INTRODUCTION

Background

Coastal salt marshes are complex ecosystems located at the interface between terrestrial and marine habitats. Salt marshes are defined as areas vegetated by herbs, grasses, and low shrubs, which border saline water bodies, and are subjected to periodic water level fluctuations caused by either tidal or non-tidal events (Adam, 1990). Typically, they are located within low energy intertidal zones in which accretion is sufficient to maintain surficial elevation above relative mean sea level. Salt marshes most commonly occur in temperate and high latitude estuaries of open coasts, which are protected from extreme wave action by wide intertidal flats or barrier complexes (Allen and Pye, 1992). These marsh systems are characterized by fine sediments and halophytic vegetation and are maintained by a combination of physical and biological processes (Reed, 1990). Marshes exhibit complex zonation of plants, animals, and microbes, all tolerant to the stresses of salinity variations, tidal fluctuations, and extreme daily and seasonal temperature changes (Mitsch and Gosselink, 1993).

Marsh systems are critical natural resources that provide significant ecological, economic, and social benefits. Tidal marshes are among the most productive ecosystems in the world, producing annually up to 80 metric tons per hectare of plant material in the southern Coastal Plain of North America (Mitsch and Gosselink, 1993). Ecologically, marsh systems function as net primary producers, major producers of food web detritus, fauna refuge, protective baffling, and physical and chemical filtering. The economies of numerous coastal communities are dependant on coastal fisheries, which rely heavily on a healthy functioning marsh to provide habitat and nursery areas. It has been estimated that more than 90 percent of the commercially important fish and shellfish of the southeast Atlantic and Gulf coasts are either estuarine or salt marsh dependent at some point in their lives (Mitsch and Gosselink, 1993). These areas are valued also for their recreational benefits such as boating, fishing, and bird watching, in addition to their aesthetic beauty.

Tidal salt marshes cover approximately 1.7 million hectares within the narrow coastline belt of United States and Alaska (Mitsch and Gosselink, 1993). Coastal plain type marshes characteristic of the Atlantic and Gulf coasts, however, are the focus of this study. The coastal plain group of marshes extends southward from New Jersey along the southeastern coast of the United States to Texas along the Gulf of Mexico (Chapman, 1960; Mitsch and Gosselink, 1993). These systems lie within either a low mesotidal or microtidal range and, therefore, may be largely vulnerable to changes in water level. A primary concern for these lower mesotidal to microtidal marshes is the possibility of submergence and marsh loss due to sea level rise or land subsidence.

The overall question of coastal salt marsh survival or submergence depends on numerous factors such as geomorphology, sediment supply, and vegetation. Stevenson et al. (1986) suggested that the amount of tidal energy is equally important in determining rates of marsh accretion. Microtidal environments, such as coastal North Carolina, tend to experience lower sediment inputs resulting in greater sediment deficits. In these microtidal areas, sporadic sedimentation during major wind-driven storm events, such as hurricanes, is usually the critical factor in the marsh sediment budget (Reed, 1989; Friedrichs and Perry, 2001). Nonetheless, in many storm-impacted systems, accumulation still remains insufficient to maintain an elevation in equilibrium with sea level rise.

The continued existence of marsh habitat depends on the marsh's ability to maintain its elevation within a specific tidal range through the processes of vertical marsh accretion and deposition. Submergence occurs when accumulation rates are unable to keep pace with the relative rise in sea level (DeLaune et al., 1990). A major example of submergence occurs along the northern Gulf of Mexico Coast where the marsh surface is rapidly subsiding due to surface compaction of deltaic sediments and downwarping of the older Pleistocene surface (Mitsch and Gosselink, 1993). In addition, the situation is exacerbated by the channeling of the Mississippi River and the transport of its sediments into deep offshore areas, thus starving these systems of their sediment supply. Artificial marker horizons and ^{137}Cs dating have been used to estimate average accretion rates of 0.8 cm yr^{-1} in a Louisiana marsh where coastal submergence rates averaged 1.2 cm yr^{-1} (DeLaune et al., 1983). In a separate study, Bauman et al. (1984) observed that marsh accretion was insufficient to keep pace with sea level rise in another Louisiana marsh.

In the southeastern U.S., marsh submergence has been documented as well. Stevenson et al. (1985) observed accretion rates ranging from 1.7 to 3.6 mm yr^{-1} within a Chesapeake Bay marsh, and concluded that these rates were not sufficient to keep pace with sea level rise (3.9 mm yr^{-1}) in that region. In other southeastern U.S. marshes, the rate of sediment accretion is approximately 1.2 mm yr^{-1}, which is less than the average (1.9 mm yr^{-1}) rate of sea-level rise for the region (Stevenson et al., 1986; Hackney and Cleary, 1987).

Inorganic sediment deficits are one contributing factor to marsh deterioration. Inorganic sediment deficits decrease sediment bulk densities and lower surface elevation thereby resulting in an increased flooding duration. When a marsh is inundated for longer periods of time, the pore waters become more reduced due to the lack of atmospheric oxygen inputs (Faulkner et al., 1989; Friedrichs and Perry, 2001). The reduced environment of a degraded marsh system can also result in significant free sulfide accumulation as well as root oxygen deficiencies (McKee and Mendelssohn, 1988). These sulfides, commonly associated with reduced marsh environments, have been shown to have harmful effects on wetland plants (Howes et al., 1981). When sulfides accumulate to toxic concentrations, plant nitrogen uptake and assimilation can be inhibited, and plant growth limited (Howes et al., 1981; McKee and Mendelssohn, 1988). Studies have long documented such effects on the coastal marshes of southern Louisiana, as well as marshes bordering the east coast and Chesapeake Bay, where loss is occurring at an alarmingly high rate (DeLaune et al., 1983; Stevenson et al., 1985; Cahoon and Reed, 1995; Ward et al., 1998).

The net effect of anoxia is plant die-back, which reduces above and belowground biomass and accelerates marsh subsidence (DeLaune et al., 1990; Conner and Chmura, 2000). Moreover, declining plant productivity will also have a deleterious effect on the plant's ability to baffle flow and, consequently, on sediment retention on the marsh surface. Typically, deposition rates of inorganic sediment increase with grass stem density (Leonard and Luther, 1995; Friedrichs and Perry, 2001). Therefore, in a system of reduced biomass, the reverse is likely to occur.

Inorganic sediment deficits may be especially devastating when coupled with sea level rise and anthropogenic practices that further deplete sediment supply to marsh surfaces. Cahoon and Reed (1995) state that accretion in systems remote from riverine sediment sources appear to rely largely on resuspension of existing materials from the bottom of adjacent creeks and bays. In southeastern U.S. barrier island systems, many back barrier marshes are removed from significant fluvial sources of sediment. Therefore, in these systems, marsh substrate is derived via overwash deposits from nearshore and abandoned flood tide deltas (Hackney and Cleary, 1987). Currently, barrier island management practices such as high density development, inlet stabilization, and post-storm bulldozing are disrupting these mechanisms and blocking crucial material needed for mash accretion. Engineering practices such as maintenance dredging and jettying of inlets prevents inlet migration and the formation of flood tide deltas upon which marshes may form (Hackney and Cleary, 1987). Moreover, sand that does accumulate in these inlets is commonly viewed as a resource for beach replenishment and is removed from back barrier regions for this purpose.

Mass construction of buildings on barrier islands also stymies back barrier marsh development. The presence of structures physically blocks sediments normally supplied to back barrier marshes by natural overwash processes. Further, any sand that is deposited, is typically bull-dozed back to the beach to form temporary dunes to protect threatened buildings. These practices, when coupled with continued rate of sea level rise, may prove disastrous for many coastal salt marshes.

Periodic inorganic sediment application to the surface of a deteriorating marsh is one remedy that may mitigate current submergence problems. The rationale is that added sediment can maintain a marsh elevation within a specific tidal range through the process of artificial marsh accretion (DeLaune et al., 1990). In many coastal systems, sediment dredged from adjacent waterways as part of channel maintenance is ideal for this purpose. These materials are too fine to be disposed of on nearby high-energy beaches. The current trend is for dredged material to be deposited on spoil islands adjacent to dredged waterways significantly higher than the surrounding marsh. This method has proved harmful to some marshes by: 1) directly converting marsh habitat to open water and spoil bank habitat; and 2) indirectly affecting marsh health by altering the local hydrologic regime including: sheetflow over the marsh, subsurface water flow, sediment dispersal, and saltwater intrusion (Cahoon and Cowan, 1988). The creation of dredge spoil islands also reduces nursery habitat that is important to the fishing industry.

With existing spoil deposition sites running low, and an ongoing need to maintain navigational channels, spoil disposal alternatives are necessary. To help alleviate this problem, several U.S. Army Corps of Engineers Districts and members of the dredging industry have proposed that placing dredged material in relatively thin, uniform layers will reduce environmental impacts associated with dredged material placement (Wilber, 1992a). High-pressure spray dredging (Jet-Spray) technology has been proposed as a mechanism to be used as opposed to traditional bucket dredging technologies (Cahoon and Cowan, 1988; Wilber, 1992a; Ford et al., 1999). Since its development, a number of state and federal regulatory agencies, such as the Louisiana Department of Natural Resources, the U.S. Army Corps of Engineers, the U.S. Fish and Wildlife Services, and the National Marine Fisheries Service, have begun to view spray disposal as the primary alternative to conventional disposal methods because it allows the operator to more accurately place disposed material. The high-pressure spray is capable of depositing spoil over an area up to 80-m wide, with a thickness of about 10–20 cm, while avoiding sensitive habitats within the marsh (Cahoon and Cowan, 1988).

The accuracy of dredge disposal is crucial in marsh renourishment to assure that the physical threshold, at which time negative biological impacts occur, is not surpassed. Most coastal vegetation can only tolerate sediment inputs up to a certain limit that, once surpassed, may be lethal (Zhang and Maun, 1989). Furthermore, an addition of too much sediment may convert tidal wetland habitat into upland habitat. Although thin-layer disposal may reduce marsh subsidence, few studies have evaluated its effect on other aspects of marsh function (Cahoon and Cowan, 1988; Wilber, 1992a; Wilber, 1992b; Ford et al., 1999).

In previous sediment addition studies, the sediment additions have ranged from a few millimeters to over 30 centimeters in thickness (Cahoon and Cowan, 1988; Wilber, 1992a; Wilber, 1992b; Ford et al., 1999). DeLaune et al. (1990) observed that raising the surface of a deteriorating *S. alterniflora* salt marsh by 10 cm resulted in a two-fold increase in above ground biomass production after the second growing season. Ford et al. (1999) demonstrated a three-fold increase in percent cover of a deteriorating *S. alterniflora* salt marsh one year after 23 mm of dredged material was applied to the surface. Wilber's (1992a; 1992b) studies were conducted in healthy marsh systems, therefore, resulting in an unnoticeable change in biomass. Further, Wilber (1992a) showed that placing dredged material in a layer generally 5-cm thick did not lead to a significant change in the vegetation community or use of the marsh by animals. The impact of placing dredged material 10-cm thick was less clear (Wilber, 1992a).

Sediment additions may also have a positive effect on the redox potential of marsh soil. Some studies have shown that marsh surface elevation and biomass differences may affect oxygen levels within sediments (Howes et al. 1981; DeLaune et al.1983; McKee and Mendelssohn, 1988). McKee and Mendelssohn (1988) demonstrated this by varying inundation depths of transplanted marsh communities. When the marsh was transplanted to a lower elevation, the soil conditions became more reduced due to increased hydroperiod. Conversely, soil conditions became more oxygenated when the marsh was transplanted at a higher elevation with less inundation (MeKee and Mendelssohn, 1988). Similarly, DeLaune et al. (1983) found higher redox potentials in the surface sediments of a streamside *Spartina alterniflora* marsh compared to an adjoining interior marsh and attributed this difference to higher elevation of the streamside location. In an east coast *S. alterniflora* marsh, Howes et al. (1981) observed that sediments underlying stands of tall *S. alterniflora* were more oxygenated than those underlying the short form at depths of 2, 5, and 15 cm. In Howes et al. (1981), the difference in redox potential was associated with the ability of tall *S. alterniflora* to oxygenate the sediment more efficiently than the short form.

Study Objectives

The results of the aforementioned studies suggest that thin-layer dredge disposable may be a viable solution to offset submergence problems in deteriorating marshes, while having no impact on adjacent healthy marsh systems. The purpose of this study was to investigate the effects of dredged material on the surface of a tidal salt marsh. Specific study objectives were:

1) To examine the effect of sediment placement on *S. alterniflora* density, sedimentation rate, and soil redox conditions in deteriorating and non-deteriorating *S. alterniflora* marsh sites,
2) To evaluate temporal changes in granulometry, stem density, redox potential, and sediment accumulation for treated and non-treated marsh areas, and
3) To constrain the optimal thickness of sediment placement that yield positive benefits for treated deteriorated sites without being deleterious to treated healthy sites.

METHODS

Study Site

This study was conducted in the Masonboro Island component of the North Carolina National Estuarine Research Reserve (Figure 1). Marshes in the study area consist of monospecific stands of *S. alterniflora* that are dissected by numerous tidal creeks and bays. Numerous areas of intertidal flats and oyster bars are prevalent within the tidal creeks and embayments. Sediments in the study area consist mainly of sandy muds (approximately 50 percent fine sand and 50 percent mud). Tides in the study area are low mesotidal with an average mean range of approximately 1.2 m. Astronomical tides at Masonboro are mixed semi-diurnal. The tides within this system are also strongly influenced by wind events, especially during the passage of tropical storms and nor'easters.

