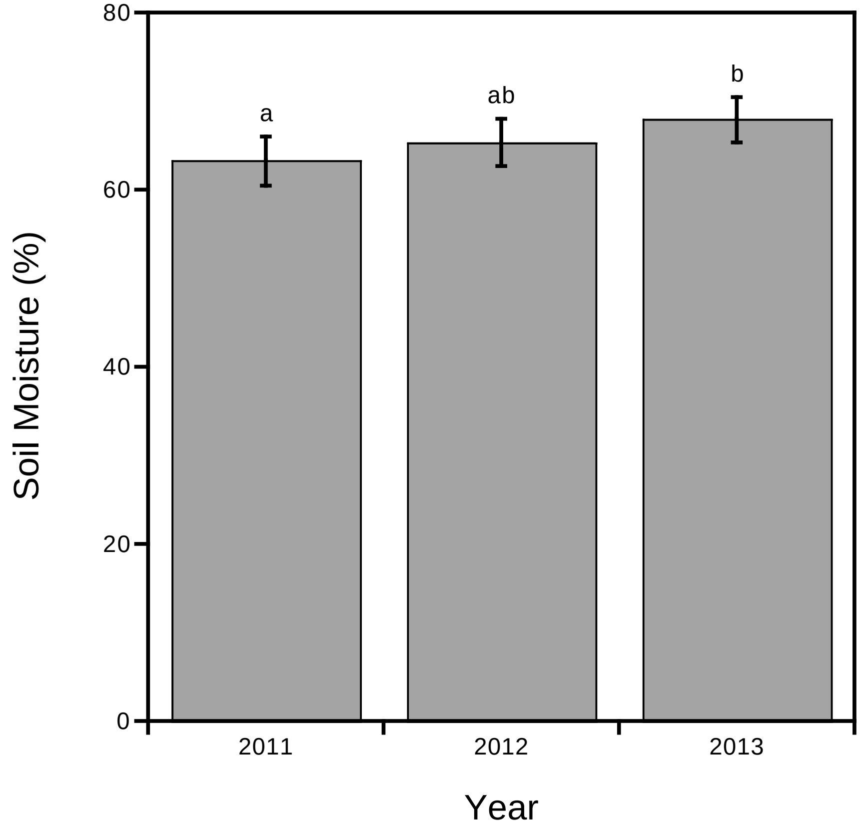


Figure A-5 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on soil moisture content. Values presented as the mean ± 1 SE (error bars) (n=35) for each treatment. Means without letters in common are significantly different.

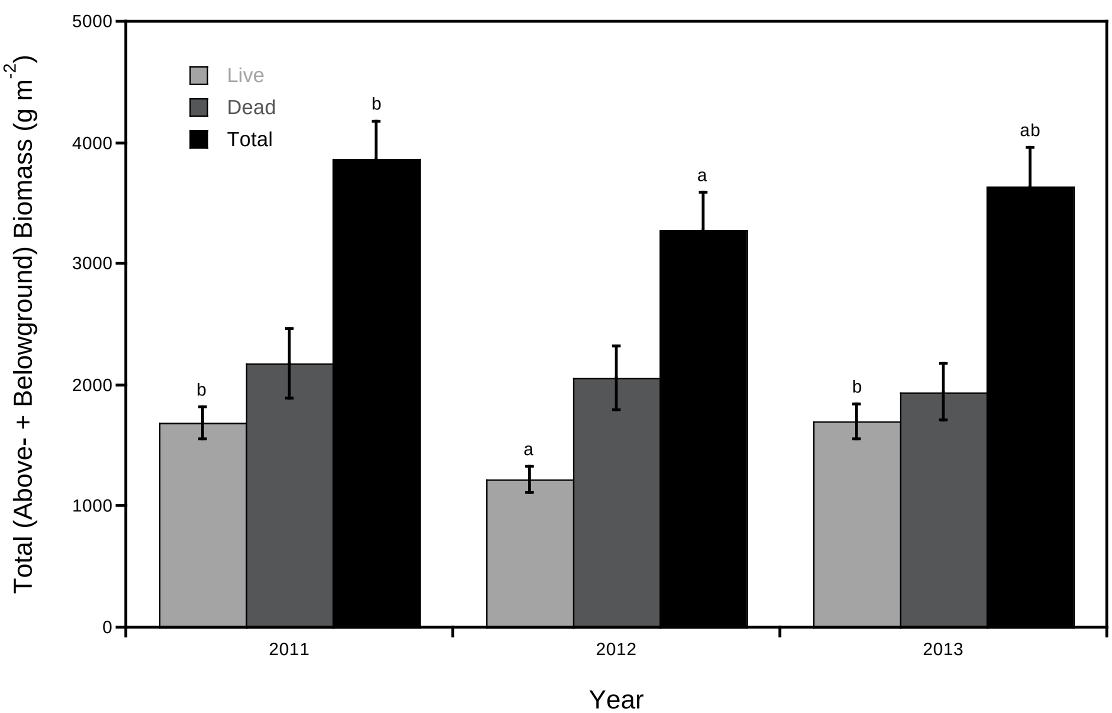


Figure A-6 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on live, dead, and total (above- and belowground) biomass. Values presented as the mean ± 1 SE (error bars) (n=35) or each treatment. Means without letters in common are significantly different within response variables.

