2006

Salt marsh restoration with sediment-slurry amendments following a drought-induced, large-scale disturbance

Angela Marie Schrift
Louisiana State University and Agricultural and Mechanical College

SALT MARSH RESTORATION WITH SEDIMENT-SLURRY AMENDMENTS FOLLOWING A DROUGHT-INDUCED, LARGE-SCALE DISTURBANCE

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agriculture and Mechanical College
in partial fulfillment of the
requirements for the degree of
Master of Science

In

The Department of Oceanography and Coastal Sciences

by
Angela Marie Schrift
B.S., University of West Florida, 2002
August 2006

ACKNOWLEDGEMENTS

I would like to thank my major professor, Dr. Irving Mendelssohn, for his encouragement and support throughout my tenure at LSU. He has helped me accomplish tasks I never thought possible. I would also like to thank my committee members Dr. Robert Gambrell and Dr. Jaye Cable for their help and guidance throughout my project. I would also like to extend my thanks to Mike Breithaupt, Gretchen Brown, and Daniel Bond from the Soil Testing Laboratories. The statistical advice that I received from Dr. James Geaghan, Dr. Matthew Slocum, Susanne Hoeppner, Rebecca Christofferson, and Lisa Morris was invaluable. I would also like to give a special thanks to Mike Materne for all his work on this project. Additionally, I would like to thank all the research associates and graduate students who participated in various stages of this project: Alan Shadow, Brian Orth, Ashley Wilson, Lee Stanton, Kristen Laursen, Josh Roberts, Matthew Slocum, Hongjun Chen, Sean Graham, Camille Stagg, Luke Difulco, Carey Perry, Kevin Boswell, Whitney Broussard, Emily Hyfield, Hillary Collis, and Mike McDonough. The Department of Oceanography and Coastal Science and the Baton Rouge community has made my time at LSU one of the most rewarding experiences of my life. Funding for this research from the National Oceanographic and Atmospheric Administration through the Louisiana Department of Natural Resources was greatly appreciated.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

A large-scale, drought-induced disturbance occurred in Louisiana during the spring and summer of 2000. Approximately 100,000 acres of Spartina dominated marshes died-back and turned brown. This die-off caused considerable concern because in the absence of recovery dieback marshes can transform to mudflats, which can subside leading to open ponds. The state of Louisiana is attempting to restore some of the dieback marshes through the addition of sediment-slurries. The sediment-slurry generated significantly different marsh elevations: high elevation (mean and 95 % confidence interval: 29, 26 to 32 cm above ambient marsh), medium elevation (21, 19 to 24 cm), low elevation (14, 11 to 16 cm), pop-up (36, 32 to 40 cm), and vegetated (20, 17 to 22 cm), which were compared to reference marshes: healthy marshes (4, -1 to 9 cm) and dieback marshes (-2, -6 to 3 cm). High and medium elevations had minimal recovery two years following the slurry addition. These areas had plant cover similar to the reference dieback marshes, which did not receive the sediment-slurry amendment. The low elevation, popup (highly organic sections of the original substrate that detached during slurry application and settled on top of the sediment-slurry), and vegetated (dieback areas that recovered by the start of the study) areas that received the sediment-slurry had rapid plant recovery. Two years following the slurry addition, vegetation structure in the low and vegetated areas was the most similar to reference healthy marshes in plant cover (~100 %) and species richness (~1.25); pop-ups had the highest species richness (2.35, 1.8 to 2.9). Marshes that did not receive the sediment-slurry amendments were more frequently flooded and had higher sulfide concentrations (~1 mM) than marshes that received the sediment-slurry. Soil salinity was similar throughout the study site and did not limit plant recruitment. Rapid recovery was

governed by optimal inundation, high organic matter content concurrent with high elevation, and/or rhizome survivability following burial. If applied appropriately, sediment-slurry amendments can restore salt marshes that have subsided as a result of a drought-induced disturbance or other events that cause a lowering of marsh surface elevation.

INTRODUCTION

Disturbance drives the structure of many ecosystems (Foster et al. 1998, Turner et al. 2003, Walker and Del Moral 2003, Callaway 2005, Nieuwstadt and Sheil 2005). The severity of some disturbances can drastically alter the landscape and cause primary or secondary succession to be reinitiated. For example, the eruption of Mount St. Helens in 1980 violently impacted the adjacent volcanic plain with searing blasts and pumice and tephra deposits. This disturbance was so severe that all plant life was effaced and seven years later only a few plants had recolonized the area (Del Moral and Wood 1993). Although, succession on denuded landscapes has been extensively investigated (Del Moral and Wood 1993, Shumway 1994, Allison 1996), ecological trajectories are still difficult to accurately predict, in part, because plant recruitment is often dependent on stochastic processes (Del Moral and Wood 1993).

During the spring and summer of 2000, a record drought in the northern Gulf of Mexico caused a severe, large-scale disturbance of coastal salt marshes. This event, known as the "brown marsh phenomenon," resulted in the sudden dieback of large expanses of *Spartina alterniflora* dominated salt marshes (hereafter referred to as "brown marshes") (McKee et al. 2004) within the Mississippi River Deltaic Plain (MRDP), Louisiana. Approximately 28% (44,500 ha) of intertidal salt marshes in southeast Louisiana were severely affected by the brown marsh event (http://www.brownmarsh.net/qa.htm). Although some salt marshes did revegetate, many became mudflats and are still devoid of vegetation (I. A. Mendelssohn, personal observation, 2006). The loss of live plant material from the substrate can lead to soil compaction and marsh subsidence as plant roots collapse and organic material in the soil

decomposes (DeLaune et al. 1994). Unvegetated mudflats may convert to shallow ponds as the marsh surface further subsides and erodes.

An understanding of how disturbances control succession is a pre-requisite to successful restoration of disturbed ecosystems. A plethora of research has been conducted on disturbances (Pennings and Richards 1998, Brinson and Christian 1999, Vankvik 2004) resulting in the development of assembly rules that predict plant succession subsequent to small patch formation (Wu and Levin 1994, Gutzerova and Herben 2001, Platt and Connell 2003). However, these assembly rules do not often apply to patches created from large, severe disturbances (Turner and Dale 1998). Although there has been some research on plant reestablishment following large, severe disturbances (Turner and Dale 1998), the low frequency of mega-disturbances has prevented detailed research in a variety of ecosystems. To my knowledge, there have been no studies of primary or secondary succession following a sudden, large-scale, severe disturbance in a salt marsh. The size and severity of the brown marsh event of 2000 provided the opportunity to evaluate effects of a mega-disturbance, to quantify primary and secondary succession, and to assess the restoration potential of mudflats created by the die-off.

Because wetlands in the MRDP are rapidly subsiding and subsidence rates were likely accelerated at dieback sites by the brown marsh event (DeLaune et al. 1994), the hydraulic application of sediment-slurries, a relatively new wetland restoration technique (Closure Report: Initial Funding Allocation, DNR Dedicated Dredging Program (LA-1) 2000, Mendelssohn and Kuhn 2003, Slocum et al. 2005), was tested in this research to increase mudflat elevation and to stimulate restoration. This restoration technique utilizes low-pressure hydraulic dredging to disperse sediments relatively long distances

(ca. 900 m) from the discharge pipe (Cheramie et al. 1995, Slocum et al. 2005). I used this application technique to test how different levels of sediment addition effect plant recruitment, vegetation recovery, and successional trajectories.

Although the cause of the brown marsh event has not been unequivocally identified (McKee et al. 2004), the brown marsh event provided the unique opportunity to determine if a sediment-slurry subsidy could rehabilitate these degraded marshes, located in a region with very high rates of relative sea-level rise (Penland and Ramsey 1990). The objective of this study was to determine if sediment-slurry application could create a substrate and a suitable elevation that would allow successful restoration of the brown marsh sites. I investigated how initial plant recruitment differed between an experimental area that received sediment-slurry amendments and adjacent reference areas that did not receive any additional sediment. Also, soil physico-chemical properties were measured to evaluate their control on plant recolonization and vegetation recovery. I sought to answer the following questions: (1) What hydro-edaphic factors control vegetation recruitment and recovery, with and without sediment amendments, after sudden marsh dieback? (2) Do sediment-slurry amendments accelerate vegetation restoration? If so, 3) Is the speed of recovery directly related to the degree of sediment addition?

METHODS

Site Description

The study site is in a rapidly subsiding salt marsh located within the Terrebonne Basin and is part of the Mississippi River Deltaic Plain (MRDP). A reduction of sediment input resulting from a combination of delta lobe switching, canal construction, hydrologic alterations, and artificial levees (Day et al. 1993) has resulted in high rates of relative sea level rise (1.11cm/yr, 1947-1986) according to (Penland and Ramsey 1990) The diminished sediment supply has reduced the capacity of the marsh to keep pace with relative sea-level rise (a combination of eustacy and isostacy) (Penland et al. 1990). This process has facilitated high rates of land loss, averaging 77 km^2/year from 1978 to 2000. Although net land loss rates are expected to decrease over the next 50 years, they are predicted to remain high at 26 km^2/year (Barras et al. 2004).

The specific study site is located within a salt marsh approximately 8 kilometers south-southwest of Leeville, LA, adjacent to the west bank of Bayou Lafourche at 29º 10' 58.88"N and 90º 14' 23.01"W (NAD 1983, UTM Zone 15). Soils are characterized as a scatlake muck, which is a semifluid, mineral soil that is frequently flooded with salt water (Matthews 1984). Vegetation in reference healthy areas is dominated by *Spartina alterniflora* and is sparsely interspersed with *Salicornia virginica*. Large expanses of vegetation within the study site were denuded by the dieback event of 2000 and remained unvegetated while other adjacent areas remained unaffected by the dieback event (McKee et al. 2004).

Experimental Design

The study site was divided into an experimental area and a reference area. The experimental area consisted of a salt marsh that died as a result of the brown marsh event

of 2000. This impacted marsh was divided into five cells, described below, and each cell independently received hydraulically dredged material from Bayou Lafourche in the form of a sediment-slurry. The sediment-slurry was approximately 20-30 % solids and 70-80 % water by volume (Brian Kendrick, personal communication, Morris P. Hebert, Inc., Houma, LA 2005). The experimental area was bordered and divided into five cells by small earthen levees, four of which were used in this research as statistical blocks (Fig. 1). The cells provided replicated brown marshes that independently received sediment-slurry amendments. Cells were hydraulically connected with culverts and breaks in the levees provided tidal exchange. Five different conditions were created by the application of the sediment-slurry and were used as treatment-levels: 1) Low elevation: 13-18 cm above ambient healthy marsh and unvegetated in the fall of 2003, 2) Medium elevation: 20-25 cm above ambient healthy marsh and unvegetated in the fall of 2003, 3) High elevation: 28-36 cm above ambient healthy marsh and unvegetated in the fall of 2003, 4) Vegetated: areas with 100 % vegetative cover in the fall of 2003 with an average elevation of 20 cm above ambient healthy marsh, and 5) Pop-up: portions of the original substrate consisting of a thick root and rhizome mat which, separated from the underlying substrate, became buoyant, and settled on top of the sediment-slurry amendment resulting in an average elevation of 36 cm above ambient healthy marsh. The formation of pop-ups during the application of the sediment-slurry is a common occurrence when sediment-slurries are added to a confined site (Brian Kendrick, personal communication). High, medium, low elevation and vegetated treatment-levels were identified in each of the four cells while the pop-up treatment-level occurred in two cells. Ten sampling transects, 2.75 m in length were established within each high, medium, and low elevation treatment-level. In each vegetated area, I established seven sampling

transects, 2.75 m in length. Each pop-up area had ten sampling transects, 2.00 m in length for a total of 168 haphazardly placed, experimental sampling transects.

To assess the effectiveness of the sediment-slurry amendment, I compared the experimental area with two different types of reference marshes which did not receive the sediment-slurry amendment: 1) Healthy marsh: unaffected by the brown marsh event of 2000 and dominated by *Spartina alterniflora* and interspersed with *Salicornia virginica*, and 2) Brown marsh: denuded as a result of the brown marsh event of 2000 and remained unvegetated through the fall of 2003 (Fig. 1). The two healthy marsh sites and the two brown marsh sites were located adjacent to the experimental area to minimize spatial variability. Within each reference marsh I haphazardly established, ten sampling transects, 2.75 m in length, for a total of 40 reference sampling transects.

Elevation and Hydrological Measurements

I used a laser level (Sokkia LP30) to determine elevation in both the experimental and reference areas. Initially I installed a permanent benchmark and referenced it to a common datum (April 2003), which was the average elevation in the surrounding healthy marshes. All future transect elevation measurements were referenced to the common datum. Average elevations of each transect treatment-level and reference marsh type were determined in summer 2004. On average, reference healthy marsh transects were 4.0 cm higher than the common datum, while reference brown marsh transects were 1.5 cm below the common datum. All other transects were above the common datum. Sediment-slurry thickness was determined by measuring from the marsh surface (top of the sediment-slurry layer) to the top of the former substrate (bottom of the sediment-slurry layer). The top of the former substrate was easily identifiable because the former marsh surface was highly organic comprised of peat and root mass.

A water sonde (YSI 600LS, YSI Inc., Yellow Springs, OH) was installed in a bayou adjacent to the study site to record water depth from August 2004 until December 2004. All water depths were referenced to the common datum. Flooding in the marshes was determined relative to the water depth in the channel. Thus, to be classified as flooded, water depth in the channel had to be greater than the elevation of the respective treatment-level or reference marsh type.

Plant and Soil Physico-Chemical Measurements

I analyzed vegetation parameters during the fall of 2003, spring of 2004, and fall of 2004 to assess initial plant recovery. Stem density was measured within a 0.1 m^2 quadrat at five randomly chosen points along each transect for all treatment-levels except pop-ups. The short transect length on pop-ups forced us to systematically select five sampling points to avoid overlapping quadrats. Stem density (stems/m^2) was calculated by summing the stem counts from the five sampling points and multiplying by two. Frequency of occurrence was calculated by dividing the number of times species A intersected one of ten fixed points along each transect by the total number of fixed points on a transect. Percent of unvegetated transect was calculated by dividing the total length of transect devoid of vegetation by total transect length and multiplying by 100. Plant cover was then calculated by subtracting the percent of unvegetated transect from 100. The influence of each species within a treatment-level/reference marsh type was rated with an importance value. To calculate the importance value, I first determined species cover by dividing the total distance species A covered the transect by total transect length and multiplying by 100. Relative species cover was then calculated by dividing the cover of species A by the total cover of all species and multiplying by 100. Next, I determined relative frequency by dividing the frequency of species A by the frequency of all species

and multiplying by 100. Finally, I determined relative density by dividing the density of species A by the density of all species and multiplying by 100. Once these three relative values were determined, I added relative species cover, relative frequency, and relative density together to determine the importance value. Because I added three relative values together, which separately had a maximum value of 100, the maximum importance value was 300.