For the study, marsh areas were classified as non-deteriorated or deteriorated. Non-deteriorated marsh exhibited dense stands of *S. alterniflora* (> 350 stems m^{-2}) while deteriorated areas were characterized by sparse stands (fewer than 200 stems m^{-2}). Sediment deficits and water logging are probably the controlling factors leading to areas of sparse vegetation as evidenced by the generally more 'waterloggged' nature of the deteriorated substrate. In the study area, stabilized inlets separate undeveloped Masonboro Island from the developed barrier islands of Wrightsville Beach to the north and Carolina Beach to the south. Inlet stabilization practices including jettying and continuous channel dredging have likely restricted the amount of inorganic sediment available to back barrier marshes of the island (Hackney and Cleary, 1987). Occasionally sediment accumulation from overwash occurs; however, the unpredictable nature of strong storms and limited sand supply on the shoreface renders overwash an unreliable source of inorganic sediment in this system. For these reasons, the sediment-starved marshes behind Masonboro Island were ideal for this study.

Experimental Design

The experiment consisted of four treated sample plots each measuring 6.4 m by 6.4 m. Two deteriorated sites (A and B), two non-deteriorated sites (A and B) received sediment additions, whereas the control areas one deteriorated and one non-deteriorated did not receive sediment (Figure 2). Preliminary elevation surveys showed a difference between non-deteriorated and deteriorated sites of approximately 23 cm, with the deteriorated sites being lower. The deteriorated sites were flooded for approximately 1.5 to 2 hours longer then the non-deteriorated sites. Hydroperiods were calculated under the assumption that healthy sites were flooded for 6 hours per tidal inundation. Boardwalks were constructed over sample sites to help eliminate impacts of human disturbance

Figure 1. Map of Masonboro Island. NAD 83 North Carolina State Plane Units (ft).

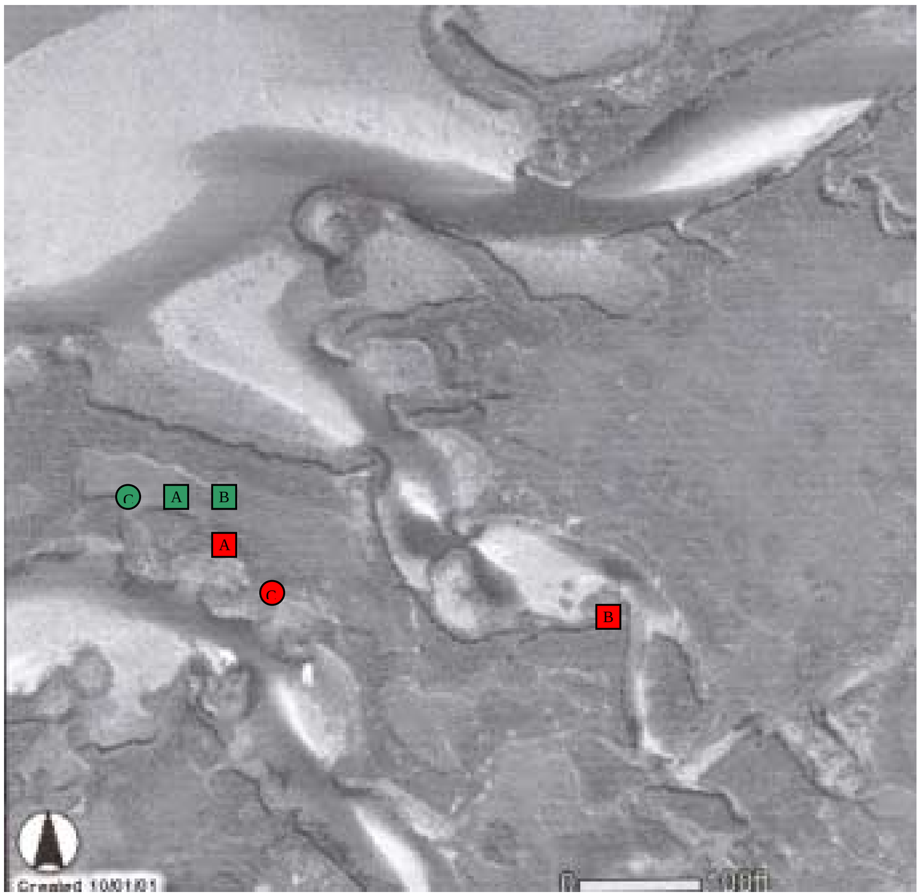


Figure 2. Aerial photograph of study site (non-deteriorated sites shown in green, and deteriorated sites shown in red). Amended sites shown with squares (A and B) and control areas shown with circles (C).

during the study. Due to permit regulations requiring limited impact to marsh areas outside the study sites, manual application was used as opposed to the spray dredge technique discussed in the introduction. Application of sediment occurred during May 2000 when approximately 8.2 m^3 of dredge material was placed on the experimental plots. Placement was always performed at high tide when sufficient water was on the marsh surface to reduce impact on vegetation, to simulate slurry disposal, and to promote uniform distribution. Before application, the sediment was uniformly homogenized to account for any grain size biases between sites. The grain size of the fill sediment consisted of medium to coarse-grained sand. The fill material was transported from a small dredge disposal island about 1 km from the study sites. The sediment was applied as a varying wedge across the cross section of each plot from 0 – 10 cm in thickness (Figure 3). Random coring was conducted post-placement to verify the thickness of deposited material. For the purpose of the study, each plot was divided along the 0 – 10 cm soil gradient into four experimental units: thick, 10 cm; medium, 5 cm; thin, 2.5 cm; and control, 0 cm. Site elevations were determined following sand placement using the non-deteriorated control area as the reference site. The elevations were as follows: non-deteriorated control, 0 cm; non-deteriorate site A, 8.2 cm; non-deteriorated site B, 9.1 cm; deteriorated control, -22.5 cm; deteriorated site A, -13.4; and deteriorated site B, -7.3 cm. Immediately following sediment additions, a predetermined sampling schedule was implemented to monitor changes in canopy characteristics, soil redox conditions, substrate characteristics, and deposition. The sample collection schedule is shown in Table 1.

	May-00	Jun-00	Jul-00	Aug-00	Sep-00	Oct-00	Nov-00	Dec-00	Jan-01	Feb-01	Mar-01	Apr-01
Plant biomass		6/7		8/1		10/9			1/7			4/20
Soil redox potential				8/17		10/23		12/6			3/16	
Particle deposition						10/27 - 10/28					3/23 - 3/24	
Grain size analysis	5/1								1/9			
Hydroperiod evaluation									1/9 - 2/7	1/9 - 2/7		

	May-01	Jun-01	Jul-01	Aug-01	Sep-01	Oct-01	Nov-01	Dec-01	Jan-02	Feb-02	Mar-02	Apr-02
Plant biomass		6/29		8/30		10/30						
Soil redox potential	5/16		7/31		9/26		11/5					
Particle deposition			7/18 - 7/19			10/10 - 10/11						
Grain size analysis												
Hydroperiod evaluation					9/5 - 10/3	9/5 - 10/3						

	May-02	Jun-02	Jul-02	Aug-02	Sep-02	Oct-02
Bulk density						10/7
Tidal velocity					9/13	10/7
Grain size analysis		6/15				

Dredge placement shown in blue

Table 1. Sampling components and sampling dates.

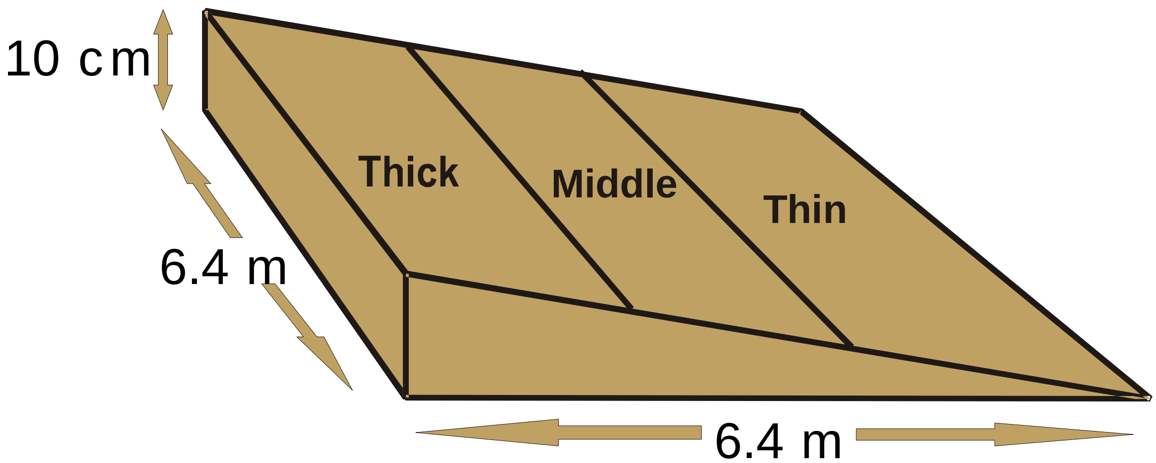


Figure 3. Diagram showing geometry of sediment fill.

Research Components

Physical Parameters

Water levels were monitored on the marsh surface over a four week interval in Winter 2000 and Summer 2001 with RDS water level recorders to quantify differences in hydroperiod between sites following sediment addition. RDS water level recorders were installed at the center of each site and programmed to record water level every 23 minutes. Sediment characteristics, including dry bulk densities and organic content were measured annually to determine any return to pre-addition conditions. Dry and wet sieving was used to quantify the weight percent of material larger than 62.5 µm in diameter. The coarse fraction was dry sieved at whole phi intervals. A LS 230 Beckman-Coulter particle sizing instrument was used to determine the grain size distribution of material finer than 62.5 microns. Folk and Ward grain size statistics were calculated according to methods discussed in Folk (1980). Sediment bulk density was determined by random coring of the study sites to approximately 10 cm. The samples were dried at 60^{o}C, weighed, and density given as gram dry weight per unit volume. Surface samples were also combusted at 500 ^{o}C to determine the percent organic content. A SonTek hand-held acoustic velocity meter was used to measure tidal inundation speeds within the four treated sites and the two non-treated control areas. Velocity data were collected along transects established normal to the creek edge in one non-deteriorated and one deteriorated site. These measurements were included near the end of the study in an attempt to identify a mechanism that might explain some of the unexpected results.

Particle Deposition

Petri-dish sediment traps were deployed to determine sediment deposition rates within the sites according to methods discussed by Reed (1990). Traps consisted of preweighed glass fiber filters attached to petri-dish lids and anchored to the marsh surface using wire staples. Three sets of three replicate traps, were deployed within each treatment (thick, medium, and thin) for each of the four treated sites. An additional set of traps was deployed in each of the non-treated (control) areas. Traps were deployed at low tide and retrieved after 24 hours. Following retrieval, the traps were oven dried at 60 °C and weighed to determine total accumulation (mg cm^{-2} day^{-1}). Organic content was determined by combusting sediment traps at 450 °C for 4 hours. Preliminary deposition rates were determined prior to sediment addition during the spring of 2000, and subsequent deposition rates were monitored quarterly (Table 1). Organic content and total accumulation on traps was evaluated during each of these periods.

Chemical Parameters

Soil redox potential (eH) was examined using a portable voltmeter, platinum electrode, and reference electrode (Faulkner, 1989; Langmuir, 1971). Eh measurements were obtained at 2-cm increments from the surface of the sediment to a depth where values became constant. Soil redox profiles were collected bimonthly for each treatment thickness (e.g. thick, medium, thin) in all four treatment sites and at both control sites between August 2000 and November 2001 (Table 1).

Vascular Plants

Stem density, plant height, and numbers of live versus dead shoots were monitored to asses vascular plant response to sediment addition. Preliminary measurements were obtained prior to application, and subsequent measurements were compiled bi-monthly from June 2000 to October 2001 (Table 1). Five replicate 10 x 10-cm quadrats were used to measure stem densities for each treatment thickness within the four study sites and the two controls. Quadrat locations were selected blindly to allow for random sampling. The height of live and dead stems within each quadrat was also measured and recorded.

Statistical Analyses

Paired t-tests were first conducted in mean eH, stem density, stem height, and sediment deposition to determine if significant differences existed between the non-deteriorated sites (A versus B) or the deteriorated sites (A versus B). If no significant difference existed, data from both the A and B site were grouped for subsequent analyses. A one, two, and/or three way Analysis of Variance was then used to determine which study variables significantly differed between the non-deteriorated treated, deteriorated treated, non-deteriorated control, and the deteriorated control. The independent variables used for the ANOVAs included marsh type (non-deteriorated and deteriorated), treatment thickness (high, medium, low, and control), and season (Summer 2000, Winter 2001, and Summer 2001). Three way ANOVAs were conducted first to identify interactions between the different variables. If significant interactions were observed, two and then one way (if needed) Analysis of Variance were conducted. A Post Hoc LSD test was also performed on individually paired variables to determine significant differences among main effects.