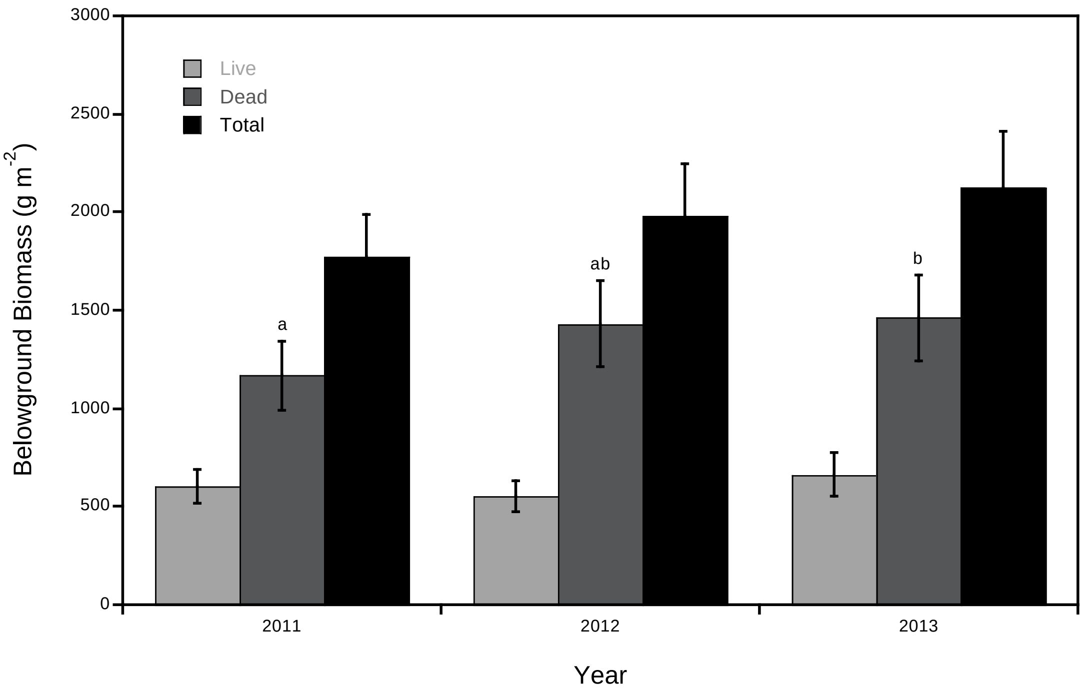


Figure A-7 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on live, dead and total (live + dead) belowground biomass. Values presented as the mean ± 1 SE (error bars) (n=35) for each treatment. Means without letters in common are significantly different within response variables.

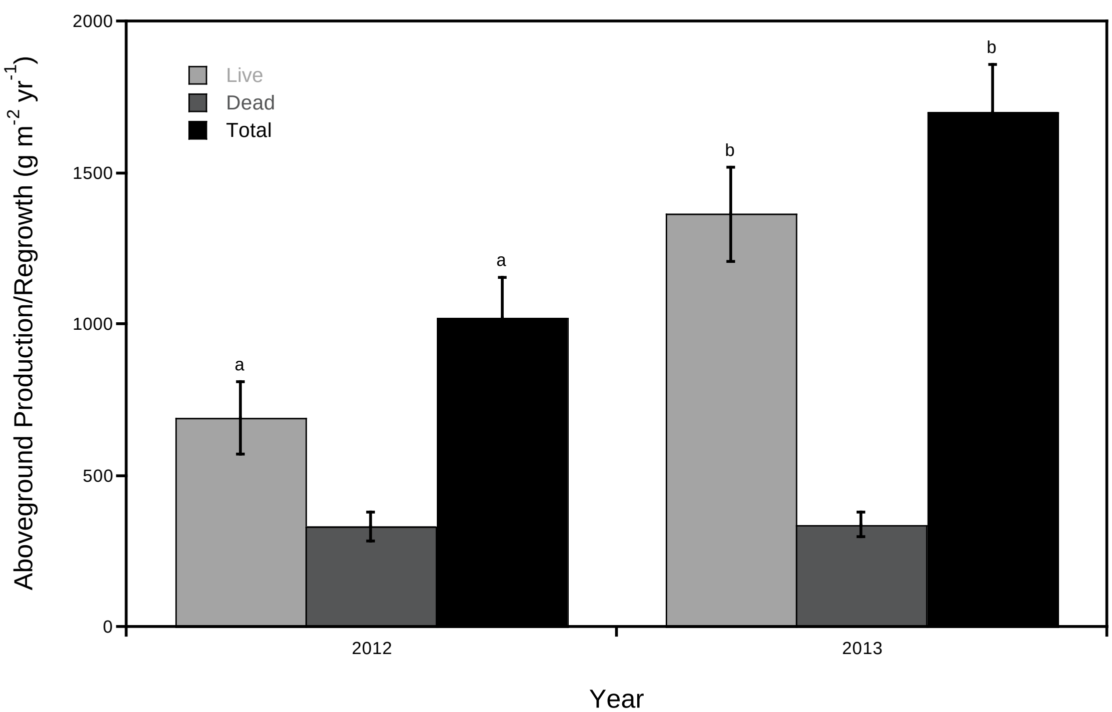


Figure A-8 - Effects of sampling year (Fall 2011, Fall 2012, and Fall 2013) on live, dead and total (live + dead) aboveground production. Values presented as the mean ± 1 SE (error bars) (n=35) for each treatment. Means without letters in common are significantly different within response variables.