Several soil physico-chemical variables were measured during each sampling period (fall of 2003, spring of 2004, and fall of 2004). Two adjacent soil cores were taken at a haphazardly chosen location along each transect. At the time of soil coring, I used three bright platinum electrodes and a calomel reference electrode to measure redox potential at 15 cm depth along each transect. An average of the three readings was used for statistical analysis. The smaller soil core (5 cm in diameter x 10 cm in length) was used to determine bulk density, organic matter, percent moisture, electrical conductivity, and particle size. In the laboratory, the core was weighed, placed in a drying oven at 65 °C until a constant weight was reached, and weighed again to determine dry bulk density and percent moisture. Electrical conductivity was determined by shaking 5 grams of dry soil with 30 mL of distilled water for one hour in a centrifuge tube which was subsequently centrifuged (Suprafuge 22, model 6415 Heraeus Sepatech GmbH, Germany) at 6000 rpm for five minutes, decanted, and measured for electrical conductivity (Cole Parmer 19820-00, Vernon Hills, IL). To determine organic matter content, approximately 2-3 grams of dry soil was treated with 1N HCl until all of the inorganic carbonates were volatilized. The soil was then analyzed for percent organic matter using the loss on ignition method (Nelson and Sommers 1996). The remaining portion of the cores were consolidated, homogenized, and analyzed for particle size using

the pipet method (Soil Survey Laboratory Investigations Manual 2004). Once collected, the second soil core (5 cm in diameter x 15 cm in length) was immediately put in a ziplock bag and placed on ice. Once in the laboratory, the soil core was homogenized and analyzed for pH and extractable NH_4-N, NO_3-N, P, and exchangeable Ca, Mg, K, Na, Fe, Mn, Cu, and Zn. The soil extractions were as follows: NH_4-N and NO_3-N with 2 M KCl (Bremner and Kenney 1966); P with Bray-2 (Byrnside and Sturgis 1958); Ca, Mg, K, Na with ammonium acetate (Thomas 1982); Fe, Mn, Cu, Zn with DTPA (Lindsay and Norvell 1978). Following extraction, NH_4-N and NO_3-N samples were filtered through a 0.45 μm syringe filter and were measured using a segmented flow AutoAnalyzer (Flow Solution IV AutoAnalyzer, O-I Analytical, USA). The remaining extracts were measured with an inductively coupled argon plasma emission spectrometer (ICP) (Spectro Ciros CCE, Spectro Analytical Instruments, Germany). During the fall of 2004, an additional soil core (2 cm in diameter x 10 cm in length) was taken adjacent to the other two cores to measure interstitial sulfide concentration. The soil core was immediately placed in an air-tight centrifuge tube (50 cm^3) with a septum cap, purged with N_2 for a minute, and placed on ice. In the laboratory, these samples were centrifuged (International Centrifuge, Model V, International Equipment Co., Boston, MA) at 6000 rpm for ten minutes. The interstitial water was immediately decanted, placed into an antioxidant buffer, and analyzed for total soluble sulfide (Sulfide electrode model DJM-146, Lazar Research Laboratories, Los Angeles, CA, USA).

Statistical Methods

Because the experimental design for the experimental area was different from that of the reference area, the data from the two designs were analyzed separately with different models. The experimental area was analyzed as a randomized block design with

the cells (four of the five were used in this study) serving as statistical blocks (replicates). The reference area was analyzed as a nested completely randomized design. I analyzed the vegetative and soil physico-chemical data from the experimental area and reference area with separate one-way ANOVAs using the PROC MIXED procedure of SAS version 9.1 (Base SAS 9.1 Procedures Guide 2004). In the experimental area, I tested the effect of treatment-level, time, and the treatment-level by time interaction. In the reference area, I tested the effect of reference marsh type, time, and the reference marsh type by time interaction. Model residuals were tested for normality (Shapiro-Wilks test) and homogeneity of variance (plot of residuals). Where necessary, log, natural log, arc-sine root, square, and square-root transformations were used to improve normality and homogeneity of variance.

The dimensionality of the experimental and reference soil (not vegetation) data sets was reduced by a Principal Components Analysis (PCA) using the FACTOR procedure of SAS version 9.1 (Base SAS 9.1 Procedures Guide 2004) with a varimax rotation. First, I combined the experimental and reference datasets to run the PCA. Following the PCA, I separated the data back into the experimental and reference datasets and then analyzed the principal components with one-way ANOVAs to determine the effect of treatment-level/reference marsh type, time, and the treatment-level/reference marsh type by time interaction on the soil principal components. Elevation, which was measured during one sampling period (spring 2004), was assumed to remain constant over the one-year study and was included in the PCA. Since organic matter was only measured during one sampling period, values for the missing sampling periods were estimated through a regression with bulk density and included in the PCA. Soil variables with high temporal variability that were not measured during all sampling periods, such

as NH_4-N, and NH_4-N + NO_3-N, were analyzed separately, and not included in the PCA of the soil data.

Soil physico-chemical variables were analyzed separately with a MANOVA using the PROC GLM procedure of SAS version 9.1 (Base SAS 9.1 Procedures Guide 2004). Highly significant effects for treatment level/reference marsh type, time, and treatment level/reference marsh type by time interactions (experimental area: $p < 0.0001$; reference area: $p = 0.0001$) enabled us to use univariate ANOVAs to analyze the individual soil environmental variables in both the experimental and reference areas.

Differences between treatment-levels/reference marsh types, time periods, and treatment-levels/reference marsh types by time interactions were tested with post-hoc, Tukey-adjusted pairwise comparisons. All tests of significance used an alpha level of 0.05 unless otherwise stated. Least-square means and confidence intervals are reported and graphed in their original units. The data from the experimental and reference areas were compared using 95 % confidence intervals. In addition, relationships between variables were analyzed with a correlation analysis using the PROC CORR procedure of SAS version 9.1 (Base SAS 9.1 Procedures Guide 2004).

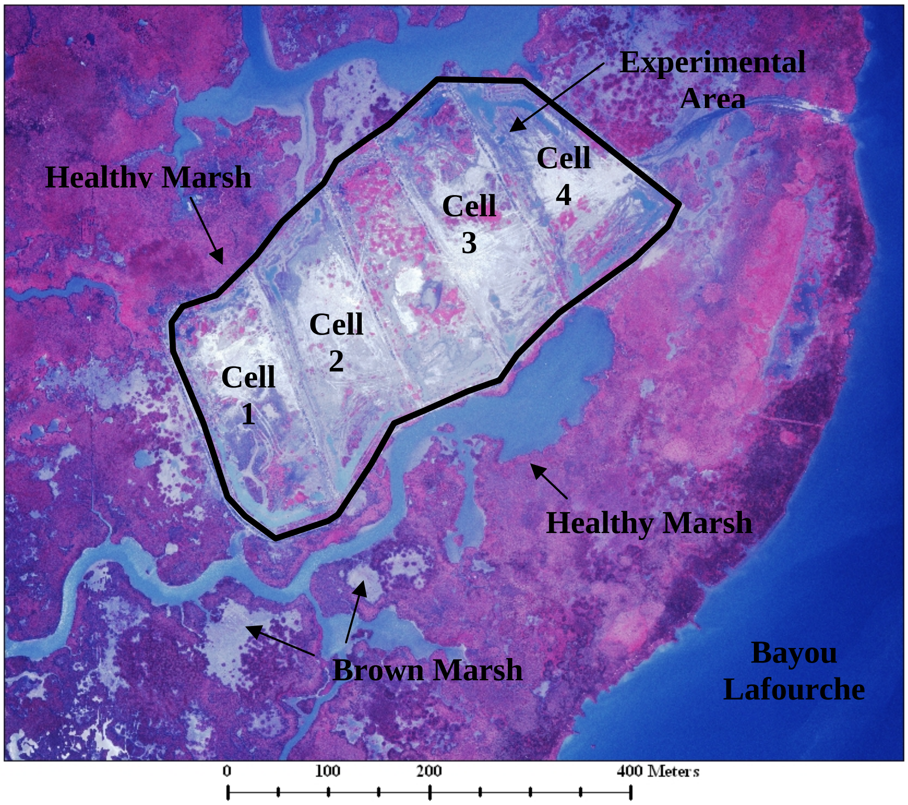


Figure 1. The geographical relationship between the experimental area (cells 1, 2, 3, and 4) and the reference areas (brown marshes and healthy marshes).

RESULTS

Vegetation Parameters

Vegetation in the experimental area showed a positive response to the slurry addition (Figs. 2, A1). The extent of vegetation development over time was dependent on the sediment-slurry treatment-level (treatment-level by time interaction, $p < 0.0001$; Fig. 2). The low elevation and pop-up treatment-level had significant increases in plant cover over time while the high, medium, and vegetated treatment-levels did not demonstrate increases in recovery during the study period. By the fall of 2004, three of the five treatment-levels (low elevation, vegetated, and pop-up) had plant cover equal to that of the healthy reference marshes. In contrast, the reference brown marsh sites, which received no sediment-slurry additions, showed minimal plant cover, i.e., recovery (Fig. 2). The high elevation treatment-level had as little plant cover as the reference brown marsh sites. The medium elevation treatment-level had in an intermediate degree of plant cover, which was significantly more than the reference brown marsh sites (92 % confidence interval, Fig. 2).

Species richness, like plant cover, increased with recovery duration, but this increase significantly differed with sediment-slurry treatment-level (significant treatment-level by time interaction, $p < 0.0001$, Fig. 3). Species richness in the medium, low, and pop-up treatment-levels increased over time while species richness in the high and vegetated treatment-levels remained constant. Species richness in the medium and low elevation, vegetated, and pop-up treatment-levels was equal to that of the reference healthy marsh sites by the spring of 2004 (Fig. 3). Conversely, low species richness occurred in the high elevation treatment-level and reference brown marsh sites almost

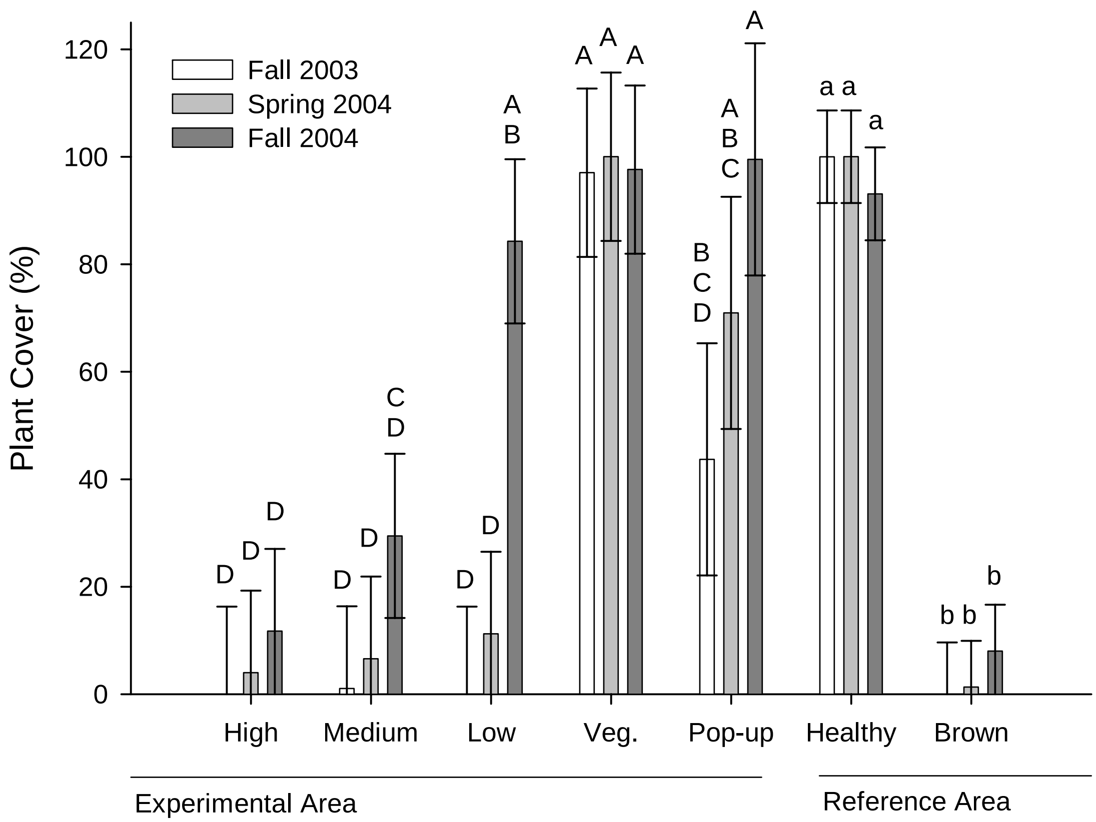


Figure 2. Percent vegetative cover (live and dead) over time in sediment-amended and reference marshes (least-square mean with 95% confidence intervals) following sediment slurry addition in 2002. The same letters indicate no significant differences between treatment means within either the experimental or reference areas ($p < 0.05$). Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas.

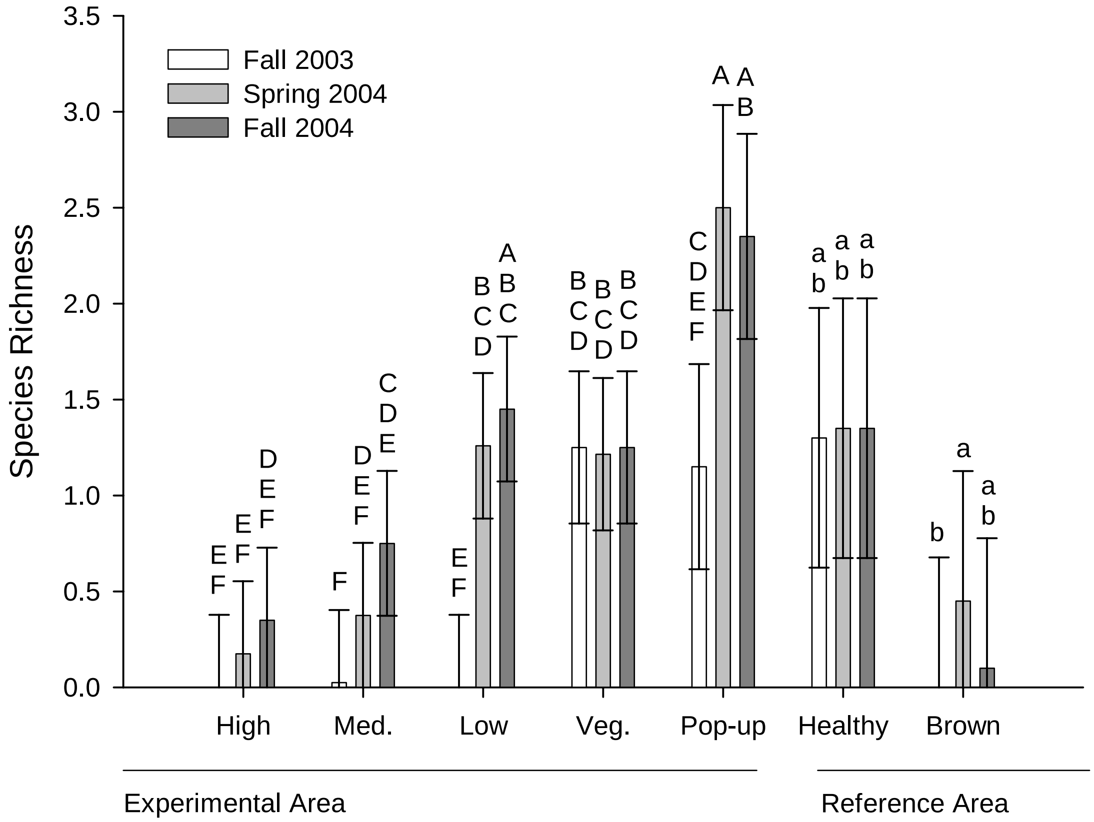


Figure 3. Species richness over time in the experimental area that received sediment-slurry amendments and in reference areas that did not receive sediment-slurry amendments (least-square mean with 95 % confidence interval). The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas.