RESULTS

Vascular Plant Stem Density

Vascular plant data were collected bi-monthly in treated and control plots from June 2000 until October 2001. Survey data were subsequently grouped into three categories: summer 1 (first growing season after sediment placement), summer 2 (second growing season) and winter (the winter between growing seasons). The mean stem densities of all treatments combined (excluding controls) over the entire first growing season (June 2000 – October 2000) were 149 stems m^{-2} and 256 stems m^{-2} in the deteriorated and non-deteriorated sites, respectively (Table 2). Mean stem densities in the control plots for this time period were 137 and 200 stems m^{-2} for the deteriorated and non-deteriorated controls, respectively. Over the first growing season, mean stem densities were significantly ($p<0.0001$, $F= 27.22$) lower in the deteriorated sites compared to the non-deteriorated sites. When mean stem densities were compared among treatment types (i.e., thick, middle, thin, and control) for the first season, the densities in the non-deteriorated sites were significantly greater ($p= 0.0001$) than densities in the deteriorated sites for all treatment types (Table 3).

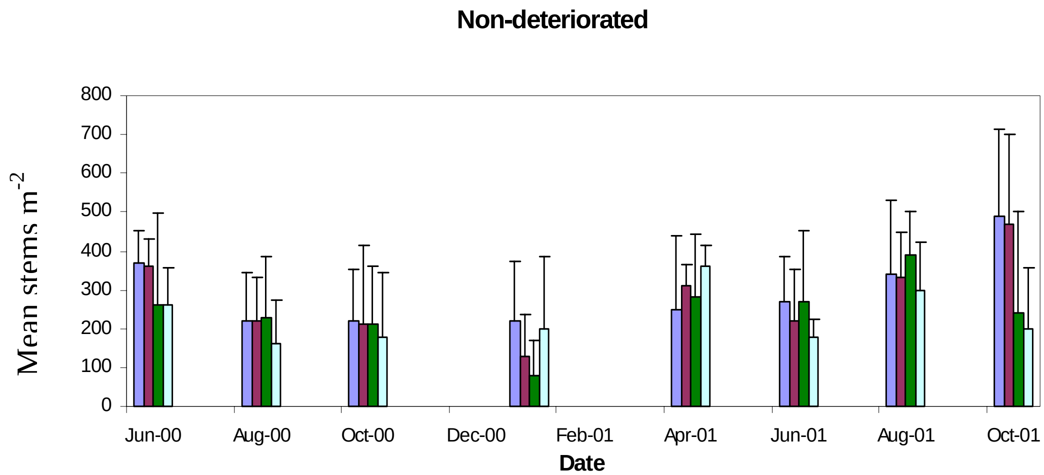
As expected, the number of living shoots in both non-deteriorated and deteriorated sites decreased during the winter season. This trend was observed for all treatment types and also for the controls. During the winter (November 2000 – April 2001), mean stem densities for sites that received sediment additions were significantly higher in the non-deteriorated (212 stems m^{-2}) sites than in the deteriorated sites (137 stems m^{-2}). Mean stem densities for treated sites were consistently higher than corresponding control sites, although the difference was not always significant (Figure 4). The thickness of sediment placed on the surface of each marsh type did not significantly affect winter stem densities in either the non-deteriorated or the deteriorated plots.

	Mean (cm)	Std Error (cm)
ND-T Sum 1	255.5	16.6
ND-T Sum 2	335.5	16.6
ND-C Sum 1	215	35.4
ND-C Sum 2	226.6	40.9
DET-T Sum 1	151.1	23.6
DET-T Sum 2	308.9	23.6
DET-C Sum 1	130	35.4
DET-C Sum 2	113.3	40.9

Table 2. Plant density means and Std Error for non-deteriorated and deteriorated amended sites and control areas for the 2000 and 2001 growing season.

Variable	df effect	MS effect	dF Error	MS Error	F	p-level
1	1	942156.9	546	23337	40.37	.000000
2	3	142348.9	546	23337	6.09	.000437
3	2	595135.1	546	23337	25.50	.000000
12	3	31967.9	546	23337	1.37	.251088
13	2	48266.7	546	23337	2.07	.127396
23	6	59496.7	546	23337	2.55	.019185
123	6	17026.9	546	23337	0.73	.625912

Table 3. One, Two, and Three way ANOVAs for plant density. Variable 1 is marsh type (non-deteriorated or deteriorated), variable 2 is treatment level (thick, medium, thin, and control), and variable 3 is season (summer 1, winter, and summer 2).



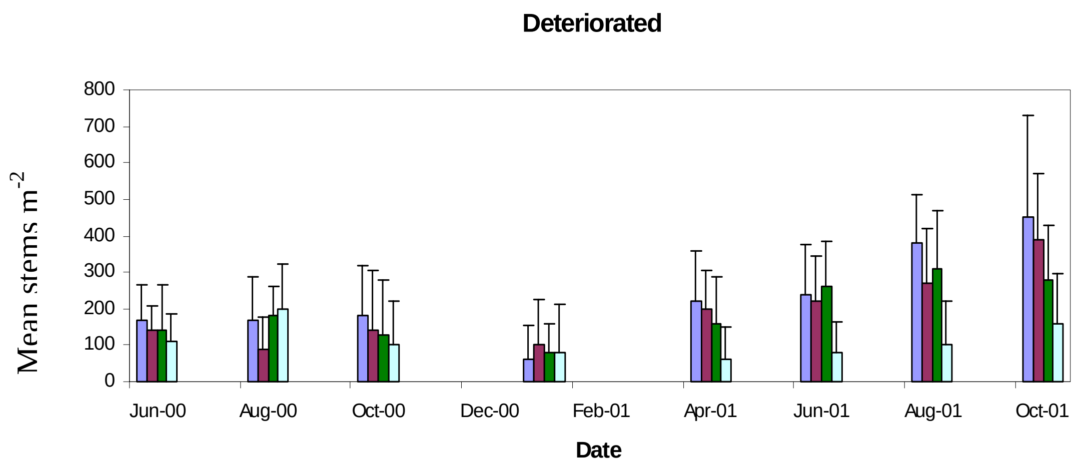


Figure 4. Mean stem densities of non-deteriorated and deteriorated sites. Thick treatments are shown in blue, medium treatments in red, thin treatments in green and controls in aqua. Error bars indicate + one standard deviation.

During the second growing season (May 2001 – Oct 2001), stem densities in the control sites were 226 stems m^{-2} for the non-deteriorated and 113 stems m^{-2} for the deteriorated site (Table 2). These means were not significantly different from the mean stem densities reported for the first growing season for either deteriorated or non-deteriorated controls. Mean stem densities increased for all sites that received sediment additions. The increase was most profound in the deteriorated plots (Figure 4). Seasonal density box plots comparing treated non-deteriorated and deteriorated sites and control sites are shown in Figure 5. Mean stem densities for all treatment types (excluding controls) in both the non-deteriorated and deteriorated sites increased significantly from the first growing season. The mean stem density for the non-deteriorated sites was 336 stems m^{-2} and the deteriorated mean stem density was 309 stems m^{-2} (Table 2, Figure 5). By the end of the second growing season, no significant difference in mean stem density was detected between the non-deteriorated and deteriorated sites that received sediment additions (Table 4). Further, all of the deteriorated plots that received sediment additions showed significant increases in stem density (Figure 5) while stem densities in the control sites did not significantly change (Table 4 and 5).

When mean stem densities for individual non-deteriorated and deteriorated treatment levels (i.e. thick, middle, thin) were compared, all cases proved non significant with the exception of the non-deteriorated-thick treatment, which had significantly more *Spartina* shoots than the deteriorated-thin. In addition, the deteriorated-thick had significantly more biomass than the deteriorated-thin (Table 5). It is apparent that by the second growing season the convergence of stem densities in the non-deteriorated and the deteriorated sites was greatest for the areas that received the thickest additions (Figure 6).

Vascular Plant Height

Vascular plant heights were measured bi-monthly from June 2000 until October 2001. Plant heights within each of the four study sites were lowest in the winter and highest in the two summer growing seasons. Mean plant heights in the non-deteriorated were significantly greater than mean plant heights in deteriorated sites (Figure 7, Table 6). The Students t-test indicated significantly greater plant heights for both summers in the non-deteriorated sites than in the deteriorated sites, but no difference in plant height between marsh types during the winter.

Comparisons of growing season means yielded no significant difference in plant height between the first and second growing season (summer 1 and summer 2) for treated areas in both marsh types. Further, the thickness of emplaced sediment did not significantly affect plant height in either marsh type. In the control areas, mean plant heights in the non-deteriorated control did not differ appreciably between the first growing season (72.7 cm) and the second growing season (69.7 cm). The deteriorated control area, however, experienced a significant decrease in mean height between the first (49.8 cm) and second (28.1 cm) growing seasons (Table 6). For both summer seasons, non-deteriorated and deteriorated control mean plant heights were significantly different from one another, $p = 0.01$ ($F = 5.33$) and $p = 0.0001$ ($F = 18.17$), for summer 1 and summer 2, respectively.

ND-T Sum 1	ND-T Sum 2	ND-C Sum 1	ND-C Sum 2	DET-T Sum 1	DET-T Sum 2	DET-C Sum 1	DET-C Sum 2	
	34.00	-35.73	-57.17	60.67	7.33	49.27	56.16	ND-T Sum 1
		44.27	22.83	140.67	-19.33	129.27	136.16	ND-T Sum 2
			-93.73	-10.17	17.61	-12.58	-3.73	ND-C Sum 1
				-8.28	-3.85	-8.73	0.66	ND-C Sum 2
					114.00	-57.39	-50.50	DET-T Sum 1
						102.61	109.50	DET-T Sum 2
							-112.68	DET-C Sum 1
								DET-C Sum 2

Table 4. Comparison of mean stem density for non-deteriorated and deteriorated amended sites and control areas for the 2000 and 2001 growing season using the Student's t-test (Abs(dif)-LSD). Significant differences are the positive values shown in gold.

DET S1 Thick	DET S1 Mid	DET S1 Thin	DET S1 Con	DET S2 Thick	DET S2 Mid	DET S2 Thin	DET S2 Con	
	-29.53	-56.20	-45.59	103.80	40.47	23.80	-37.41	DET S1 Thick
		-52.86	-82.25	153.80	90.47	73.80	-87.41	DET S1 Mid
			-68.92	94.41047	31.07713	14.41047	-60.74	DET S1 Thin
				127.14	63.80	47.14	-131.83	DET S1 Con
					-16.20	0.47	243.33	DET S2 Thick
						-16.20	129.26	DET S2 Mid
							65.93	DET S2 Thin
								DET S2 Con

Table 5. Comparison of mean stem densities for all deteriorated treatment levels and control areas for the 2000 and 2001 growing season using the Student's t-test (Abs(dif)-LSD). Significant differences are the positive values shown in gold.

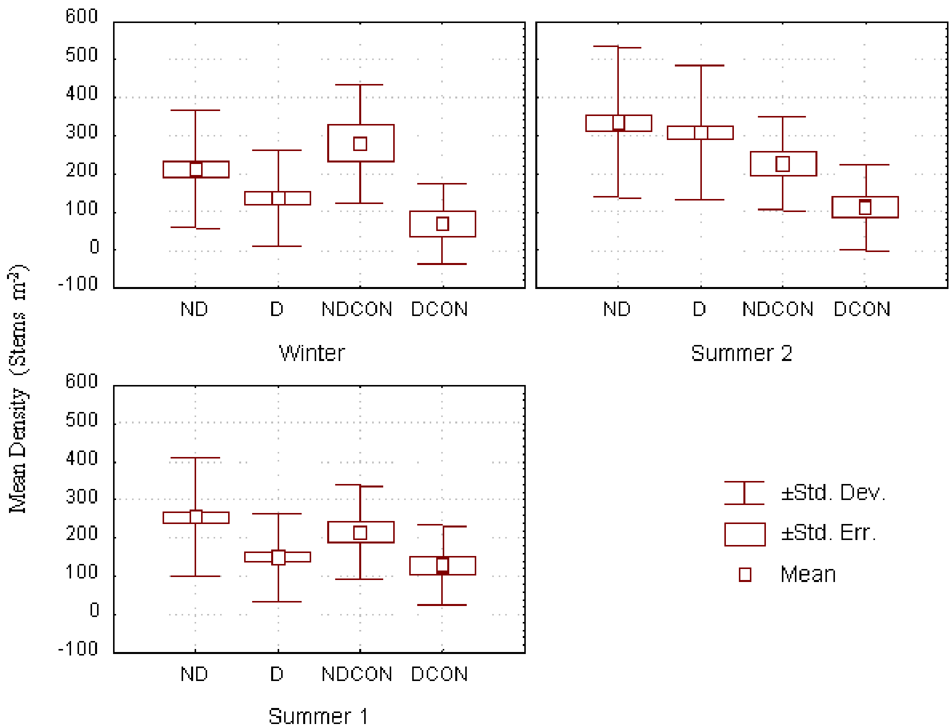


Figure 5. Box plot showing mean seasonal stem densities for non-deteriorated, deteriorated, and control sites.