Table 1. Presence (+) or absence (-) of live plant species.

	Experimental Area					Reference Area	
Plant species	High	Medium	Low	Vegetated	Pop-up	Healthy	Brown
Avicennia germinans	-	-	-	-	+	-	-
Batis maritima	-	-	-	+	+	-	-
Blutaparon vermiculare	-	-	-	+	+	-	-
Cyperus oxylepis	-	-	-	-	+	-	-
Distichlis spicata	-	-	-	-	+	-	-
Salicornia bigelovii	+	+	+	-	+	-	-
Salicornia virginica	+	+	+	+	+	+	-
Sesuvium maritimum	-	-	-	-	+	-	-
Spartina alterniflora	+	+	+	+	+	+	+

two years after the sediment-slurry addition (Fig. 3). The pop-up treatment-level had the highest species richness (Table 1, Fig. 3).

Importance values (Table 2), which are indices of species dominance, were determined for all live species (Table 1) within each sediment-slurry treatment-level (SSTL) and reference marsh type. *Spartina alterniflora* was the most important initial colonizer and, overall, the dominant species within the experimental and reference areas (Table 2). The pop-up treatment-level, which had the highest species richness, was also dominated by *Spartina alterniflora;* the succulent, *Blutaparon vermiculare,* and the salt marsh grass, *Distichlis spicata*, were important subdominants (Table 1, Fig. 4).

Depending on treatment-level, the importance value of some species increased over time while others decreased (significant treatment-level by time interaction, Table 3). From the fall of 2003 to the fall of 2004, the importance values for *S. alterniflora* increased in the high, medium, low, and pop-up treatment-levels, and reference brown marsh sites (Fig. 5). By the fall of 2004, the low elevation and vegetated treatment-levels had *S. alterniflora* importance values equivalent to that of the reference healthy marsh sites (Fig. 5). In the pop-up treatment-level, the dominance of *S. alterniflora* increased over time, as did that of *D. spicata* and the succulent *Salicornia virginica* (Fig. 4). In contrast, other species, like *Blutaparon vermiculare*, decreased over time. By the fall of 2004, *S. alterniflora* was the overwhelming dominant of the pop-up treatment-level (Fig. 4). Due to the limited species richness and abundance in treatment-levels other than the pop-up, the overall time effect followed the trends of species dominance over time in the pop-up treatment-level (Table 3). *Sesuvium maritimum* was only present in the pop-up treatment-level during the spring of 2004 and hence, the overall effect of time (Table 3) indicated that the spring of 2004 had significantly higher importance values

Table 2. Importance value for each live plant species two years (fall 2004) following the sediment amendment (least-squared mean with 95 % confidence interval). Values are not listed for species that were absent (n/a) within a treatment-level or reference marsh type. The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant.

Experimental Area	*Avicennia germinans*	*Batis maritima*	*Blutaparon vermiculare*	*Cyperus oxylepis*	*Distichlis spicata*
High	n/a	n/a	n/a	n/a	n/a
Medium	n/a	n/a	n/a	n/a	n/a
Low	n/a	n/a	n/a	n/a	n/a
Vegetated	n/a	1.65 A (-1.44 to 4.75)	n/a	n/a	n/a
Pop-up	0.02 (-0.14 to 0.18)	7.99 A (4.01 to 11.96)	14.95 (1.68 to 28.21)	2.35 (-0.80 to 5.50)	28.49 (16.44 to 40.53)
Reference Area					
Healthy	n/a	n/a	n/a	n/a	n/a
Brown	n/a	n/a	n/a	n/a	n/a

Experimental Area	*Salicornia bigelovii*	*Salicornia virginica*	*Sesuvium maritimum*	*Spartina alterniflora*
High	n/a	0.29 B (-0.17 to 1.00)	n/a	32.38 B (-20.51 to 85.27)
Medium	n/a	0.43 B (-0.08 to 1.20)	n/a	129.92 B (77.03 to 182.81)
Low	n/a	0.88 B (0.21 to 1.91)	n/a	296.34 A (243.45 to 349.23)
Vegetated	n/a	0.87 B (0.51 to 2.03)	n/a	286.46 A (231.58 to 341.35)
Pop-up	0.44 (-3.83 to 4.70)	13.84 A (7.00 to 26.52)	n/a	131.25 AB (56.45 to 206.04)
Reference Area				
Healthy	n/a	10.21 (-2.37 to 22.78)	n/a	288.31 a (217.44 to 369.16)
Brown	n/a	n/a	n/a	3.00 b (0.25 to 15.73)

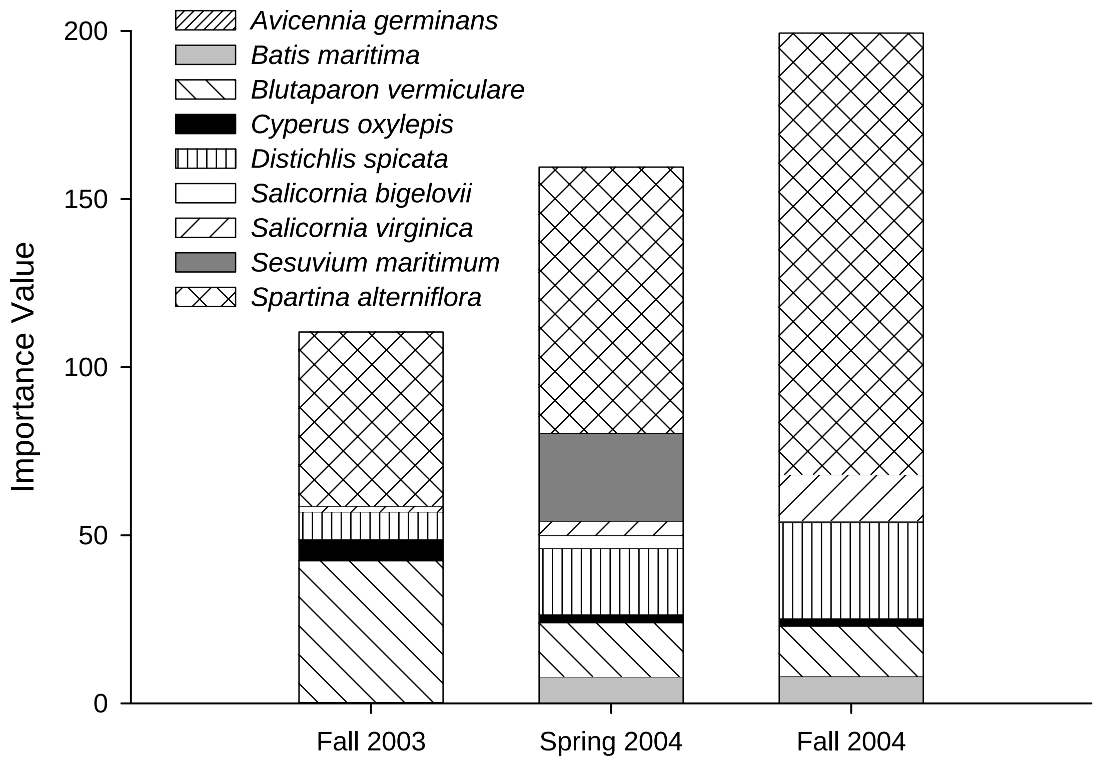


Figure 4. Plant importance over time in the pop-up treatment-level, an area with a high diversity (least-square mean).

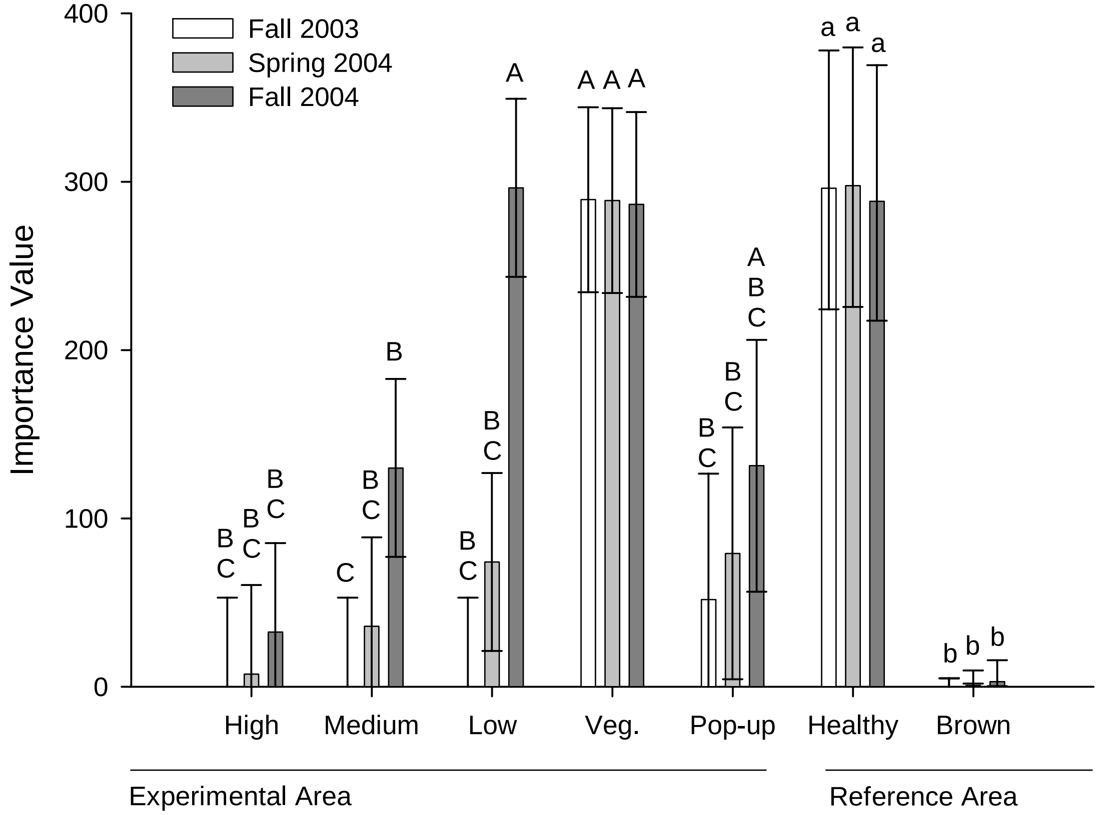


Figure 5. Importance of the primary initial colonizer, *Spartina alterniflora* (least-square mean with 95 % confidence interval). The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas.

Table 3. The effect of treatment level (TL) / reference marsh type (MT), time (T) (fall 2003, spring 2004, and fall 2004), and the interaction of treatment level/reference marsh type and time on the importance of a plant species. Plants that were not present (n/a) could not be analyzed. Bold indicates significant differences.

	Experimental Area			Reference Area		
Plant Species	TL	T	TL*T	MT	T	MT*T
Avicennia germinans	0.1463	0.1041	0.1339	n/a	n/a	n/a
Batis maritima	0.1109	**0.0095**	**0.0389**	n/a	n/a	n/a
Blutaparon vermiculare	**0.0333**	**0.0067**	**0.0071**	n/a	n/a	n/a
Cyperus oxylepis	0.0887	0.4291	0.7541	n/a	n/a	n/a
Distichlis spicata	0.0816	0.0640	0.0651	n/a	n/a	n/a
Salicornia bigelovii	0.3963	0.1275	0.4077	n/a	n/a	n/a
Salicornia virginica	**0.0001**	**0.0009**	0.2660	0.4226	0.4444	0.4444
Sesuvium maritimum	**0.0016**	**<.0001**	**<.0001**	n/a	n/a	n/a
Spartina alterniflora	**<.0001**	**<.0001**	**<.0001**	**0.0029**	0.5307	0.3776

than the fall of 2003 or the fall of 2004 (data not shown). Similarly, the importance values of *Batis maritima* and *Salicornia virginica* in the pop-up treatment-level significantly varied over time (Table 3, Fig. 4).

Soil Physico-Chemical Characteristics

Principal Components Analysis. The PCA grouped 16 of the soil environmental variables into four principal components (PC1, PC2, PC3, PC4), which explained 86% of the variation (Table 4; Fig. 6). The factor scores of the principal components from the experimental area were significantly affected by treatment-level, time, and/or treatment-level by time interactions. However, principal component factor scores in the reference area only had one significant main effect and one significant interaction (Table 5).

Principal component 1 explained 52% of the variation of the soil environmental data and had high positive loadings for Mn, Zn, and Cu, and this principal component can be interpreted as a trace metal-related factor (Table 4). PC1 significantly varied among treatment-levels (Table 5). PC1 was significantly higher in the high and medium elevation treatment-levels than the low and vegetated treatment-levels. The pop-up treatment-level had the lowest PC1 factor score and was significantly lower than the reference marsh types. PC1 was similar between reference marsh types and the vegetated treatment-levels. PC1 was also statistically similar between the reference brown marsh and the low elevation treatment-level (Fig. 6a). There was also a significant treatment-level by time interaction for PC1's factor scores (Table 5). All SSTLs remained constant over time except for the vegetated treatment-level, which had a slight increase from the fall of 2003 to the spring of 2004 (data not shown). PC1 in the reference area had a significant time effect and a significant reference marsh type by time interaction (Table 5). PC1 was significantly lower in the fall of 2003 compared to the spring of 2004 and

the fall of 2004. PC1 decreased over time in the reference brown marsh sites while PC1 in the reference healthy marsh sites did not have a consistent trend over time.