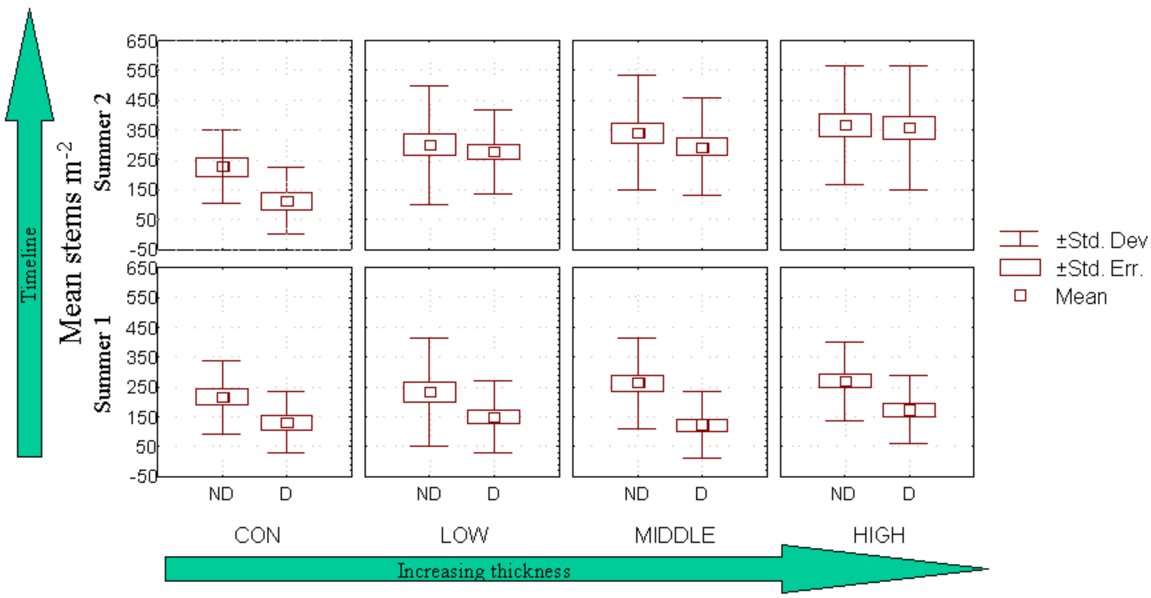


Figure 6. Box plots of mean stem densities by treatment level in summer 1 and summer 2.

Site	Season	Mean (cm)	Std Error	Control (cm)	Std Error
ND	Summer 1	69.0	2.9	72.7	6.2
ND	Winter	46.3	3.6	46.9	8.8
ND	Summer 2	66.0	2.9	69.7	7.2
DET	Summer 1	45.6	2.9	49.8	6.2
DET	Winter	23.5	3.6	15.3	8.8
DET	Summer 2	49.5	2.9	28.1	7.2

Table 6. Mean plant heights and Std Error in treated and control areas for both deteriorated and non-deteriorated marsh types by season.

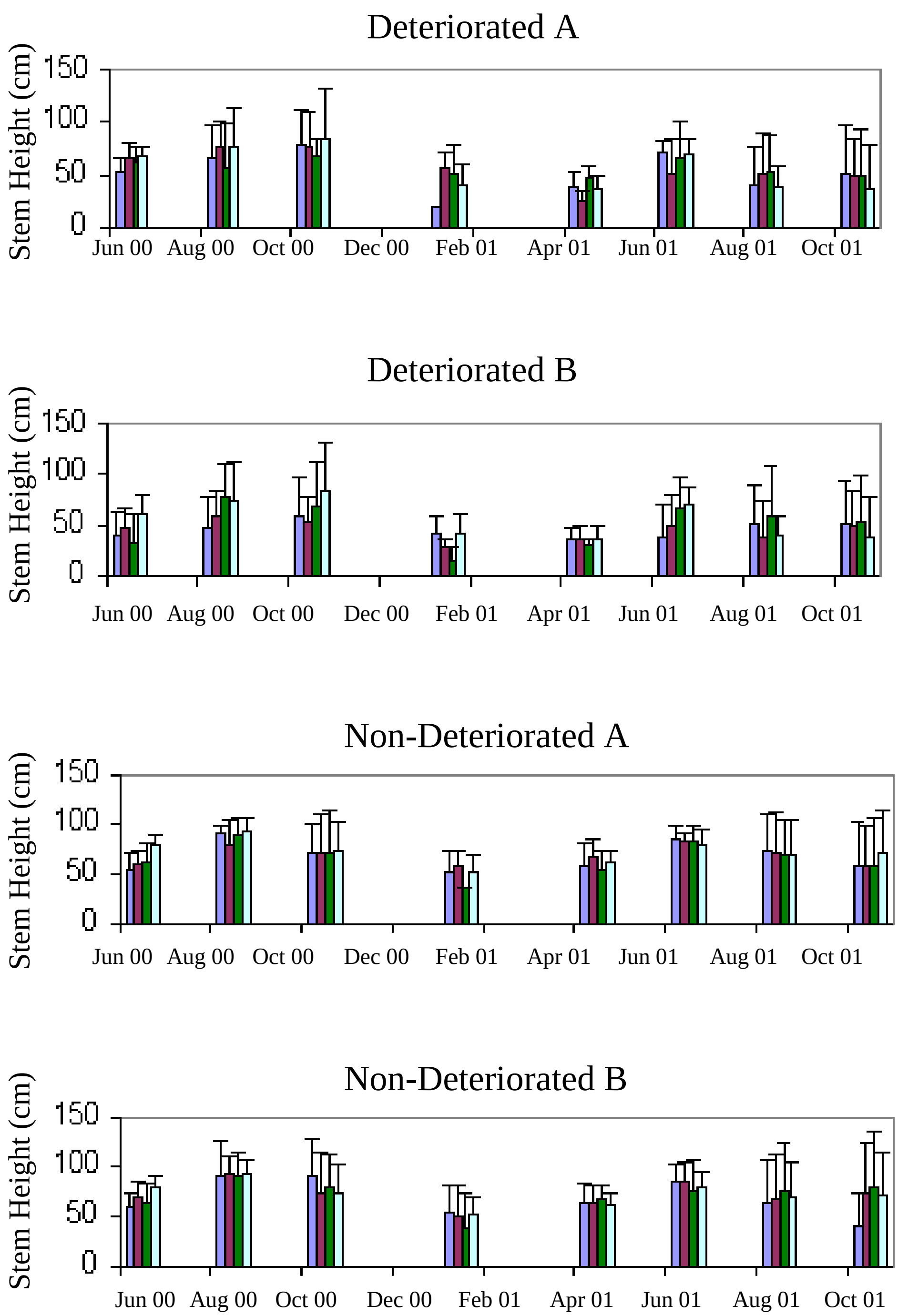


Figure 7. Monthly mean plant heights for various treatments in deteriorated and non-deteriorated marsh types. Thick treatments are shown in blue, medium treatments in red, thin treatments in green, and controls in aqua. Error bars indicate + one standard deviation.

Chemical Parameters

The redox potential of the marsh sediments was recorded bi-monthly from August 2000 to November 2001. Eh profiles were generated for each treatment (i.e. thick, middle, and thin) in each of the four sites, and both the non-deteriorated and deteriorated controls. For statistical comparisons of eH between treatment and marsh types, the upper 10 cm of each profile was vertically averaged. Ten centimeters was selected because the greatest variability in eH usually occurred above this depth. Figure 8 shows eH profiles temporally averaged over the entire study period for both non-deteriorated and deteriorated sites. In general, the sediment became more reduced (anoxic) with depth. Sediments in the non-deteriorated sites that received sediment additions exhibited higher eH levels (i.e., more oxygenated) than sediments in the deteriorated sites that received sediment additions. This difference in eH was highly significant ($p<0.0001$, $F=248.93$). Sediments in the deteriorated control were more reduced than sediments in the non-deteriorated controls.

During the Winter 2001 sampling season (December 2000 to March 2001), eH profiles became more oxygenated compared to summer 1. Further, the mean eH level for most sites increased in summer 2 compared to the first growing season after sediment addition. Although not always significantly elevated, this trend was generally observed in both non-deteriorated and deteriorated marsh types whether or not sediment additions were applied (Figure 9). For the control sites, the mean eH values in summer 1 in the non-deteriorated marsh was –57.3 mV, while the depth integrated mean eH values for the deteriorated controls were –125.4 mV, -62 mV, and -94.4 mV for summer 1, winter, and summer 2, respectively. In the control sites, mean eH values were significantly higher in the non-deteriorated areas compared to the deteriorated areas except during summer 1. During this time, the mean eH in the non-deteriorated control was not significantly different from the mean eH measured in the deteriorated controls. The thickness of sediment addition appears to have influenced eH levels in the treated marshes. A one-way ANOVA resulted in highly significant differences ($p<0.0001$, $F=16.54$) among all of the treatments when averaged for all amended and control sites. For all three seasons, the thickest treatment in both the deteriorated and non-deteriorated sites exhibited the highest mean eH value. In general, the most oxygenated profiles were associated with thicker treatments, while a more reduced eH profile was associated with thinner treatments (Figure 10).

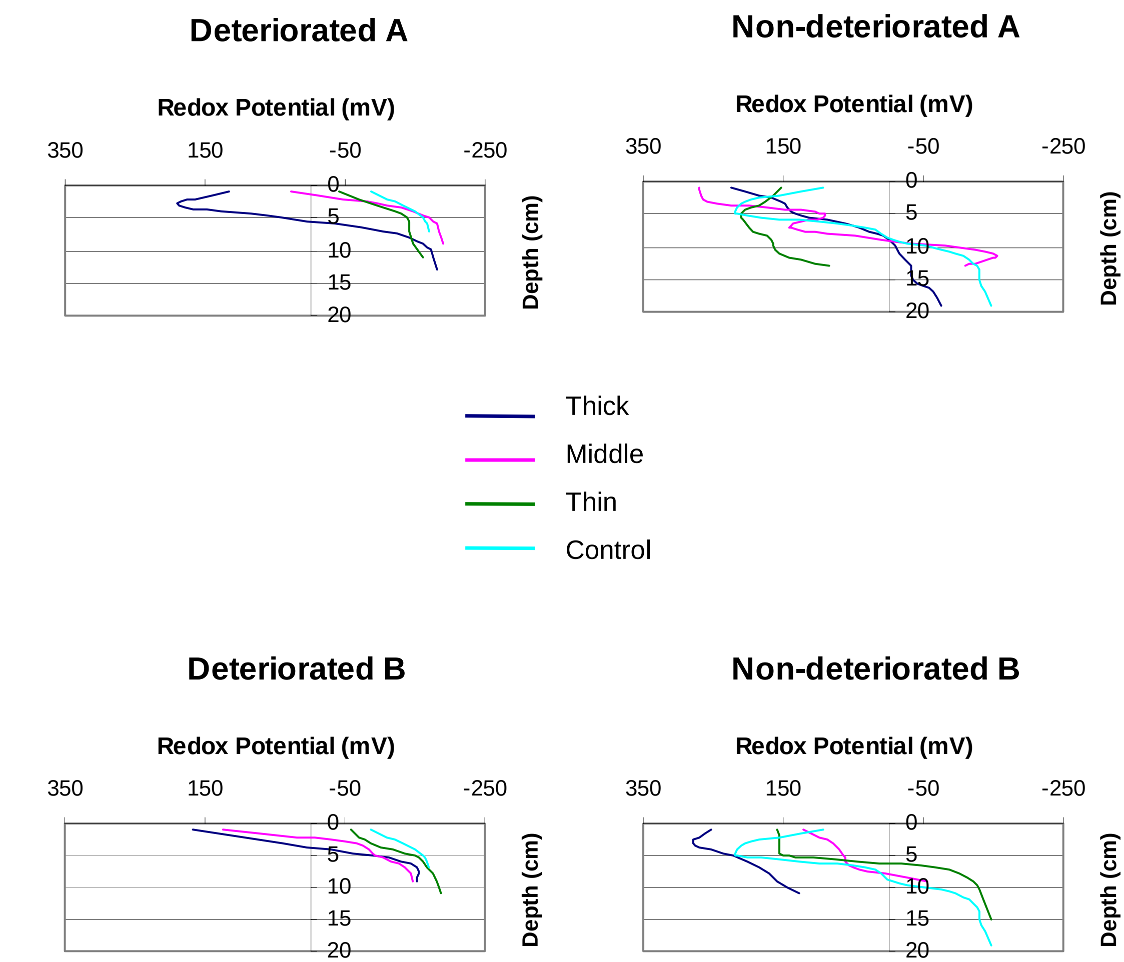


Figure 8. Temporally averaged eH profiles for each study site (August 2000 – November 2001).