PC2 explained 15 % of the variation and had high positive loadings for bulk density, Fe, and P and highly negative loadings for organic matter, moisture, and conductivity, and can therefore be interpreted as a mineral-related component. Treatment-level had a significant effect on PC2's factor scores (Table 4). PC2 was significantly lower in the high, medium, and vegetated treatment-levels than the low elevation treatment-level. The pop-up treatment-level, which had the lowest PC2 scores in the experimental area, and the high elevation treatment-level were statistically similar to the reference marsh types (Fig. 6b).

PC3 explained 10 % of the variation, had high positive loadings for Mg, K, Na, and can be interpreted as a salt-related component (Table 4). The factor scores from PC3 varied between treatment-levels ($p = 0.0286$; Table 5). PC3 was significantly higher in the vegetated treatment-level than in the low elevation treatment-level; all other treatment-levels were statistically similar. The reference marsh types were statistically similar to all treatment-levels in the experimental area (Fig. 6c). In addition, PC3 had a significant interaction between treatment-level and time (Table 5). Over time, factor scores from PC3 remained constant within all SSTLs except the high elevation treatment-level, which increased from the fall of 2003 to the spring of 2004 (data not shown).

PC4 explained 9 % of the variation and had high loadings for elevation and Eh, and can be interpreted as an inundation-related component (Table 4). Treatment-level significantly affected PC4 (Table 5). PC4's factor scores were highest in the pop-up treatment-level and in general, significantly decreased with decreasing elevation. PC4 significantly decreased from the pop-up to high elevation treatment-level and from the

high to low elevation treatment-level. The vegetated treatment-level was statistically similar to both the medium and low elevation treatment-levels. The factor scores for the reference marsh types were statistically similar to those for the low elevation treatment-level and were significantly lower than those for all other SSTLs (Fig. 6d). Time also significantly affected PC4 (Table 5). Factor scores were significantly higher in the spring of 2004 than the fall of 2003 and the fall of 2004 (data not shown).

Univariate Comparisons. The following results present responses of specific variables of interest that were either grouped into a principal component via the PCA or that could not be analyzed by the PCA because of the frequency of data collection. Because of their importance in interpreting plant response and recovery, I present them individually.

Sediment-slurry amendments significantly increased elevation within the experimental area (Table 6). Elevation within the sediment-slurry treatment area was directly related to the nature of treatment-level and all levels were significantly different from each other except for the vegetated and medium elevation treatment-levels ($p = 0.8491$; Table 7, Fig. 7). Elevation of the pop-up treatment-level was the highest because sections of the original marsh (pop-ups) settled on top of the added sediment-slurry. The designation of high, medium, and low elevation treatment-levels were decided based on elevation measurements taken near the sampling transects (before transects were established) and, as expected, these three elevations were significantly different (Fig. 7). The vegetated treatment-level, which was identified and selected based on the presence of live vegetation in the fall of 2003 when most of the experimental area was devoid of vegetation, had an elevation higher than the reference marshes but similar to the medium treatment-level (Fig. 7). Compared to the experimental area, elevation

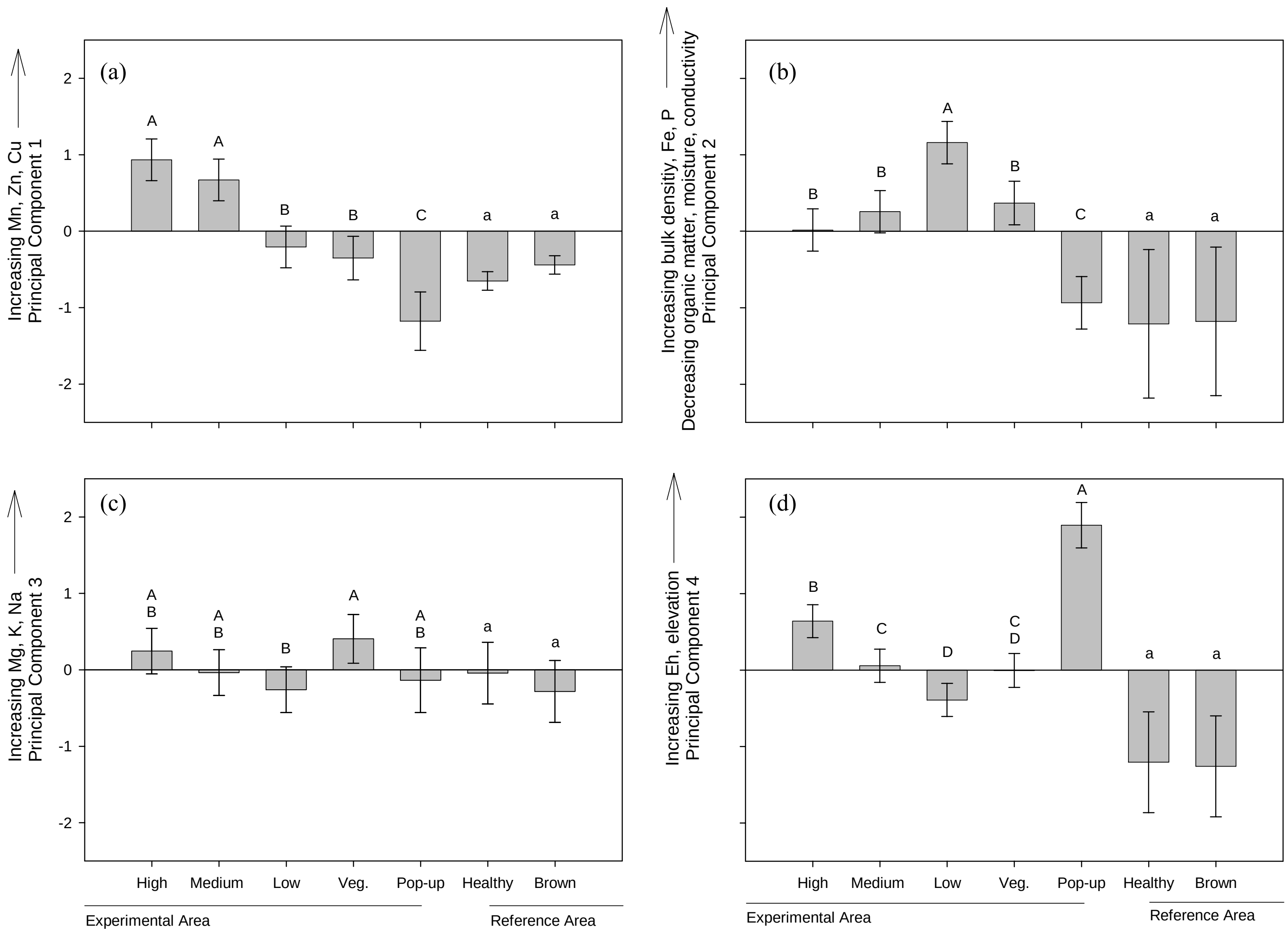


Figure 6. Principal component loadings across treatment-levels and reference marsh types averaged over time. The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas.

Table 4. Results of a principal components analysis that combined 16 soil variables into 4 principal components. Values shown are coefficients that describe how strongly a variable relates to the component. Coefficients with high absolute values are in bold because they define the principal component.

Variable	PC 1	PC 2	PC 3	PC 4
Moisture	-0.55	**-0.74**	-0.04	-0.27
Bulk Density	0.59	**0.74**	0.13	0.18
% Organic Matter	-0.57	**-0.74**	-0.1	-0.14
Electrical Conductivity	-0.33	**-0.72**	0.06	-0.39
pH	0.59	0.50	0.08	-0.30
P	0.47	**0.72**	0.18	0.10
Fe	-0.09	**0.84**	0.10	0.06
Cu	**0.92**	0.16	0.07	0.20
Mn	**0.92**	0.21	0.11	0.19
Zn	**0.92**	0.21	0.11	0.19
Ca	0.49	0.57	0.50	0.17
Mg	0.08	0.17	**0.95**	-0.02
K	0.18	0.55	**0.69**	-0.14
Na	0.04	-0.12	**0.92**	0.07
Eh	0.17	0.17	-0.03	**0.86**
Elevation	0.17	0.12	0.05	**0.88**
Variation Explained	52%	15%	10%	9%

Table 5. The effect of treatment level/reference marsh type, time (fall 2003, spring 2004, and fall 2004), and the interaction of treatment level/reference marsh type by time on principal component scores. Data in bold indicate significant differences.

	Experimental Area			Reference Area		
Principal Component	Treatment-Level	Time	Interaction	Marsh Type	Time	Interaction
(PC1) Trace metals	**< .0001**	0.9914	**0.0201**	0.0549	**0.0332**	**0.0421**
(PC2) Minerals	**< .0001**	0.8764	0.7463	0.9453	0.3627	0.4549
(PC3) Salts	**0.0286**	0.1543	**0.0214**	0.2961	0.4018	0.5405
(PC4) Inundation	**< .0001**	**0.0084**	0.7569	0.8665	0.0762	0.1446

was significantly lower in marshes that did not receive the sediment-slurry amendment (Fig. 7). The reference healthy marsh sites had significantly higher elevation than the reference brown marsh sites ($p < 0.0729$; Fig. 7).

Percent time flooded (Fig. 8) and water depth (Fig. 9, A2) were inversely related to elevation in both the experimental and reference areas. The reference marshes, which were at the lowest elevations, were flooded approximately 30-50 % of the time during a month of high water levels (October 2004) while all of the SSTLs were flooded less than half the time of the reference marshes (Fig. 8). On average, minimum water depths during a five-month period demonstrate a daily draining of all marshes in the experimental area and reference area (Fig. 9). Elevation controlled water depth in the experimental and reference areas. Treatment-levels with high elevations were infrequently flooded and as elevation decreased, water depth was more commonly above treatment-level/reference marsh type elevation (Fig. 9).

Time effects for Eh (redox potential) were highly significant in the experimental area ($p < 0.0001$), with non-significant treatment-level effects and interactions (Table 6). All three sampling periods were significantly different with the highest overall Eh (mean, 95 % confidence interval: 230, 200 to 261) in the spring of 2004 and the lowest overall Eh (135, 105 to 165) in the fall of 2004. There were no significant main effects or interactions with Eh in the reference area. During the fall of 2004, a period of relatively low redox potential, the experimental (135, 105 to 165) and reference (85, 45 to 139) area had similar redox potentials. In comparison, during a period of high redox potential (spring 2004), the experimental area (230, 200 to 261) had significantly higher redox potentials than the reference area (107, 61 to 166).

Table 6. The degrees of freedom (numerator, denominator) and the effect of treatment level (TL) / reference marsh type (MT), time (T) (fall 2003, spring 2004, and fall 2004), and the interaction of treatment level/reference marsh type and time on soil characteristics. Data for specific variables are not available (n/a) because they were not sampled in multiple time periods. Data is not available for sulfide in the experimental area because sulfide was either below detection limits or unable to be analyzed. Data for organic matter was obtained by direct measurement in the spring of 2004. A linear regression with bulk density was used to extrapolate values for organic matter during the fall of 2003 and the fall of 2004.

	Experimental Area						Reference Area					
	Degrees of Freedom			Significance			Degrees of Freedom			Significance		
Variable	TL	T	TL*T	TL	T	TL*T	MT	T	MT*T	MT	T	MT*T
Moisture	4,10	2,26	8,26	**<.0001**	**0.0063**	0.3049	1,2	2,4	2,4	0.7599	0.7257	0.1922
Bulk Density	4,10	2,26	8,26	**<.0001**	**0.0025**	0.6202	1,2	2,4	2,4	0.7881	0.3975	**0.0285**
Elevation	4,10	n/a	n/a	**<.0001**	n/a	n/a	1,2	n/a	n/a	0.0729	n/a	n/a
% Sand	4,10	n/a	n/a	**0.0057**	n/a	n/a	1,2	n/a	n/a	0.3898	n/a	n/a
% Silt	4,10	n/a	n/a	**0.0085**	n/a	n/a	1,2	n/a	n/a	0.6894	n/a	n/a
% Clay	4,10	n/a	n/a	**0.0337**	n/a	n/a	1,2	n/a	n/a	0.5032	n/a	n/a
% Organic Matter	4,10	2,26	8,26	**<.0001**	**0.0221**	0.1672	1,2	2,4	2,4	0.6863	0.9968	0.7028
Electrical Conductivity	4,10	2,26	8,26	**0.0361**	**<.0001**	**0.0004**	1,2	2,4	2,4	0.6768	**0.0066**	0.3749
Eh	4,10	2,26	8,26	0.0571	**<.0001**	0.2065	1,2	2,4	2,4	0.5370	0.2508	0.1580
Ca	4,10	2,26	8,26	**0.0014**	**<.0001**	0.1935	1,2	2,4	2,4	0.9187	0.3741	0.8335
Mg	4,10	2,26	8,26	**0.0269**	**0.0146**	**0.0414**	1,2	2,4	2,4	0.8020	0.2289	0.3330
K	4,10	2,26	8,26	**0.0006**	0.1256	0.3903	1,2	2,4	2,4	0.9931	**0.0430**	0.2596
Na	4,10	2,26	8,26	0.2031	**0.0006**	**0.0003**	1,2	2,4	2,4	0.3128	0.9408	0.4651
pH	4,10	2,26	8,26	**<.0001**	**0.0002**	**0.0216**	1,2	2,4	2,4	0.5917	**0.0490**	0.8812
Cu	4,10	2,26	8,26	**0.0012**	**0.0412**	**0.0002**	1,2	2,4	2,4	0.5869	**0.0144**	0.0957
Fe	4,10	2,26	8,26	**0.0049**	0.2317	0.2955	1,2	2,4	2,4	0.7963	**0.0106**	0.0974
Mn	4,10	2,26	8,26	**0.0005**	0.2804	**0.0002**	1,2	2,4	2,4	0.3692	**0.0261**	0.3181
Zn	4,10	2,26	8,26	**0.0005**	0.2830	**0.0002**	1,2	2,4	2,4	0.3691	**0.0262**	0.3175
P	4,10	2,26	8,26	**0.0001**	**0.0406**	0.4146	1,2	2,4	2,4	0.9061	**0.0435**	0.9449
NH_4-N	4,10	1,13	4,13	**0.0453**	0.2861	0.3761	1,2	1,2	1,2	0.5461	0.0587	0.9352
NH4-N + NO3¯-N	4,10	1,13	4,13	0.2375	**0.0267**	0.5695	1,2	1,2	1,2	0.5837	0.1110	0.7212
Sulfide	n/a	n/a	n/a	n/a	n/a	n/a	1,2	n/a	n/a	0.3469	n/a	n/a

Table 7. Substrate characteristics averaged over time (treatment-level and reference marsh type effects) in sediment-amended marshes and reference marshes (least-square means with 95 % confidence intervals). The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas. Sulfide values not available (n/a) were below the detection limits (0.003 mM) or unable to be analyzed.

Experimental Area	Elevation (cm)	Bulk Density (g cm^{-3})	% Moisture	% Organic Matter
High	29.03 B (26.49 to 31.59)	1.04 A (0.91 to 1.18)	31.41 C (27.56 to 35.79)	5.08 C (1.95 to 8.2)
Medium	21.45 C (18.91 to 24.01)	0.97 A (0.83 to 1.10)	33.86 C (29.72 to 38.57)	6.48 C (3.36 to 9.61)
Low	13.63 D (11.08 to 16.19)	0.98 A (0.84 to 1.11)	35.86 C (31.47 to 40.83)	6.34 C (3.23 to 9.47)
Vegetated	19.81 C (17.17 to 22.45)	0.75 B (0.61 to 0.89)	43.81 B (38.41 to 49.96)	11.94 B (8.75 to 15.12)
Pop-up	36.27 A (32.37 to 39.89)	0.34 C (0.17 to 0.51)	65.30 A (55.40 to 76.94)	24.65 A (20.69 to 28.60)
Reference Area				
Healthy	4.04 a (-0.79 to 8.88)	0.19 a (-0.05 to 0.44)	81.36 a (58.54 to 96.31)	28.76 a (12.74 to 44.79)
Brown	-1.52 a (-6.35 to 3.32)	0.20 a (-0.04 to 0.45)	79.03 a (55.66 to 95.13)	26.28 a (10.26 to 42.30)

Experimental Area	Sulfide (mM)	Eh (mV)	Electrical Conductivity (mS)	pH
High	n/a	211.64 A (154.61 to 268.67)	9.33 AB (7.17 to 12.07)	7.63 A (7.45 to 7.8)
Medium	n/a	169.35 A (112.32 to 226.38)	8.83 AB (6.77 to 11.44)	7.71 A (7.54 to 7.89)
Low	n/a	141.90 A (84.87 to 198.92)	7.05 B (5.67 to 9.19)	7.56 A (7.38 to 7.73)
Vegetated	n/a	121.77 A (64.21 to 179.32)	9.53 AB (7.30 to 12.36)	7.14 B (6.96 to 7.32)
Pop-up	n/a	258.45 A (177.80 to 339.09)	13.77 A (9.77 to 19.26)	5.91 C (5.67 to 6.15)
Reference Area				
Healthy	2.00 a (-0.24 to 10.90)	-11.57 a (-74.59 to 83.00)	27.08 a (11.06 to 64.38)	6.78 a (6.10 to 7.46)
Brown	0.77 a (-0.55 to 6.05)	8.49 a (-62.19 to 110.72)	23.55 a (9.54 to 56.17)	6.64 a (5.96 to 7.32)

Table 7. Continued.

Experimental Area	% Sand	% Silt	% Clay
High	8.80 A (7.07 to 10.90)	41.33 A (34.89 to 48.92)	48.61 AB (40.14 to 58.82)
Medium	8.44 A (6.77 to 10.46)	42.88 A (36.21 to 50.76)	47.79 AB (39.47 to 57.83)
Low	9.18 A (7.39 to 11.36)	43.39 A (36.64 to 51.36)	46.31 B (38.24 to 56.05)
Vegetated	9.35 A (7.52 to 11.56)	43.82 A (37.01 to 51.87)	46.11 B (38.07 to 55.81)
Pop-up	4.23 B (3.00 to 5.85)	31.29 B (25.66 to 38.12)	60.65 A (48.87to 75.22)
Reference Area			
Healthy	10.25 a (1.09 to 59.37)	38.65 a (0.17 to 105.76)	39.49 a (7.69 to 187.69)
Brown	5.16 a (0.14 to 32.08)	33.11 a (0.10 to 90.84)	59.98 a (12.08 to 283.18)

* Percent sand, silt, and clay were determined after organic matter was oxidized by combustion.

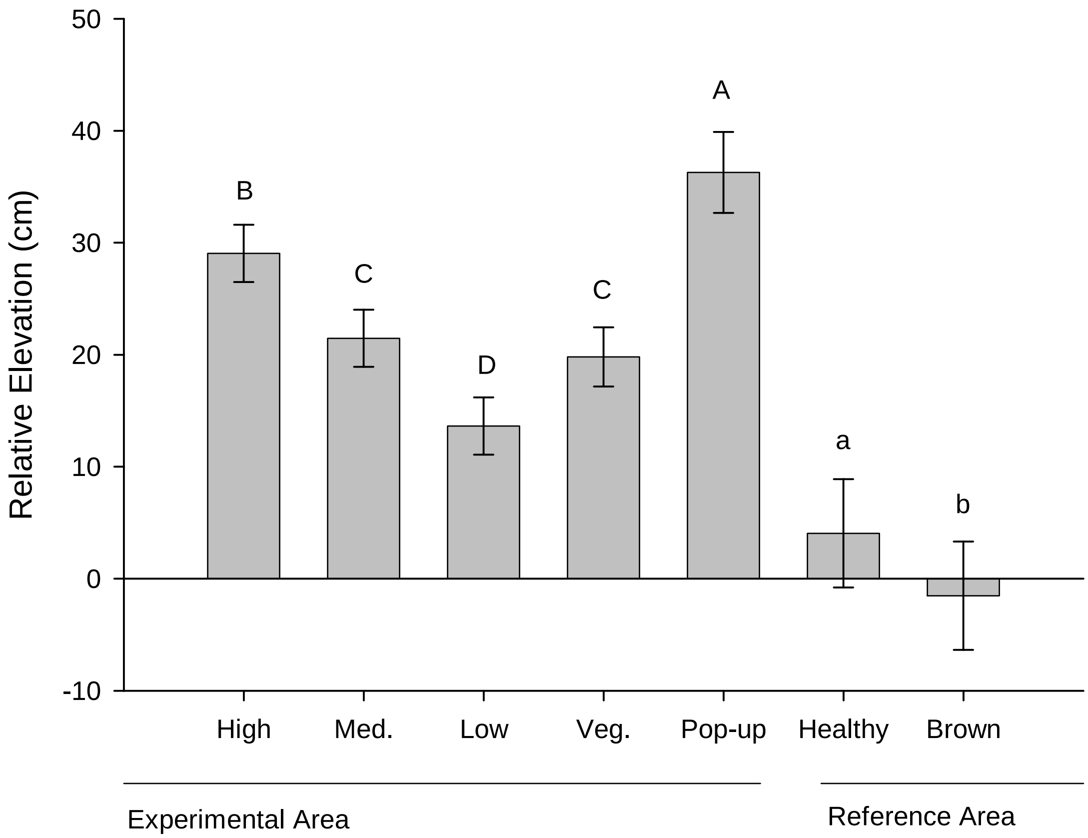


Figure 7. Elevation of all treatment-levels and reference marsh types referenced to ambient healthy marsh elevation (least-square mean with 95 % confidence interval). The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental and the reference areas. Reference healthy marshes and reference brown marshes are significantly different at $p < 0.0729$.

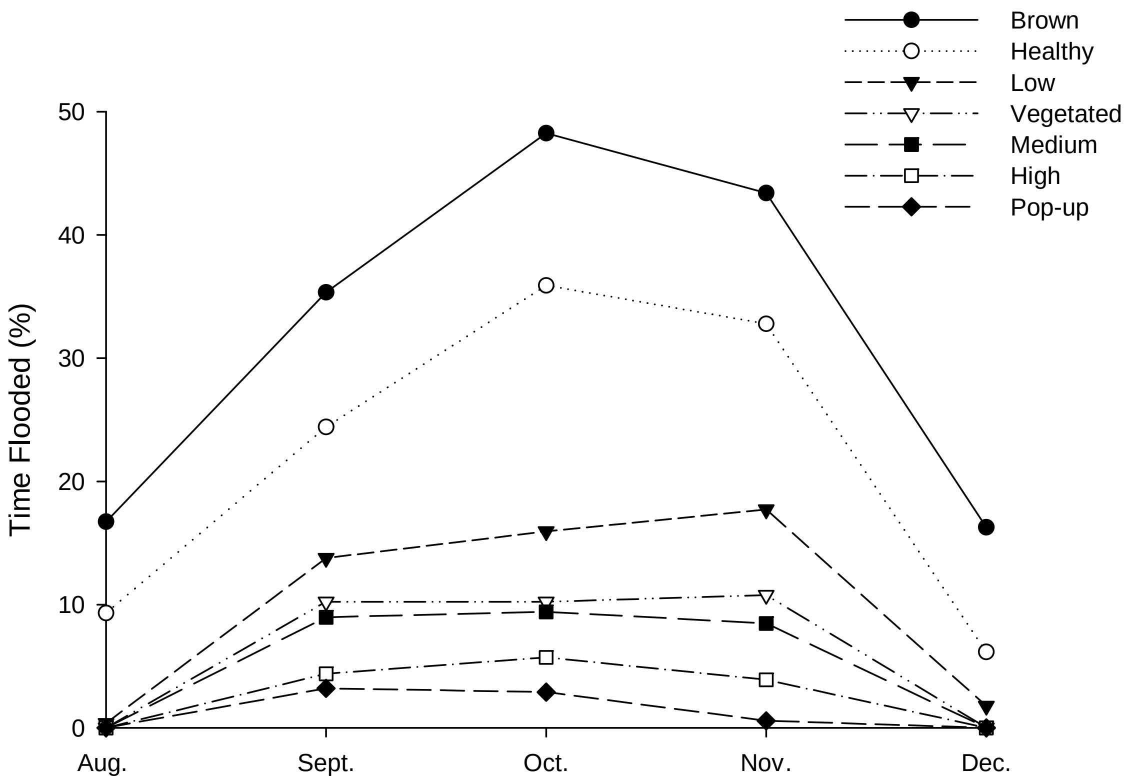


Figure 8. Percent time flooded from August thru December of 2004. Percent time flooded is based upon the water depth recorded from the water sonde and the average elevation of each treatment-level and reference marsh type.

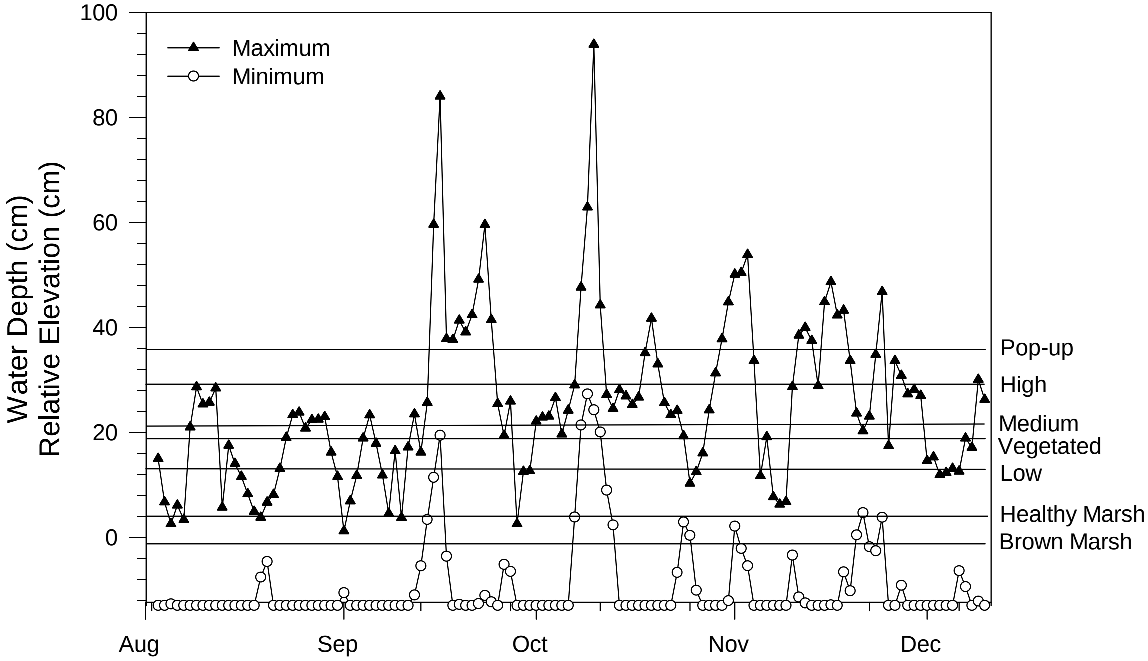


Figure 9. Maximum (▲) and minimum (o) daily water depth referenced to a common datum is presented with the relative land elevations of the experimental and reference areas from August thru December 2004. Horizontal lines indicate the relative elevation of each treatment-level and reference marsh type. Only the reference brown marshes were flooded persistently for daily maximum water depth. Negative values indicate the extent of marsh draining. The detection limit for the water sonde was -13 cm relative to the ambient marsh surface.

The high elevations, relatively low percent time flooded, and high redox potentials in the experimental area resulted in sulfide levels below detection limits (0.003 mM). Conversely, the low elevations and long hydroperiods (Figs. 8 and 9) in the reference marshes resulted in low redox potentials and sulfide concentrations (Table 7) above or near levels (1.0mM) known to cause reductions in growth of *Spartina alterniflora* (Koch et al. 1990).

Bulk density significantly varied among treatment-levels in the experimental area (Table 6) but did not significantly differ between the reference marsh types. High, medium, and low elevation treatment-levels had statistically similar bulk densities that were significantly higher than the vegetated and pop-up treatment-levels (Table 7). The

vegetated treatment-level had a higher bulk density than the pop-up treatment-level (Table 7). The reference marshes had low bulk densities that were equivalent to that of the pop-up treatment-level but significantly lower than all other SSTLs. There was also a significant time effect in the experimental area, but differences between time periods were minimal (fall 2003: 0.77, 0.67 to 0.87; spring 2004: 0.81, 0.71 to 0.92; fall 2004: 0.86, 0.76 to 0.96). Similarly, there was a significant interaction with time (Table 6) in the reference area. However, the Tukey-Kramer adjustment was not significant, indicating that the interaction was minor.

Treatment-level effects for organic matter were highly significant (Table 6). The pop-up treatment-level had a high organic matter content that was significantly higher than all other SSTLs (Table 7). The vegetated treatment-level had a moderate organic matter content that was significantly greater than the high, medium, and low treatment-levels, which were statistically similar. References marshes were equivalent to the pop-up and vegetated treatment-levels and had significantly more organic matter than the high, medium, and low elevation treatment-levels. Although time was significant in the experimental area ($p = 0.0221$), differences between time periods were minimal. As expected, similar, but inverse, trends were seen between bulk density and organic matter because they were negatively correlated ($r = -0.87$, $p < 0.0001$).