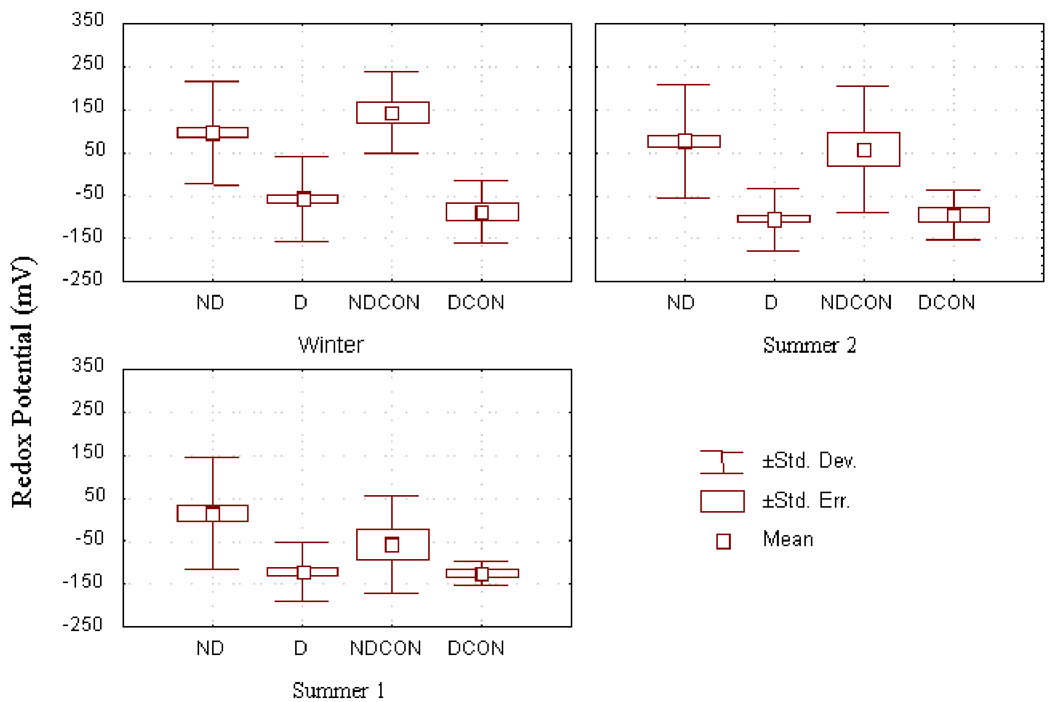


Figure 9. Box plots of depth averaged redox potential for treated (averaged) and control non-deteriorated and deteriorated sites by season.

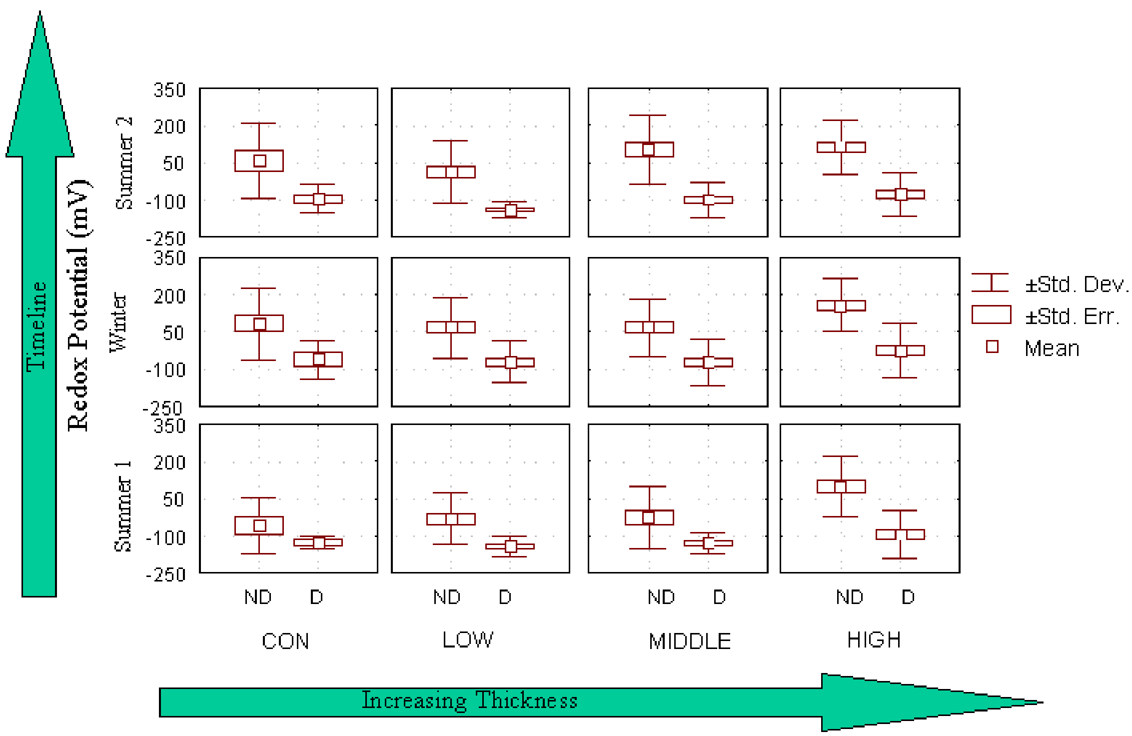


Figure 10. Temporally averaged eH values comparing seasonal trends within treatment and marsh type.

Additional eH analyses were made using eH levels measured only within the introduced sand wedge layer and then comparing these to depth-integrated eH levels measured over the same depth in non-amended controls. For example, when controls were compared to the "thick" treatment, eH profiles for both the treatment and control were depth-averaged over 10 cm because this was the mean thickness of the sediment addition for this treatment. When controls were compared to the "middle" and "thin" treatments, all profiles were depth-averaged over 5 cm and 2 cm, respectively. Using this approach, the treated areas generally exhibited greater redox potentials than the non-amended controls. This was the case for 10 out of the 12 comparisons (Figure 11). The two exceptions occurred when the deteriorated thin (B) treatment and the non-deteriorated thin (B) treatment were compared with their respective controls.

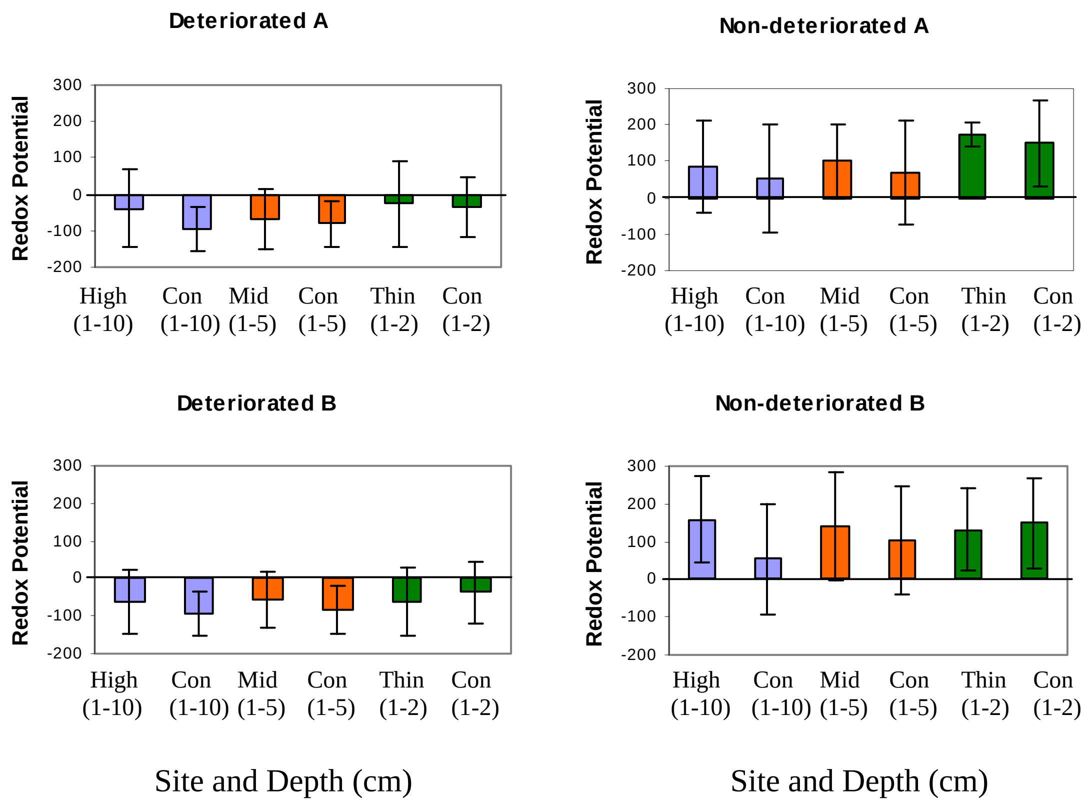


Figure 11. Mean depth-integrated redox potentials measured within the sediment addition layer and an equivalent depth in the associated control. Error bars indicate $\pm$ one standard deviation of the mean. A high degree of seasonal variation and the decreasing exponential shape of the profiles account for the large standard deviations shown.

Particulate Deposition Rate and Organic Content

Particle deposition was monitored quarterly from October 2000 through September 2001. Both total deposition and percent organic content of deposited materials were determined for each of the four sampling dates. Particle deposition was greater in the deteriorated sites (treatments and control) than in the non-deteriorated sites for all 3 seasons (Figure 12).

Total deposition also was significantly lower during the winter sampling than during summer sampling with non-deteriorated and deteriorated mean treatment means of 17.8 grams m^{-2} day^{-1} and 25.6 grams m^{-2} day^{-1}, respectively, and control means of 16.9 grams m^{-2} day^{-1} and 39.6 grams m^{-2} day^{-1}, respectively. Mean deposition rates did not significantly differ between the first growing season and the second growing season in either control area. Further, mean deposition rates in summer 1 were not significantly different from mean deposition rates measured during summer 2 in the non-deteriorated sites that received sediment additions (Figure 12). The highest deposition rates recorded during the entire study occurred in the deteriorated sites in summer 1 when mean deposition rates of 297, 259, and 306 grams m^{-2} day^{-1} were recorded for the thick, medium, and thin treatments, respectively. These values were significantly higher than comparable sites in the non-deteriorated marsh ($p < 0.0001$). In summer 2, mean deposition rates in the sediment amended deteriorated sites decreased to 182, 211, and 132 grams m^{-2} day^{-1} for the thick, medium, and thin treatments, respectively. The difference between summer 1 and summer 2 means was significant in the deteriorated amendment sites ($p<0.0005$, $F=15.94$). Further, the last sampling period (October 2001) showed no statistical difference when comparing amended deteriorated sites to either summer 1 or summer 2 controls for both marsh types (Figure 13). Deposition rates in the treated deteriorated sites remained significantly higher than deposition rates in corresponding non-deteriorated rates in summer 2 (Figure 12). Two-way ANOVAs between marsh type (i.e., deteriorated or non-deteriorated) and treatment (i.e., sediment thickness); and marsh type and season indicated significant interaction (Figure 7). The interaction between marsh type and season was especially high relative to the statistical main effects. When this relationship was analyzed for individual seasons, a pattern of decreasing sedimentation in the amended deteriorated sites was observed.

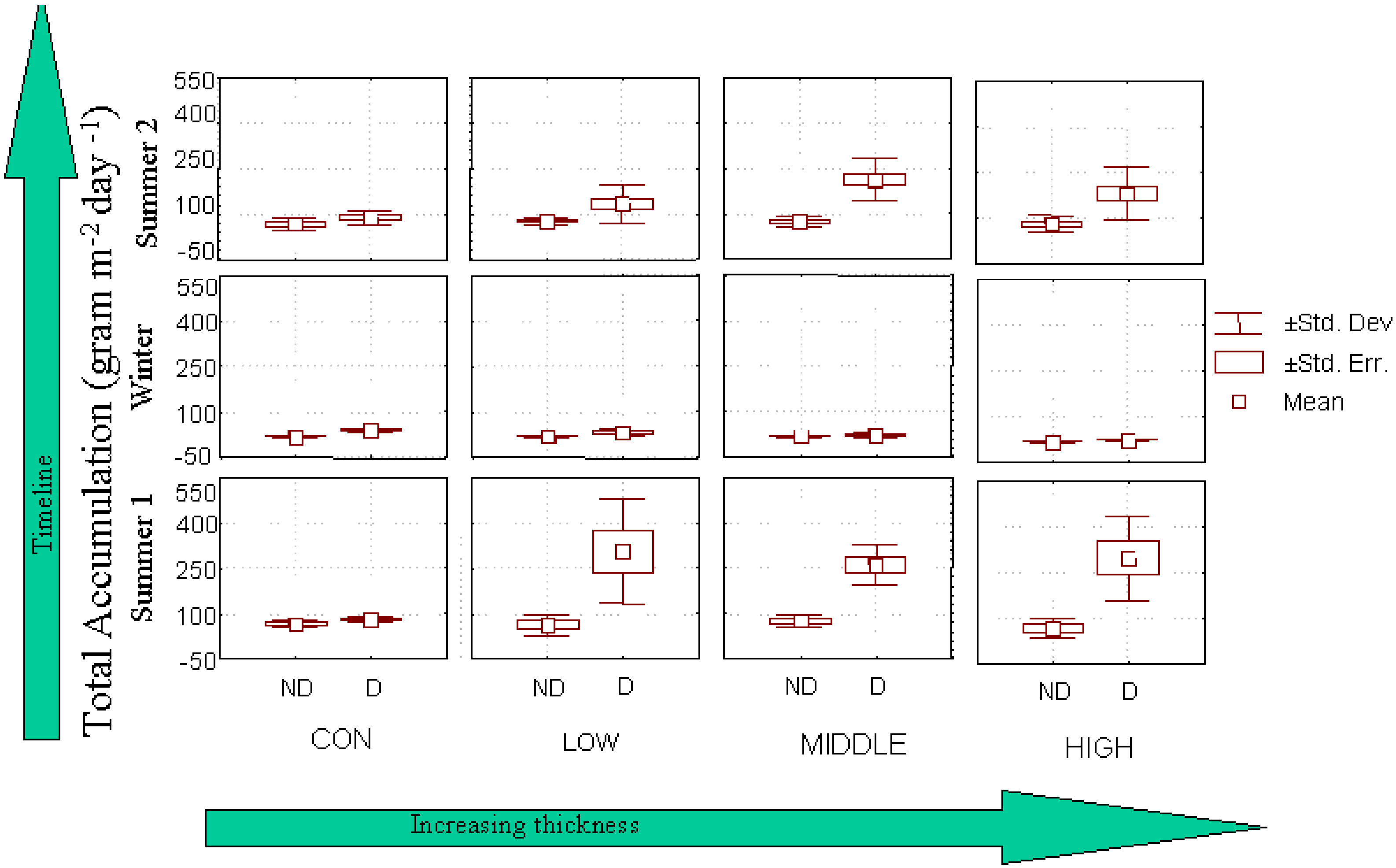


Figure 12. Surficial sediment deposition measured in deteriorated (D) and non-deteriorated (ND) experimental marsh sites.

Variable	df effect	MS effect	dF Error	MS Error	F	p-level
1	1	256037.3	144	3354.334	76.330	.000000
2	3	14803.6	144	3354.334	4.41	.005309
3	2	172042.7	144	3354.334	51.29	.000000
12	3	12413.3	144	3354.334	3.70	.013269
13	2	59468.2	144	3354.334	17.73	.000000
23	6	7898.3	144	3354.334	2.35	.033648
123	6	8704.7	144	3354.334	2.59	.020356

Table 7. One, Two, and Three way ANOVAs for total deposition. Variable 1 is marsh type (non-deteriorated or deteriorated), variable 2 is treatment level (thick, medium, thin, and control), and variable 3 is season (summer 1, winter, and summer 2).

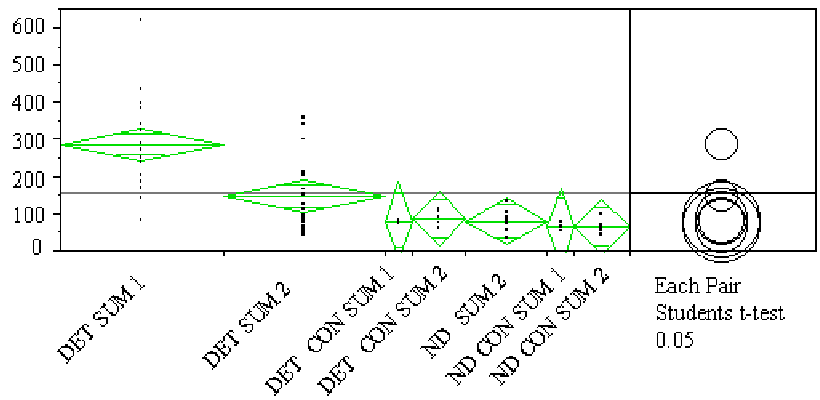


Figure 13. Post Hoc Students t-test comparing sedimentation of amended deteriorated sites (October 2001) with non-deteriorated and deteriorated controls (summer 1 and 2), amended deteriorated sites (summer 1), and amended non-deteriorated sites (summer 2).

Significant differences in the percent organic content of deposited materials were found between marsh type (variable 1), treatment levels (variable 2), and season (variable 3), as well as interactions between marsh type and treatment level (variables 1 and 2), marsh type and season (variables 1 and 3), and marsh treatment and season (variables 2 and 3), although, the level of significance was much higher in the main effects (Table 8). When comparing main effects, mean percent organic content percentages were highest in the winter with values approaching 100 percent for all treatments and controls in both marsh types. These values were approximately one order of magnitude greater than percentages observed in either the first or second growing seasons. The difference in mean organic content in treated deteriorated and non-deteriorated sites between the first and second summers was highly significant ($p < 0.0001$). The non-deteriorated treatments exhibited greater organic content percentages than the deteriorated locations in both summer seasons, although the difference between the marsh types during summer 2 was approximately one-half the difference measured during summer 1.

The deposition of organic material on sediment traps in the deteriorated sites appears to have increased for all treatment levels from the first to second growing season (Figure 14). This increase in organic content deposition, however, was significant only in the thin treatment. The deteriorated control area exhibited no significant difference in organic deposition between summer 1 and summer 2. For the non-deteriorated sites, there was no significant difference in treatment type between the two growing seasons, although organic deposition in the non-deteriorated control significantly increased over this time ($p < 0.0001$). When treatments levels (high, middle, and low) were averaged for both sites (i.e., deteriorated and non-deteriorated) and compared, a significant increase in the deteriorated site was found during the second growing season when compared to the first. No difference was found when comparing the same two seasons for the non-deteriorated site.

Sediment deposition in treated areas exhibited significantly lower mean organic contents than the non-amended control areas. The percent organic material retained on sediment traps decreased over the treatment spectrum from thin to thick (Figure 15). When a paired LSD test was conducted on all treatment interactions, all were determined to be significant ($p < 0.05$) except for comparisons between the think and medium treatments.

Variable	df effect	MS effect	dF Error	MS Error	F	p-level
1	1	135.2	144	3.825	35.34	.000000
2	3	34.9	144	3.825	9.12	.000014
3	2	117241.4	144	3.825	30650.28	.000000
12	3	11.0	144	3.825	2.88	.038319
13	2	31.8	144	3.825	8.32	.000382
23	6	17.8	144	3.825	4.66	.000231
123	6	6.3	144	3.825	1.65	.136299

Table 8. One, Two, and Three way ANOVAs for organic particle deposition. Variable 1 is marsh type (non-deteriorated or deteriorated), variable 2 is treatment level (thick, medium, thin, and control), and variable 3 is season (summer 1, winter, and summer 2).

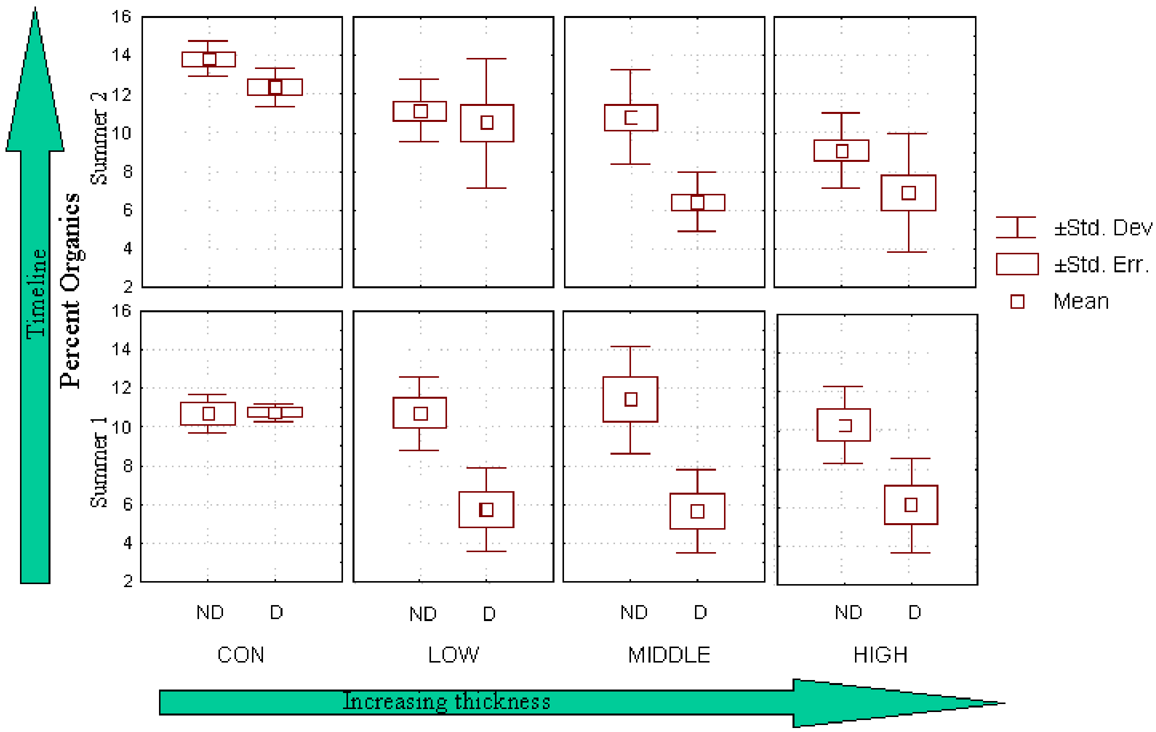


Figure 14. Percent organic content of total deposition for non-deteriorated and deteriorated treatments and control. Data are presented for the first and second growing seasons.

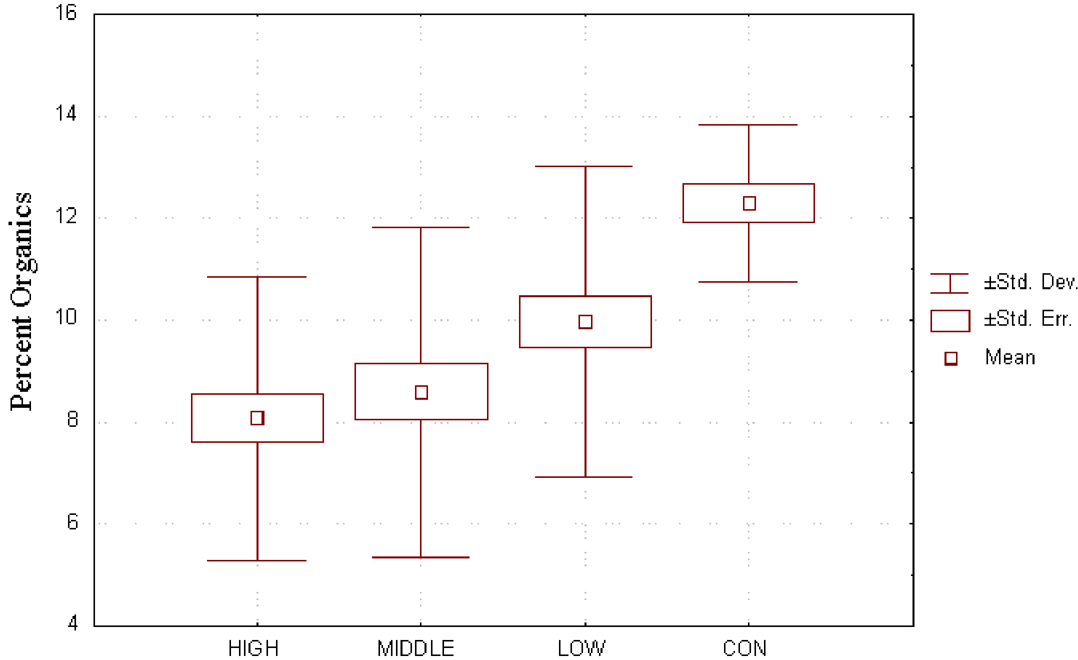


Figure 15. Mean percent organic content of material deposited on sediment traps related to treatment level and control. Means were calculated using data from all sites that received sediment additions.

Grain size of surface sediments

Grain size was determined for the dredge fill material (June 2000), and for surficial sediments of all the non-deteriorated and deteriorated treatment and control areas in January 2001 and again in June 2002. The dredge fill material consisted of medium to coarse sand with no fine sands or muds and a Folk and Ward mean of 0.57 mm. Deteriorated and non-deteriorated control grain sizes from January 2001 were used as a proxy for background conditions because pre-addition grain sizes for these areas were unavailable. These sediments consisted of 50 percent medium to fine sand and 50 percent muds, and a Folk-Ward mean of 0.11 mm for both marsh types. Since sediment placement, the mean grain size of surface sediments has decreased in the sediment amended regions of both the deteriorated and non-deteriorated sites (Figure 16). The most abrupt change occurred between the time of sediment placement and January 2001. Between January 2001 and June 2002, the mean grain size of surface sediments in the control areas also showed a slight decrease, although the change was much less than the change in the treated areas.

The non-deteriorated sites exhibited the most profound changes in grain size post sediment addition. During the final analysis (June 2002), the mean diameter of surface sediments in the non-deteriorated thin treatment (0.088 mm) was most similar to the mean size of non-deteriorated control sediments (0.073 mm). Further, the mean grain sizes of sediments in the non-deteriorated thick, medium, and thin treatments were lower than mean grain sizes of the corresponding deteriorated treatments (Table 9, Figure 16). A trend was also observed in the deteriorated and the non-deteriorated sites that showed the largest diameter means corresponding to areas receiving the most fill-sediment. Areas that received little or no fill material exhibited the lowest mean grain sizes.

Treatment	Jun-00	Jan-01	Jun-02
D-Thick	0.565	0.344	0.332
D-Med	0.565	0.337	0.304
D-Thin	0.565	0.282	0.188
D-Con		0.112	0.067
ND-Thick	0.565	0.280	0.212
ND-Med	0.565	0.271	0.170
ND-Thin	0.565	0.182	0.088
ND-Con		0.107	0.073

Table 9. Mean grain size of the dredged fill material, January 2001 sampling, and June 2002 sampling. For the January 2001 and the June 2002 sampling, grain size was determined within all treatment levels and control for both the non-deteriorated and deteriorated marsh. Units are shown in mm.