Treatment-level had a significant effect on the percent sand, silt, and clay while reference marsh type had no effect (Table 6). The pop-up treatment-level had significantly lower percentages of sand and silt than all other SSTLs (Table 7). There were no statistical differences in the percentages of sand, silt, and clay between the high, medium, low, and vegetated treatment-levels (Table 7). Although, the percent of clay within the pop-ups was substantially higher than all other SSTLs, it was only

significantly different from the vegetated and low treatment-levels. The small sample size and low replication in the reference area reduced the power of the model resulting in confidence intervals that do not accurately reflect the raw data (Table 7), i.e., confidence intervals were much larger than the variance in the raw data would suggest.

Inorganic nitrogen, one of the most important and often limiting nutrients in the salt marsh, did not vary among reference marsh types. Although there was a significant treatment-level effect for NH_4-N, the Tukey-Kramer adjustment showed no significant difference, indicating that the effect was minor (Table 8). NH_4-N + NO_3^--N did vary over time in the experimental area and was significantly higher in the spring of 2004 compared to the fall of 2004. Both NH_4-N + NO_3^--N and NH_4-N did not vary between the experimental and reference areas (Table 8).

Phosphorus varied significantly over treatment-level ($p < 0.0001$; Table 6). Phosphorus concentrations were statistically similar between high, medium, low, and vegetated treatment-levels (Table 8), which were greater than for the pop-up treatment-level. The pop-up treatment-level had phosphorus concentrations equivalent to that of the reference marshes. Both the pop-up treatment-level and the reference marshes had significantly lower phosphorus compared to all other treatment-levels (Table 8). Phosphorus also varied over time in both the experimental ($p < 0.0406$) and reference areas ($p < 0.0435$; Table 6). Phosphorus concentrations significantly decreased from fall of 2003 to the fall of 2004 (data not shown).

Table 8. Average exchangeable soil nutrient concentrations (treatment-level/reference marsh type effects; umol/cm^3) in marshes that received a sediment amendment and in reference marshes (least-square means with 95 % confidence intervals). The same letters indicate no significant differences between treatment means ($p < 0.05$) within either the experimental or reference areas. Non-overlapping confidence intervals identify significant differences between treatment-levels in experimental area and the reference area.

Experimental Area	Ca	Mg	K	Na
High	50.38 A (42.68 to 58.08)	45.00 AB (40.91 to 49.07)	18.92 A (16.89 to 20.94)	251.69 A (213.76 to 289.62)
Medium	41.57 A (33.87 to 49.27)	42.22 AB (38.14 to 46.30)	19.71 A (17.69 to 21.74)	227.85 A (189.92 to 265.77)
Low	40.40 A (32.70 to 48.10)	41.94 AB (37.86 to 46.02)	21.21 A (19.18 to 23.23)	189.74 A (151.81 to 227.67)
Vegetated	37.62 A (29.76 to 45.48)	49.49 A (45.14 to 58.84)	21.34 A (19.19 to 23.49)	233.43 A (194.54 to 272.33)
Pop-up	17.33 B (6.93 to 27.73)	37.06 B (31.28 to 42.83)	10.86 B (8.00 to 13.73)	223.87 A (170.28 to 277.45)
Reference Area				
Healthy	12.58 a (6.41 to 23.88)	37.39 a (26.77 to 52.07)	12.56 a (5.10 to 29.14)	215.33 a (185.09 to 250.49)
Brown	12.27 a (6.24 to 23.32)	36.24 a (25.94 to 50.48)	12.59 a (5.12 to 29.21)	201.47 a (173.16 to 234.38)

Experimental Area	Fe	Mn	Cu	Zn
High	2.50 B (1.84 to 3.23)	0.06 A (0.05 to 0.06)	0.04 A (0.03 to 0.04)	0.05 A (0.04 to 0.05)
Medium	2.88 B (2.18 to 3.65)	0.05 AB (0.04 to 0.06)	0.03 AB (0.03 to 0.04)	0.04 AB (0.04 to 0.05)
Low	4.81 A (3.94 to 5.74)	0.04 BC (0.03 to 0.04)	0.02 BC (0.01 to 0.03)	0.03 BC (0.02 to 0.04)
Vegetated	3.23 AB (2.48 to 4.05)	0.03 C (0.02 to 0.04)	0.02 BC (0.02 to 0.03)	0.03 C (0.02 to 0.03)
Pop-up	2.13 B (1.26 to 3.13)	0.02 C (0.01 to 0.03)	0.01 C (0.00 to 0.02)	0.02 C (0.01 to 0.02)
Reference Area				
Healthy	1.00 a (-0.15 to 3.37)	0.01 a (-0.00 to 0.03)	0.01 a (-0.00 to 0.02)	0.01 a (-0.00 to 0.02)
Brown	0.84 a (-0.21 to 3.30)	0.02 a (0.00 to 0.03)	0.01 a (-0.00 to 0.02)	0.01 a (-0.00 to 0.03)

Table 8. Continued.

Experimental Area	P	NH_4-N	NH_4-N + NO_3^--N
High	5.47 A (4.75 to 6.20)	0.05 A (-0.00 to 0.10)	0.11 A (0.04 to 0.18)
Medium	5.15 A (4.42 to 5.88)	0.11 A (0.06 to 0.16)	0.14 A (0.07 to 0.22)
Low	5.48 A (4.76 to 6.20)	0.14 A (0.09 to 0.19)	0.16 A (0.09 to 0.24)
Vegetated	4.55 A (3.80 to 5.30)	0.04 A (-0.02 to 0.09)	0.06 A (-0.02 to 0.13)
Pop-up	1.38 B (0.37 to 2.38)	0.06 A (-0.02 to 0.13)	0.07 A (-0.03 to 0.18)
Reference Area			
Healthy	0.60 a (-0.30 to 2.64)	0.19 a (-0.06 to 0.45)	0.19 a (-0.05 to 0.43)
Brown	0.54 a (-0.32 to 2.51)	0.13 a (-0.12 to 0.39)	0.14 a (-0.10 to 0.38)

DISCUSSION

This research demonstrated that the addition of sediment-slurries could increase the rate of recovery following disturbance in a rapidly subsiding salt marsh. Elevations averaging 14 and 20 cm above ambient marsh located in the low and vegetated treatment-levels, respectively, had rapid plant recruitment and species richness similar to that of the healthy reference marsh sites. The medium elevation treatment-level, averaging 21 cm in elevation, had marginal recovery (~1/3 the plant cover of the healthy reference marshes). Pop-ups, which were highly organic and had elevations averaging 36 cm, were characterized by rapid vegetation establishment and high species diversity. Recovery was the lowest in the high elevation treatment-level and in the reference brown marsh sites, which both showed little recovery by fall 2004.

One of the primary factors influencing marsh plant distribution (Adams 1963, DeLaune et al. 1983b, Bertness 1991b, Edwards and Proffitt 2003) and function (DeLaune et al. 1983b, Reed and Cahoon 1992, Proffitt et al. 2003) is elevation. In Louisiana salt marshes, *Spartina alterniflora* is the dominant low marsh plant, in part, because of high flood- and sulfide tolerances. As waterlogging stresses decrease via increasing elevation, competitive interactions cause the replacement of *S. alterniflora* with other species such as *S. patens* and *Iva frutescens* (Bertness 1991a, 1991b, Mitsch and Gosselink 2000). Although elevations optimal for rapid plant colonization in salt marshes have not been conclusively identified, Cornu and Sadro (2002) showed colonization of restored wetlands to be directly related to the degree of the sediment amendment. Additionally, DeLaune et al. (1990) found significant increases in aboveground biomass with 8–10 cm of sediment additions and Mendelssohn and Kuhn (2003) concluded that sediment additions of 15 cm significantly increased plant cover

and additions of 15-30 cm significantly increased plant biomass. Raising marsh surface elevation in dieback areas of a Louisiana salt marsh by 30 cm, Wilsey et al. (1992) demonstrated that there were significant increases in number of culms, belowground biomass, and total biomass of *S. alterniflora*.

Frequency and duration of inundation (Sasser 1977, Day et al. 1993, Brinson et al. 1995, Hacker and Bertness 1999) and redox potential (DeLaune et al. 1983b, Wilsey et al. 1992, Hacker and Bertness 1999, Mendelssohn and Kuhn 2003) have been found to correlate with elevation in tidal salt marshes. Similarly, I found Eh and elevation to be highly related (Table 4) and I identified an inverse relationship between percent time flooded and elevation at the study site (Fig. 9). The relatively low redox potentials seen in the reference marshes can be attributed to their low elevation and resultant increases in flooding (DeLaune et al. 1983a, Mendelssohn and McKee 1988b, Wilsey et al. 1992, Cornu and Sadro 2002, Mendelssohn and Kuhn 2003). The reference area was low in elevation and frequently flooded while the experimental area had higher elevations, which resulted in less flooding. When soils become flooded, oxygen depletion is rapid due to its slow rate of diffusion (Gambrell and Patrick 1978) and consumption by roots and facultative and anaerobic microorganisms during respiration (Mitsch and Gosselink 2000). These microorganisms use oxidized compounds as terminal electron acceptors converting them into their reduced states, potentially forming toxins, such as hydrogen sulfide, which are harmful to plants.

High sulfide levels, similar to those found in other Louisiana marshes (DeLaune et al. 1983b), have been found to decrease productivity (Koch and Mendelssohn 1989) and may therefore, be detrimental to the growth of *S. alterniflora* (Mendelssohn and

Morris 2000). Soil sulfide concentrations have been shown to sharply decrease with the addition of sediment (Mendelssohn and Kuhn 1999, Slocum et al. 2005). At the study site, soils that received the sediment amendment did not become reduced enough to have measurable concentrations of interstitial sulfide. In contrast, both reference marsh types had interstitial sulfide concentrations high enough to limit the uptake of nitrogen and to cause reductions in plant growth (Bradley and Dunn 1989, Bradley and Morris 1990, Koch et al. 1990). The high sulfide concentrations may have limited seedling establishment and restricted vegetative recruitment from rhizome expansion of plants surrounding the reference brown marsh sites. This would explain why the few seedlings noted in the reference brown marshes in the spring of 2004 were not present in the fall of 2004 (personal observation A. Schrift). Sulfide was not a factor limiting plant establishment in the experimental area. However, sulfide along with flooding duration may have contributed to the slow rate of recovery in the reference brown marsh sites.

Elevation and the resultant time flooded can also affect nutrient availability. Nitrogen is generally the most limiting nutrient in salt marshes (Valiela et al. 1975), in part, because the constant fluctuation between oxidized and reduced conditions promotes denitrification, rapidly reducing nitrogen concentrations. Ammonium is dominant inorganic nitrogen form available for *Spartina alterniflora*, the dominant plant in the study area, and is therefore, an important factor governing marsh vigor. However, similar concentrations of inorganic nitrogen throughout marshes with various elevations and different recovery rates imply that nitrogen was not a factor controlling recovery in the experimental area. High sulfide concentrations (Bradley and Morris 1990, Koch et al. 1990) and/or low root oxygen concentrations (Morris and Dacey 1984) in the reference marshes may have inhibited nitrogen uptake (Bradley and Morris 1990) and contributed

to the high average NH_4-N concentrations seen in the reference marshes during the fall of 2004 (data not shown).

Marshes that are seldom flushed and subjected to radiant heat and resultant water evaporation can generate high substrate salt concentrations, which have been shown to limit plant colonization in bare patches (Bertness et al. 1992) and to limit growth rates of *Spartina alterniflora* (Smart 1980). The relationship between salinity and elevation may change depending upon marsh geography. Some researchers (Adams 1963, DeLeeuw et al. 1991, Mendelssohn and Kuhn 2003) found differences in salt concentration to be based upon elevation while others (Silvestri et al. 2005) found no differences between high and low elevations. Electrical conductivity, a measure of salinity, and the concentration of salts such as Mg, Ca, Na, and K throughout the study area were within normal ranges and in many instances were not significantly different among or between treatment-levels and reference marsh types (Fig. 6, Table 8). These results imply that salt concentrations were not a factor limiting plant colonization at the study site.

High bulk density, a measure of mineral content, has been related to faster plant recovery (Mendelssohn and Kuhn 2003, Slocum et al. 2005) and productivity (DeLaune et al. 1979). Mineral matter can improve marsh vigor by increasing nutrient availability (Mitsch and Gosselink 2000) and by decreasing sulfide toxicity via precipitation with metals such as Fe and Mn (Gambrell and Patrick 1978, Gambrell 1994, Mendelssohn and Morris 2000). At the study site, bulk density was significantly higher in sediment-subsidized areas. The high and medium treatment-levels had marked increases of Cu, Mn, Zn, Ca, and P, the low and vegetated treatment-levels had increases of Ca and P, and the low treatment-level had more Fe compared to reference marshes (Table 8). Soils with extremely low bulk densities, like those seen in the reference marshes, have lower plant

production than soils with high bulk densities (DeLaune et al. 1990). However, differences in plant recovery were not consistently or statistically associated with higher bulk density.

Marsh function is sometimes affected if sediment additions alter soil texture since fine-textured sediments more readily retain nutrients than coarse-textured sediments (Broome et al. 1988). Boyer and Zedler (1998) found that a marsh amended with dredged materials resulting in coarser-textured sediments was unable to retain nutrients and production equal to that of nearby unamended marshes. Conversely, Mendelssohn and Kuhn (2003) found a marsh amended with dredged material contained more sand but still had high plant cover and total biomass even with reductions in soil ammonium compared to reference marshes. The results indicate that soil texture did not play a significant role in plant recovery. There were no differences in soil texture between reference and sediment amended marshes and differences in soil texture between treatment-levels were not associated with differences in recovery.

I found other factors, which were not wholly dependent on elevation, to be important determinants for successful recovery at the study site. The high rate of recovery in the pop-up treatment-level may be attributed to its highly organic substrate, which can readily retain moisture (Neill and Turner 1987). The pop-up treatment-level had moist soils while nearby areas in the high and medium treatment-levels were often dry enough to have cracked soils. This retention of moisture may have promoted plant recruitment, which led to the high rates of vegetation recruitment seen in the pop-up treatment-level. Although vegetation establishment on the pop-ups was rapid and total plant cover equaled that of the healthy reference marshes, the vegetation community of the pop-ups, due to their high elevation, was more diverse than the healthy reference

marshes. It is likely that species other than *S. alterniflora* will become dominant over time.