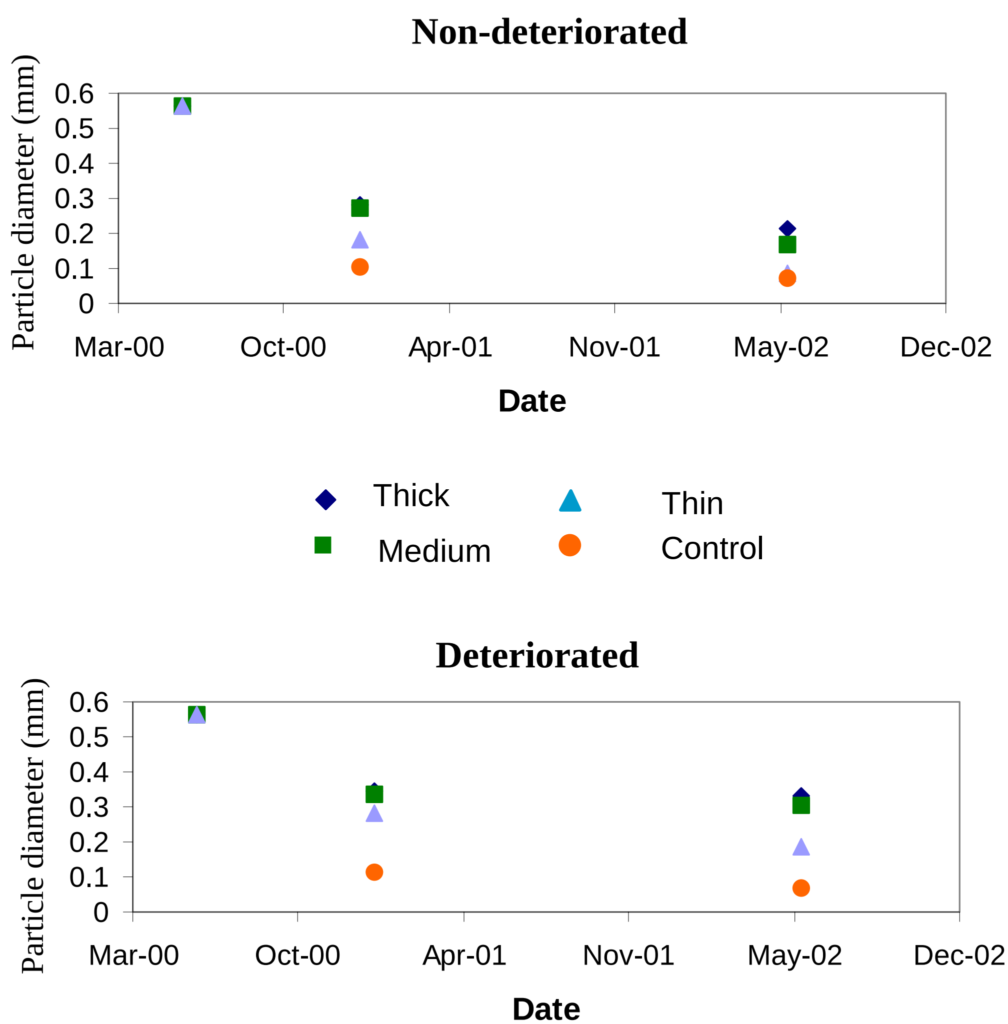


Figure 16. Surficial mean grain size for the deteriorated and non-deteriorated sites.

Flow Hydrodynamics

Flow velocities were recorded during a flooding tide in each of the four-study sites and both controls in September 2002 (Figure 16). In addition, velocities were measured over a three hour period of falling water along transects normal to the adjacent creek in one deteriorated and one non-deteriorated site in October 2002 (Figure 18 and 19). The mean flow velocities were higher in the deteriorated treatment sites (4.05 cm s^{-1}) when compared to the non-deteriorated treatment sites (1.75 cm s^{-1}). Velocities in the deteriorated control (5.68 cm s^{-1}) were also higher than those measured in non-deteriorated control (1.79 cm s^{-1}) (Figure 17). During ebb flow, velocities in the deteriorated site showed heightened velocities when compared to the non-deteriorated site. This difference in velocity was consistent over the entire three hour measurement period, with the greatest difference occurring during the third hour when velocities in the deteriorated site were almost one order of magnitude greater then velocities in the non-deteriorated site (Figure 18).

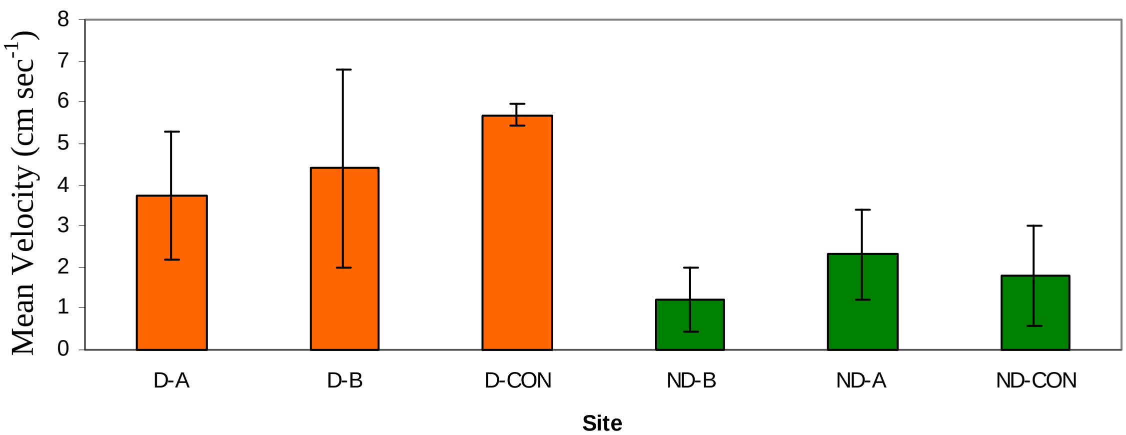


Figure 17. Mean flow velocities for the deteriorated and non-deteriorated sites. Each mean is calculated from three sampling bursts. Error bars indicate ± one standard deviation for the three sampling bursts. Deteriorated sites shown in orange and non-deteriorated sites shown in green.

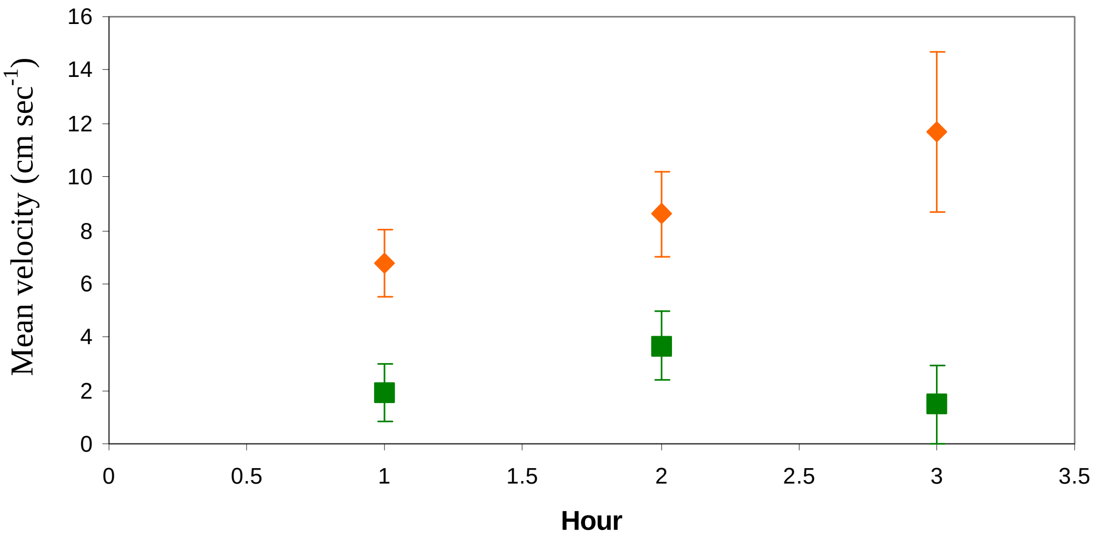


Figure 18. Mean flow velocities over a three hour ebb period for the deteriorated and non-deteriorated sites. Deteriorated sites are shown in orange and the non-deteriorated sites are shown in green. Each mean is calculated from multiple sampling bursts. Error bars indicate $\pm$ one standard deviation for the three sampling bursts.

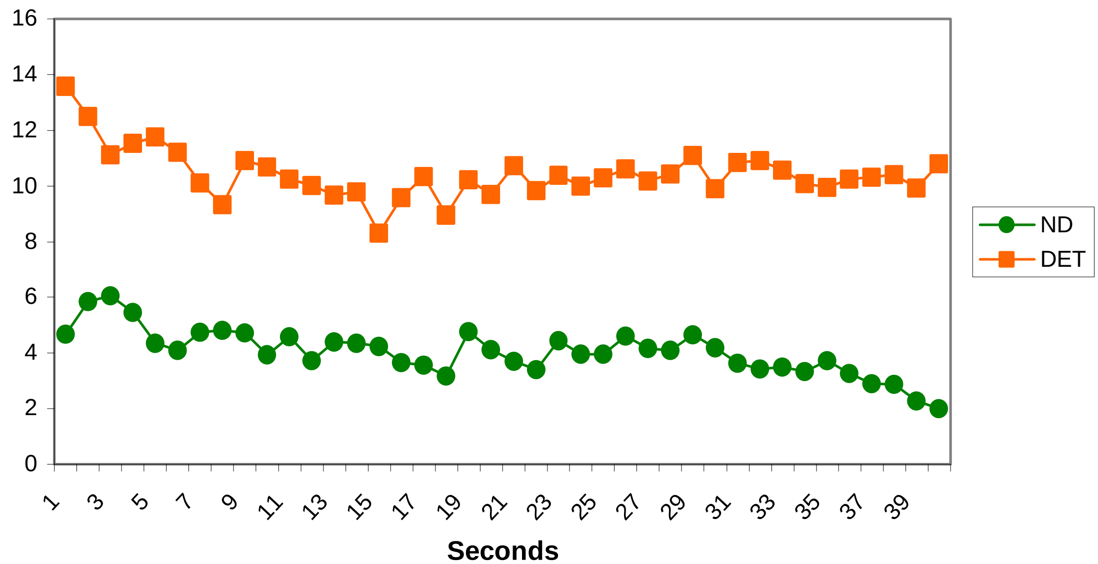


Figure 19. Third hour velocity burst data over a 40-second duration for transects entering a non-deteriorated and deteriorated site.

DISCUSSION

Vascular Plants and Marsh Chemistry

Sediment additions to deteriorated marsh resulted in more than a two-fold increase in *Spartina alterniflora* stem densities by the second summer. The observed rate of increase was essentially the same for areas receiving medium and thick treatments. Thin treatments also increased, but to a lesser extent. In the non-deteriorated sites that received sediment additions, an increase in stem density was also observed, but the effect of sediment placement was less profound than in the deteriorated sites. As discussed earlier, live plant densities in the treated deteriorated and non-deteriorated sites did not significantly differ during the second growing season. Thus, the placement of dredged sediment appears to have positively affected plant densities in the deteriorated sites without negatively impacting growth in the non-deteriorated sites.

The elevation and position of the deteriorating marshes relative to local drainages was probably the key element impeding plant growth and leading to the lower *Spartina alterniflora* stem densities in the deteriorated sites during the first growing season. Elevation surveys indicated that the deteriorated marshes were approximately 20 cm lower than the non-deteriorated marsh sites. It is my contention that the initial state of the vegetation (i.e., pre-sediment addition) in the deteriorated sites was likely associated with greater lengths of tidal inundation and water-logging effects that caused low redox potentials in the sediments. Sulfides and associated reduced compounds found in sediments with low eH values have been shown to be deleterious to *S. alterniflora* and other wetland plants (Howes et al., 1981). Sulfide toxicities are believed to slow nitrogen uptake and assimilation in plants, thus reducing plant growth in waterlogged environments (McKee and Mendelssohn; 1988). In addition, the toxicities can retard root development.

The position of each type of plot relative to local drainage may also have affected sediment geochemistry and vascular plant response. During this study, the non-deteriorated sites were located in stream-side positions whereas the deteriorated sites were located in the interior marsh. Prior to sediment placement, the non-deteriorated sites were characterized by the taller form of *S. alterniflora* and by more positive soil eH values. In contrast, the deteriorated sites were vegetated by the shorter growth form and exhibited lower eH values. These observations are consisted with DeLaune et al. (1983) and Howes et al. (1981) who have documented changes in plant productivity and height in response to marsh drainage. DeLaune et al. (1983) observed higher redox potentials in the surface sediments of a more productive streamside marsh compared to the adjoining interior marsh, and attributed this difference in productivity to elevation differences that allowed for greater drainage of the streamside site. Howes et al. (1981) found that sub-surface sediment conditions are more oxidized below tall forms of *S. alterniflora* and more reduced under the short form.

The data collected during this study indicate that sediment additions, to both non-deteriorated and deteriorated marshes, resulted in a shift of the eH profile to more positive values. Furthermore, the highest (i.e., most positive) redox potentials were observed in the thick and medium additions; areas that also exhibited the most improvement in *S. alterniflora* growth. The addition of the dredged material appears to have lessened the waterlogging problem, in turn creating a more oxidized environment, and reducing the potential for sulfide toxicity. An increase in eH values was also seen during the second growing season when compared to the first. The broad application of these results are limited because eH values in control sites also increased by the second growing season. Measurement error may account for the observed difference between the two growing seasons, because the increase occurred in both control and treatment sites. Nonetheless, sediment addition did result in more oxygenated mean profiles for deteriorated and non-deteriorated sites.

While mean canopy height appears to have been minimally affected by sediment placement, there was one rather unexpected response in the treated deteriorated sites. By the second growing season, mean plant height in the deteriorated sites had decreased relative to the first growing season. As plant heights in the non-deteriorated sites were not different between growing seasons, the decrease in mean plant height in the treated deteriorated sites was most likely due to new plant growth as opposed to a loss of old (i.e., taller) growth. Significant new growth of smaller plants (as evidenced by the increased stem densities in the amended deteriorated sites) would result in an overall lessening of the mean plant height for the entire plot. The growth form within these deteriorated sites, however, may never achieve heights associated with the *S. alterniflora* tall form because these sites are located in the marsh interior and still subject to poor drainage. Post-addition elevation surveys indicate that the deteriorated marsh surface was approximately 10 cm lower than the surface of the non-deteriorated control area, which received no sediment. If an additional 10 cm of material were added to the deteriorated sites to increase their elevation to that of the non-deteriorated control, canopy height may increase to levels equal to the non-deteriorated sites. This approach seems feasible because the addition of 10 cm (i.e., thick treatment) of material to the non-deteriorated sites at the beginning of the project did not adversely affect stem density or plant height.

Sedimentation Processes

The placement of dredged material on the surface of deteriorating marshes during this study appears to have resulted in increased sediment stability. Increased sediment retention on the marsh (or decreased re-mobilization) likely resulted from a combination of three factors: 1) increased plant growth and stabilization of substrate by roots, 2) increased flow baffling associated with increased stem densities, and 3) a net increase in mean grain size of surface sediments as a direct consequence of sediment addition.

The concentration of and proximity to source material, vegetative baffling ability, flow velocity, tidal inundation, tidal range, and marsh elevation can all affect sediment deposition in tidal marsh environments. Previous studies have shown significant correlations between length of inundation, elevation, and sediment accumulation on the marsh surface (Leonard, 1997; Cahoon and Reed, 1995; French et al., 1995). In general, sediment deposition tends to be higher in areas lower in elevation and subject to prolonged hydroperiod (i.e., inundation). These observations are consistent with the higher rates of deposition measured in deteriorated sites during this study. The lower elevation of the deteriorated sites (approximately 20 cm lower) relative to the non-deteriorated sites, however, is not likely to account for the observed 5-fold difference in deposition rate. Hydroperiods within the deteriorated sites were flooded approximately 1.5 to 2 hours longer than the non-deteriorated sites. It is highly unlikely that this difference in flooding between the non-deteriorated and deteriorated areas would result in significantly more deposition in lower elevations, because TSS values have been shown to range from 10 mg l^{-1} to 42 mg l^{-1} in nearby tidal creeks (Leonard, 1997). Further, if the difference in deposition were related solely to elevation variations, the deteriorated control area also should have yielded a high sedimentation rate, but this was not evident in the data.

Instead, it is my contention that the higher sedimentation rates measured by the sediment traps reflect local remobilization (i.e. resuspension) of the soupy waterlogged sediments in the deteriorated sites rather than trapping of new material. Dry bulk densities of non-amended sites were 0.71 grams cm^{-3} and 0.55 grams cm^{-3} for the deteriorated and non-deteriorated sites, respectively. The lower bulk density in the non-amended deteriorated area may reflect a smaller contribution of subsurface organic matter (due to fewer plants) compared to the non-amended, non-deteriorated area. This explanation is consistent with Mitsch and Gosselink (1993) who associated lower bulk densities with organic rich soils and higher bulk densities with more mineral soils. The low plant densities observed in the deteriorated site during summer 1 and the relatively poor soil conditions further indicate that insufficient below ground biomass (i.e., roots and rhizomes) existed to adequately stabilize the sediments. Inadequate below ground stabilization coupled with higher over-marsh flow velocities (due to reduced plant baffling) could result in greater remobilization of an unstable substrate. Leonard et al. (1995) found that increased flow velocities were associated with lower stem densities, thus providing a potential mechanism for sediment mobilization in deteriorating marshes.

Data collected during this study indicate that flow velocities were greater in the deteriorated sites than in the non-deteriorated sites, thus making conditions more favorable for sediment resuspension. At some points in the flooding cycle, velocities in the deteriorated sites were almost one order of magnitude greater than in the non-deteriorated sites. Moreover, these velocities were recorded after significant plant growth had occurred in the deteriorated sites. Thus, it is likely that strong velocities capable of remobilizing sediments existed in the deteriorated sites in summer 1. Postma (1967) concluded that velocities of 20 to 40 cm sec^{-1} may resuspend material between 0.1 to 0.5 mm in diameter; grain sizes consistent with surface sediments in the study area. Damage to the canopy associated with sampling efforts is another factor contributing to the higher velocities observed in the deteriorated sites. Trampling of vegetation was more pronounced in areas leading to the deteriorated sites and these sites, already subject to environmental stress, would have been slower to recover than the non-deteriorated sites.

Following sediment additions, deposition rates in the deteriorated sites decreased as plant cover increased. In the non-deteriorated sites, deposition remained constant between summer 1 and summer 2. Deposition rates in the control sites also remained constant between the two growing seasons. Although measured deposition rates in the deteriorated site at the end of the study were still significantly higher than those measured in the non-deteriorated sites, the difference between the two was less. Thus, the results of this study suggest that reduced flow speeds and increased plant coverage coupled with the addition of coarser material on the surface had a stabilizing effect on the marsh substrate.

Effects of sediment placement were also evident in the percent organic content of sediment trap data. The amount of fill-material added to sites was inversely related to the organic content of material retained on sediment traps. In general, sediment traps deployed in areas receiving the most fill material retained less organics material than traps deployed in other areas. When the percent organic content on traps was compiled for all sites including controls, the highest organic percentages occurred in control areas, and decreased with treatment thickness. This trend was also seen in the organic content data for surface sediments, which showed the highest percent organic content in the thin and control areas and the lowest percent organics in the medium and thick levels.

The organic content of accreted material in tidal marsh systems is typically a combination of plant detritus, the remains of microbes, phytoplankton and animals, fecal pellets, and decomposing root material (Frey and Basan, 1985). The low organic content of deposited sediments and high bulk densities of the deteriorating sites confirm that these sites were not functioning well. Conversely, the higher organic content in the non-deteriorated sites are consistent with a healthier system. Over the duration of the study, organic content of materials retained in the deteriorated sites increased, thus, corroborating other data indicating that sediment placement improved overall marsh conditions. Further, because the percent organic content of retained material in the amended non-deteriorated sites did not change, it can be argued that the benefits of additions to the deteriorated sites did not come at the expense of the non-deteriorated sites.

Grain Size and Soil Bulk Density

A five-fold increase in grain diameter was observed between pre-fill marsh conditions (0.11 mm) and the applied dredge fill material (0.56 mm). This increase in grain diameter was uniform throughout both deteriorated and non-deteriorated marsh sites. Size measurements taken in January 2001 and June 2002 showed decreasing grain diameters approaching pre-fill conditions within all treatments in both non-deteriorated and deteriorated sites. By the end of the study, the mean grain size was smallest within the thin treatment, followed by the medium, and thick for both types of marsh. These results suggest that the fill material was actually incorporated into the marsh substrate over the duration of the study.

These results also suggest, however, that the return to pre-addition grain sizes was more rapid the non-deteriorated sites as opposed to the deteriorated. Leonard et al. (1995) documented that increased stem densities can sufficiently lower flow speeds and turbulence within a marsh when compared to areas of lower densities. These higher grass densities allow for a greater fraction of fine grain material to settle from the water column. Within the healthy sites, higher stem densities documented throughout the first part of the study coupled with lower water velocities would allow for greater settling of fine particles. Conversely, lower stem densities in the deteriorated regions especially during the first growing and presumably higher water velocities may have impeded the quantity of fine particle deposition. Further, deposition under these conditions may have favored coarser-particles, which settle at higher water velocities. Thus, return to pre-fill condition was delayed in deteriorated sites until the deteriorated vegetation increased to levels sufficient to baffle currents and lower velocities.

Bulk densities were also different between the deteriorated and non-deteriorated treated and control sites. According to Mitsch and Gosselink (1993), sediments and soils that are high in organic matter typically have lower bulk densities, as opposed to inorganic rich sediments, which have higher bulk densities. In my study sites, bulk densities were lowest in the control areas, followed by thin, medium and thick treatments, respectively. This observation is consistent with Mitsch and Gosselink (1993) considering the effects of sediment additions; sites that received the highest amount of inorganic sediment, yielded the highest bulk densities. Densities were also higher within the deteriorated sites when compared to the non-deteriorated ones. As explained previously, this result is likely associated with reduced below ground biomass in the deteriorated sites and flow conditions unconducive to the trapping of fine particles.

CONCLUSIONS

The results of this study indicate that the application of thin-layers of sediment can be used to successfully alleviate common problems facing deteriorated marsh systems suffering from land submergence. The data also indicate that non-deteriorated areas experienced little to no negative consequences when subjected to similar sediment applications. Thus, the following conclusions can be made about the impact of thin layer sediment amendments on vascular plants, subsurface oxygen levels, inorganic and organic particle deposition, and sediment characteristics:

1. The addition of dredge material on the surface of deteriorating marshes led to a two-fold increase in vascular plant stem densities but had little effect on the overall height of *Spartina alterniflora*. An increase in stem density was observed for all deteriorated treatment levels with the greatest affect occurring in the thickest treatments. In addition, stem densities became increasingly uniform among the deteriorated and non-deteriorated

sites over the duration of the experiment. Sediment placement in non-deteriorated sites resulted in little to no impact on stem density and plant height.

2. Sediment additions resulted in higher eH values (e.g. higher oxygen levels) in both deteriorated and non-deteriorated marshes. Further, the highest eH values were associated with areas that received the thickest sediment additions. Substrates that received no sediment amendments or thin amounts of fill-material remained more reduced (e.g. lower eH values) throughout the experiment. Over the duration of the study, the non-deteriorated sites exhibited a more oxygenated subsurface when compared to the deteriorated sites. It remains likely that surface elevation differences and waterlogging effects were the main contributing factors influencing differences between redox potentials of the non-deteriorated and deteriorated sites. Nonetheless, the increase in subsurface oxygen levels observed in amended areas likely contributed to the increased vegetative canopy in the deteriorated sites.

3. Sediment deposition in the deteriorated sites exceeded deposition in the non-deteriorated sites, although, the range in deposition has decreased over the duration of the project. The higher sedimentation rates measured by the sediment traps most likely reflect local remobilization of the soupy sediments in the deteriorated sites. Velocities in the amended deteriorated sites were lower than velocities in the controls, but still higher than in the non-deteriorated sites. However, the latter were more densely vegetated originally and remained so over the duration of the study. The net effect appears to be that the soupy surface sediment in the deteriorated sites were stabilized by a combination

of three factors: 1) increased plant growth and stabilization of substrate by roots, 2) increased flow baffling associated with increased stem densities, and 3) a net increase in mean grain size of surface sediments as a direct consequence of sediment addition.

4. The organic content of deposited sediments was originally lower in the deteriorated sites when compared to the non-deteriorated sites. These data indicate that the deteriorated sites were not functioning well, as the organic matter typically found in healthy sites. Over the duration of the study, however, organic content of materials retained in the deteriorated sites increased. This result corroborates other data suggesting that sediment placement improved overall marsh conditions. Further, because the percent organic content of material retained in the amended non-deteriorated sites did not change, it can be argued that the benefits of additions to the deteriorated sites did not come at the expense of the non-deteriorated sites.

5. A five-fold increase in grain diameter was observed between pre-fill marsh conditions and the applied dredge fill material. Grain size measurements taken January 2001 and June 2002 showed decreasing grain diameters approaching pre-fill conditions within all treatments in both non-deteriorated and deteriorated sites. By the end of the study, the mean grain size was smallest within the thin treatment, followed by the medium, and thick for both types of marsh. These results suggest that the fill material was being reworked and assimilated into the marsh substrate over the duration of the study. These results also suggest, however, that the return to pre-addition grain sizes was more rapid in the non-deteriorated sites as opposed to the deteriorated. It is likely that higher stem

densities coupled with lower water velocities in the non-deteriorated sites during the first growing season allowed for a greater fraction of fine grain material to settle from the water column.

Overall, this study has showed that the application of thin-layers of sediment can be used to offset common problems affecting deteriorated marsh systems, while having little to no negative consequences on non-deteriorated marshes. Thus, indicating if applied in appropriate thickness, that high-pressure spray dredging (Jet-Spray) technology could be a viable alternative to current dredge disposal methods. The Jet-Spray technology is more beneficial than current dredge practices by: 1) eliminating the need for spoil island, which directly convert marsh habitat into terrestrial habitat; and 2) acting as an amendment to marsh systems subject to land submergence problems. Further, government agencies could use this technology as a best management practice to enhance wetland environments as opposed to current disposal practices that typically destroy wetlands. However, prior to management implementation more research is needed. Large-scale investigations are needed to determine impacts over vast areas of deteriorated and non-deteriorated marsh surfaces. In addition, an amendment thickness should be determined for placed sediment, in which negative marsh impacts may occur. From this an optimal thickness of sediment can be determined that is environmentally and economically feasible. Finally, the economical feasibility for thin layer dredge disposal should be found. This should be done to compare the price tag of this technology with the cost of existing practices to determine if government agencies would view the Jet-Spray technology as cost efficient.