Viable rhizomes in the sediment may have promoted the high rate of recovery seen in the vegetated treatment-level. Immediately following slurry deposition, standing dead *S. alterniflora* was identified in areas that were later classified as the vegetated treatment-level (observations by M. D. Materne and I. A. Mendelssohn). Rhizomes may have been able to survive sediment burial. It has been reported that rhizomes can survive sediment additions up to 15 cm thick (Ford 1999). In the vegetated treatment-level, the sediment-slurry layer varied in thickness (6 - 41 cm). It is also possible that maximum burial depth for viable rhizomes reported by Ford (1999) is underestimated for areas receiving sediment-slurries due to the high water content and low density associated with non-dewatered sediment-slurries compared to other types of sediment addition. I conclude that elevation and resulting inundation regime, in combination with other factors like organic matter, moisture retention, and possibly rhizome viability, played an important role in rapid plant recruitment.

CONCLUSIONS

Salt marsh deterioration in Louisiana is an ongoing problem as a result of both human modification of the landscape and natural processes associated with the delta cycle. Chronic natural and anthropogenic disturbances of this once pristine ecosystem have reduced the resilience of many salt marshes in the Mississippi River Deltaic Plain (MRDP) making them susceptible to natural perturbations. Before the brown marsh phenomenon of 2000, droughts had been recorded in the MRDP (Swenson et al. 2003) but they were not associated with a marsh dieback event. The brown marsh phenomenon of 2000 may be the first of many weather-induced, large-scale, and severe perturbations to affect the region, signaling the need to develop restoration techniques suitable to such events.

Sediment-slurry additions were identified as a possible restoration method because vegetation health in salt-marshes of southern Louisiana was found to depend upon the degree of plant inundation and soil reduction resulting from changes in elevation (Mendelssohn and McKee 1988a, Mendelssohn and Morris 2000). Other researchers have shown that sediment amendments can effectively rehabilitate degraded marshes (Cahoon and Cowan 1988, DeLaune et al. 1990, Wilsey et al. 1992, Ford 1999, Mendelssohn and Kuhn 1999, Shafer 2002, Slocum et al. 2005). Mounting concern regarding application distance, resultant elevation, and cost prompted researchers to investigate new sediment application techniques. The hydraulic application of sediment-slurries allows dredged material to travel an extended distance from the discharge pipe. Slocum et al. (2005) documented a 1000 m transport. The high water content and intense mixing reduces separation of particle sizes. A relatively even distribution of particle sizes reduces the number of times the dredge operator has to move the discharge pipe,

decreasing costs. Because sediment-slurries can be pumped kilometers from the dredging location, Louisiana has readily available materials from the Mississippi River, Gulf of Mexico, and other frequently dredged navigable waterways for many otherwise unreachable interior marshes.

The efficient use of available resources and marsh recovery rate are essential factors to be considered when designing marsh restoration projects. Marsh restoration projects should seek to create conditions conducive to high recovery rates since plants help sustain salt marsh integrity by maintaining substrate cohesion, which decreases erosion, and by producing organic matter, which can control vertical accretion rates (Hatton et al. 1983, DeLaune et al. 1991, Nyman et al. 1993). Additionally, the availability of dredged material is limited, and therefore, it is important to determine the appropriate amount of dredged material to deposit in order to most effectively use available resources.

I have shown that resource managers should evaluate elevation, bulk density, organic matter content, and rhizome viability when designing restoration projects involving sediment-slurries in Louisiana's coastal salt marshes. Pop-ups, which are unintended formations created by the slurry application, have a positive effect on plant establishment and should be further studied in order to assess their role in succession of restored areas. Viable rhizomes and highly organic substrates can significantly raise the elevation at which a high rate of recovery is possible. If viable rhizomes are not present and organic matter content is low, frequency and duration of inundation should be similar to that of the low elevation.

LITERATURE CITED

Adams, D. A. 1963. Factors Influencing Vascular Plant Zonation in North Carolina Salt Marshes. Ecology **44**:445-456.

Allison, S. K. 1996. Recruitment and Establishment of Salt Marsh Plants Following Disturbance by Flooding. The American Midland Naturalist **136**:232-248.

Barras, J., S. Beville, D. Britsch, S. Hartley, S. Hawes, J. Johnson, P. Kemp, Q. Kinler, A. Martucci, J. Porthouse, D. Reed, K. Roy, S. Sapkota, and J. Suhayda. 2004. Historical and projected coastal Louisiana land changes: 1978-2050. USGS Open File Report 03-334.

Base SAS 9.1 Procedures Guide.2004. *in*. SAS Institute Inc., Cary, NC.

Bertness, M., L. Gough, and S. W. Shumway. 1992. Salt tolerances and the distribution of fugitive salt marsh plants. Ecology **73**:1842-1851.

Bertness, M. D. 1991a. Interspecific Interactions Among High Marsh Perennials in a New England Salt Marsh. Ecology **72**:125-137.

Bertness, M. D. 1991b. Zonation of *Spartina patens* and *Spartina alterniflora* in a New England salt marsh. Ecology **72**:138-148.

Bradley, P. M., and E. L. Dunn. 1989. Effects of Sulfide on the Growth of Three Salt Marsh Halophytes of the Southeastern United States. American Journal of Botany **76**:1707-1713.

Bradley, P. M., and J. T. Morris. 1990. Influence of oxygen and sulfide concentration on nitrogen uptake kinetics in *Spartina alterniflora*. Ecology **71**:282-287.

Bremner, J. M., and D. R. Kenney. 1966. Determination and Isotope-Ratio Analysis of Different Forms of Nitrogen in Soils: 3. Exchangeable Ammonium, Nitrate, and Nitrite by Extraction-Distillation Methods. Soil Science Society of America Proceedings **30**:577-582.

Brinson, M. M., and R. R. Christian. 1999. Stability of Juncus roemerianus patches in a salt marsh. Wetlands **19**:65-70.

Brinson, M. M., R. R. Christian, and L. K. Blum. 1995. Multiple States in the Sea-Level Induced Transition From Terrestrial Forest to Estuary. Estuaries **11**:648-659.

Broome, S. W., E. D. Seneca, and J. W.W. Woodhouse. 1988. Tidal Salt Marsh Restoration. Aquatic Botany **32**:1-22.

Cahoon, D. R. J., and J. H. Cowan. 1988. Environmental Impacts and Regulatory Policy Implications of Spray Disposal of Dredged Material in Louisiana Wetlands. Coastal Management **16**:341-362.

Callaway, J. C. 2005. The challenge of restoring functioning salt marsh ecosystems. Journal of Coastal Research **SI 40**:24-36.

Cheramie, K., R. Francis, and K. Kilgen. 1995. Thin Layer Spoil Deposition: An Alternative Restorative Tool for Coastal Marshes. *in* Coastal Zone: Proceedings of the Symposium on Coastal Ocean Management. ASCE, Tampa, FL.

Closure Report: Initial Funding Allocation, DNR Dedicated Dredging Program (LA-1). 2000. Louisiana Department of Natural Resources, Coastal Restoration Division, Engineering Section.

Cornu, C. E., and S. Sadro. 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's South Slough. Restoration Ecology **10**:474-486.

Day, J. W., W. H. Conner, R. Costanza, G. P. Kemp, and I. A. Mendelssohn. 1993. Impacts of sea level rise on coastal systems with special emphasis on the Mississippi River deltaic plain. Pages 276-296 *in* R. A. Warrick, E. M. Barrow, and T. M. L. Wigley, editors. Climate and Sea Level Change: Observations, Projections, and Implications. Cambridge University Press, Cambridge, Great Britain.

Del Moral, R., and D. M. Wood. 1993. Early Primary Succession on a Barren Volcanic Plain at Mount St. Helens, Washington. American Journal of Botany **80**:981-991.

DeLaune, R. D., R. H. Baumann, and J. G. Gosselink. 1983a. Relationships among vertical accretion, costal submergence and erosion in a Louisiana Gulf coast marsh. Journal of Sedimentary Petrology **53**:147-157.

DeLaune, R. D., R. J. Buresh, and J. W.H. Paterick. 1979. Relationship of Soil Properties to Standing Crop Biomass of *Spartina alterniflora* in a Louisiana Marsh. Estuarine and Coastal Marine Science **8**:477-487.

DeLaune, R. D., J. A. Nyman, and J. W. H. Patrick. 1991. Sedimentation Patterns in Rapidly Deteriorating Mississippi River Deltaic Plain Coastal Marshes: Requirement and Response to Sediment Additions. Resource Development of the Lower Mississippi River: American Water Resources Association:59-68.

DeLaune, R. D., J. A. Nyman, and J. W. H. Patrick. 1994. Peat Collapse, Ponding and Wetland Loss in a Rapidly Submerging Coastal Marsh. Journal of Coastal Research **10**:1021-1030.

DeLaune, R. D., S. R. Pezeshki, J. H. Pardue, J. H. Whitcomb, and W. H. P. Jr. 1990. Some Influences of Sediment Addition to a Deteriorating Salt Marsh in the Mississippi River Deltaic Plain: A Pilot Study. Journal of Coastal Research **6**:181-188.

DeLaune, R. D., C. J. Smith, and J. W.H. Patrick. 1983b. Relationship of marsh elevation, redox potential, and sulfide to *Spartina alterniflora* productivity. Journal of the Soil Science Society of America **47**:930-935.

DeLeeuw, J., A. v. Dool, W. DeMunck, J. Nieuwenhuize, and W. G. Beeftink. 1991. Facors Influencing the Soil Salinity Regime along an Intertidal Gradient. Estuarine, Coastal and Shelf Science **32**:87-97.

Edwards, K. R., and C. E. Proffitt. 2003. Comparison of Wetland Structural Characteristics Between Created and Natural Salt Marshes in Southwest Louisiana, USA. Wetlands **23**:344-356.

Ford, M. A., D.R. Calhoon, and J.C. Lynch. 1999. Restoring Marsh Elevation in a Rapidly Subsiding Salt Marsh by Thin-layer Deposition of Dredged Material. Ecological Engineering **12**:189-205.

Foster, D. R., D. H. Knight, and J. F. Franklin. 1998. Landscape Patterns and Legacies Resulting from Large, Infrequent Forest Disturbances. **1**:497-510.

Gambrell, R. P. 1994. Trace and Toxic Metals in Wetlands--A Review. Journal of Environmental Quality **23**:883-891.

Gambrell, R. P., and W. H. Patrick. 1978. Chemical and Microbiological Properties of Anaerobic soils and Sediments. Pages 375-423 *in* D. D. Hook and R. M. Crawford, editors. Plant life in anaerobic environments. Ann Arbor Science Publishers, Ann Arbor, Michigan.

Gutzerova, N., and T. Herben. 2001. Patch dynamics and local succession a sandstone area with frequent disturbance. Journal of Vegetation Science **12**:533-544.

Hacker, S. D., and M. D. Bertness. 1999. Experimental Evidence for Factors Maintaining Plant Species Diversity in a New England Salt Marsh. Ecology **80**:2064-2073.

Hatton, R. S., R. D. DeLaune, and W. H. Patrick. 1983. Sedimentation, accretion, and subsidence in marshes of Barataria Basin, Louisiana. Liminology and Oceanography **28**:494-502.

Koch, M. S., and I. A. Mendelssohn. 1989. Sulfide as a Soil Phytotoxin: Differential Responses in Two Marsh Species. Journal of Ecology **77**:565-578.

Koch, M. S., I. A. Mendelssohn, and K. L. McKee. 1990. Mechanism for the hydrogen sulfide-induced growth limitation in wetland macrophytes. Limnology and Oceanography **35**:399-408.

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 **42**:421-428.

Matthews, S. D. 1984. Soil Survey of Lafourche Parish, Louisiana. United States Department of Agriculture, Soil Conservation Service.

McKee, K. L., I. A. Mendelssohn, and M. D. Materne. 2004. Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? Global Ecology and Biogeography **13**:65-73.

Mendelssohn, I. A., and N. L. Kuhn. 1999. The effects of sediment addition on salt marsh vegetation and soil physico-chemistry. Pages 55-61 *in* L. P. Rozas, J.A. Nyman, C.E. Proffitt, N.N. Rabalais, D.J. Reed, and R.E. Turner, editor. Recent Research in Coastal Louisiana: Natural System Function and Response to Human Influence. Louisiana Sea Grant College Program.

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. McKee. 1988a. The Use of Basic Research in Wetland Mangement Decisions. Pages 354-364 *in* W. G. Duffy and D. Clark, editors. Marsh Management in Coastal Louisiana: Effects and Issues--Proceedings of a Symposium. U.S. Fish and Wildlife Service and LA Dept. of Natural Resources, Baton Rouge, LA.

Mendelssohn, I. A., and K. L. McKee. 1988b. Spartina alterniflora Die-back in Lousiana: Time-Course Investigation of Soil Waterlogging Effects. Journal of Ecology **76**:509-521.

Mendelssohn, I. A., and J. T. Morris. 2000. Eco-physiological constraints on the primary productivity of *Spartina alterniflora*. *in* Concepts and Controversies. Elsevier Press.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands, Third Edition edition. John Wiley & Sons, Inc., Columbus, Ohio.

Morris, J. T., and J. W. H. Dacey. 1984. Effects of O_2 On Ammonium Uptake and Root Respiration by *Spartina alterniflora*. American Journal of Botany **71**:979-985.

Neill, C., and R. E. Turner. 1987. Backfilling Canals to Mitigate Wetland Dredging in Louisiana Coastal Marshes. Environmental Management **11**:823-836.

Nelson, D. W., and L. E. Sommers. 1996. Total Carbon, Organic Carbon, and Organic Matter. Pages 961-1010 *in* J. M. Bartels and J. M. Bigham, editors. Methods of Soil Analysis. Part 3. Chemical Methods. Soil Science Society of America and American Society of Agronomy, Madison, WI 53711, USA.

Nieuwstadt, M. G. L. V., and D. Sheil. 2005. Drought, fire and tree survival in a Borneo rain forest, East Kalimantan, Indonesia. Journal of Ecology **93**:191-201.

Nyman, J. A., R. D. DeLaune, H. H. Roberts, and W. H. P. Jr. 1993. Relationship between vegetation and soil formation in a rapidly submerging coastal marsh. Marine Ecology Progress Series **96**:269-279.

Penland, S., and K. E. Ramsey. 1990. Relative Sea-Level Rise In Louisiana and the Gulf of Mexico: 1908-1988. Journal of Coastal Research **6**:323-342.

Penland, S., H. H. Roberts, S. J. Williams, J. A.H. Sallenger, D. R. Cahoon, D. W. Davis, and C. G. Groat. 1990. Coastal Land Loss in Louisiana. Gulf Coast Association of Geological Societies **90**:685-699.

Pennings, S. C., and C. L. Richards. 1998. Effects of wrack burial in salt-stressed habitats: *Batis maritima* in a southwest Atlantic salt marsh. Ecography **21**:630-638.

Platt, W. J., and J. H. Connell. 2003. Natural disturbances and directional replacement of species. Ecological Monographs **73**:507-522.

Proffitt, C. E., S. E. Travis, and K. R. Edwards. 2003. Genotype and Elevation Influence *Spartina alterniflora* Colonization and Growth in a Created Salt Marsh. Ecological Applications **13**:180-192.

Reed, D. J., and D. R. Cahoon. 1992. The Relationship Between Marsh Surface Topography, Hydroperiod, and Growth of *Spartina alterniflora* in a Deteriorating Louisiana Salt Marsh. Journal of Coastal Research **8**:77-87.

Sasser, C. E. 1977. Distribution of Vegetation in Louisiana Coastal Marshes as Response to Tidal Flooding. Louisiana State University, Baton Rouge.

Shafer, D. 2002. Chapter 9: Thin Layer Placement. Pages 115-122 *in* R. E. Turner and B. Streever, editors. Approaches to Coastal Wetland Restoration. SPB Academic Publishing bv, The Hague, The Netherlands.

Shumway, S. W., and M.D. Bertness. 1994. Patch Size Effects on Marsh Plant Secondary Succession Mechanisms. Ecology **75**:564-568.

Silvestri, S., A. Defina, and M. Marani. 2005. Tidal regime, salinity and salt marsh plant zonation. Estuarine, Coastal and Shelf Science **62**:119-130.

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.

Smart, M. R., and J.W. Barko. 1980. Nitrogen Nutrition and Salinity Tolerance of *Distichlis spicata* and *Spartina alterniflora*. Ecology **61**:630-638.

Soil Survey Laboratory Investigations Manual.2004. Pages 34-55 *in* R. Burt, editor. NRCS.

Swenson, E. M., E. E. Evers, and J. M. Grymes. 2003. Task II.5--Integrative Approach to Understanding the Causes of Salt Marsh Dieback: Analysis of Climatic Drivers. LSU, Baton Rouge.

Thomas, G. W. 1982. Exchangeable Cations. Pages 159-165 *in* A. L. Page, R. H. Miller, and D. R. Keeney, editors. Methods of Soil Analysis: Part 2--Chemical and Microbiological Properties, second ed. American Society of Agronomy, Inc. and Soil Science Society of America, Inc., Madison, WI.

Turner, M. G., S. L. Collins, A. L. Lugo, J. J. Magnuson, T. S. Rupp, and F. J. Swanson. 2003. Disturbance dynamics and ecological response: The contribution of long-term ecological research. BioScience **53**:46-56.

Turner, M. J., and V. H. Dale. 1998. Large, Infrequent Disturbances: Comparing Large, Infrequent Disturbances: What Have We Learned? Ecosystems **1**:493-496.

Valiela, I., J. M. Teal, and W. J. Sass. 1975. Production and Dynamics of Salt Marsh Vegetation and the Effects of Experimental Treatment with Sewage Sludge. Journal of Applied Ecology **12**:973-981.

Vankvik, V. 2004. Gap dynamics in perennial subalpine grasslands: trends and processes change during secondary succession. Journal of Ecology **92**:86-96.

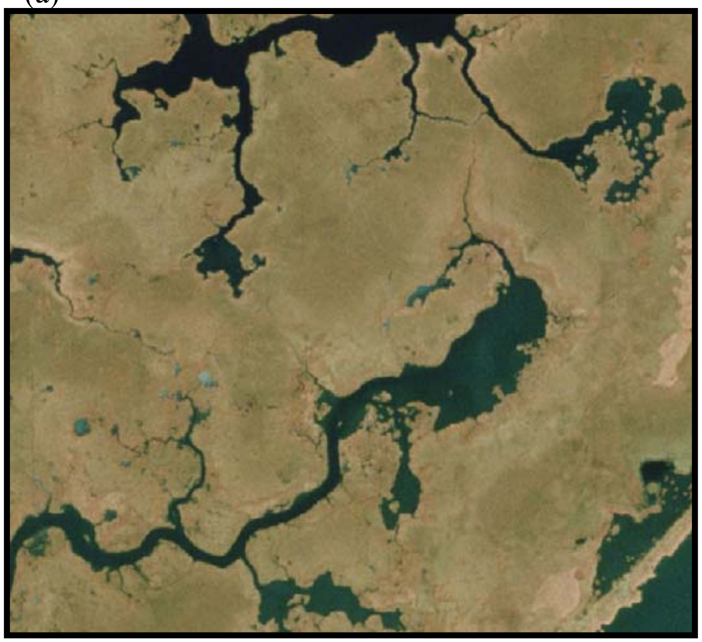
Walker, L. R., and R. Del Moral. 2003. Primary Succession and Ecosystem Rehabilitation. Cambridge University Press, Cambridge, UK.

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.

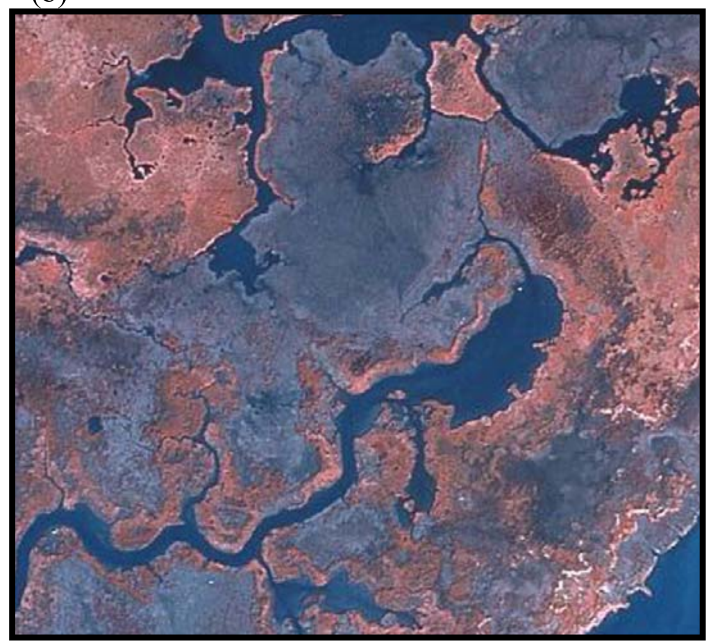
Wu, J., and S. A. Levin. 1994. A Spatial Patch Dynamic Modeling Approach to Pattern and Process in an Annual Grassland. Ecological Monographs **64**:447-464.

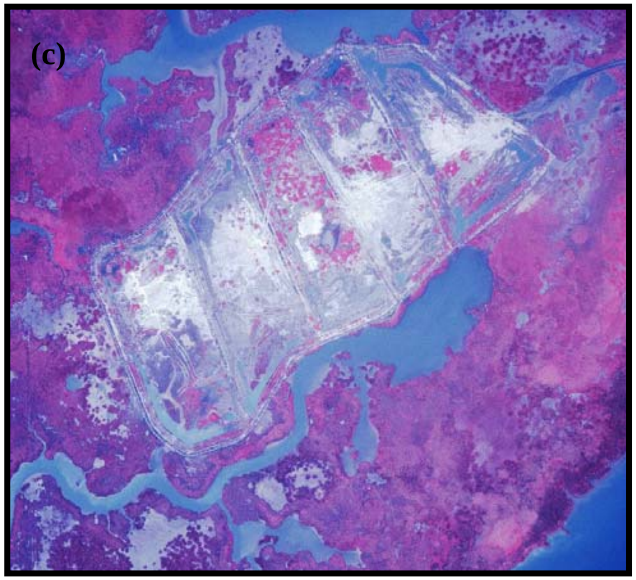
APPENDIX I: AERIAL PHOTOGRAPHS OF THE STUDY SITE FROM 1998 TO 2004

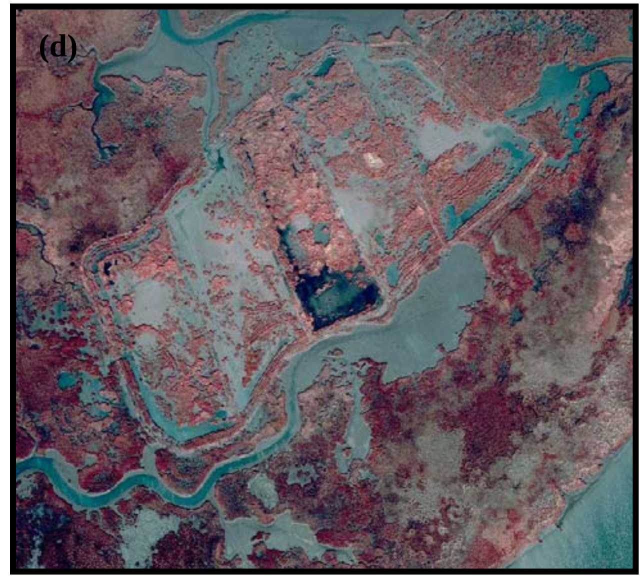
(a)



(b)







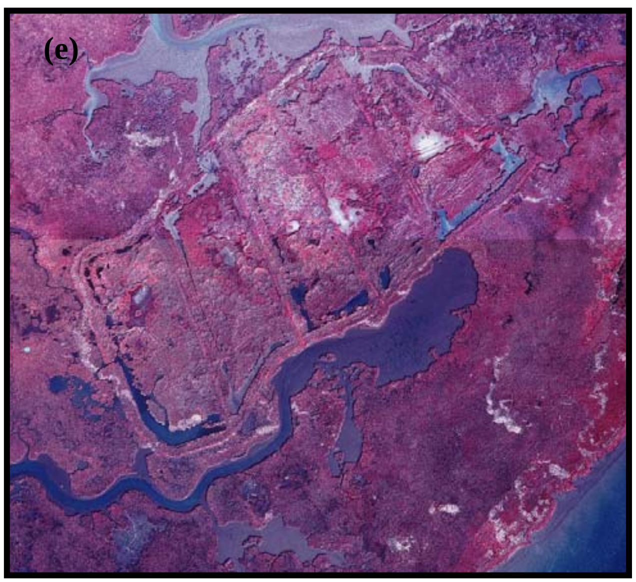


Fig. A1. Aerial view of the study site a) before the brown marsh event (Feb. 4, 1998), b) after the brown marsh event (Nov. 8, 2001), c) five months after the sediment-slurry addition (April 14, 2003), d) 12 months after the sediment-slurry addition (Nov. 28, 2003), and e) 25 months after the sediment-slurry addition (Dec. 11, 2004).

APPENDIX II: AUGUST THRU DECEMBER 2004 HYDROPERIOD

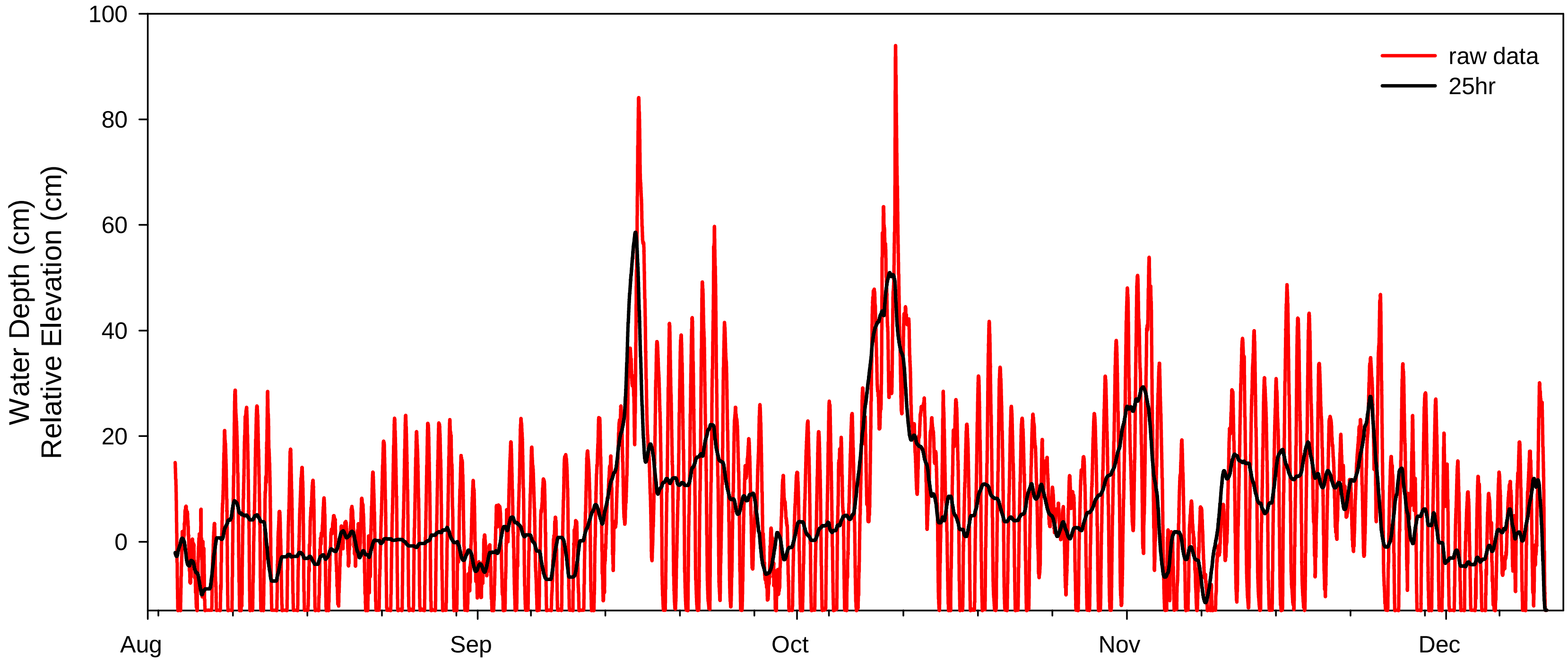


Fig. A2. Water depths (— raw data) and water depths averaged over a 25-hour time period (— 25 hr) referenced to a common datum are presented from August thru December of 2004. Negative values indicate the extent of marsh draining. The detection limit for the water sonde was -13 cm relative to the ambient marsh surface.

VITA

Angela Marie Schrift was born on 25 September 1979, in St. Petersburg, Florida, to Samuel and Kathleen Schrift. She has an older sister, Kristina, and a younger brother, David. Angela completed a bachelor of science in environmental studies with a concentration in natural science at the University of West Florida in 2002. While completing her bachelor's degree she worked for the Florida Department of Environmental Protection's biology laboratory sampling waterbodies for macroinvertebrates, which were used to assess watershed health. It was there that she became interested in the restoration of ecological systems and decided to continue her education. In August of 2002, she received a research assistantship and began working under Dr. Irving Mendelssohn at Louisiana State University towards a master's degree in oceanography and coastal sciences. In September 2005, she accepted a position as the Florida Division of Forestry's Plant Conservation Program Biologist. Angela will complete her master's degree in August 2006.