Sediment Recycling: Marsh Renourishment Through Dredged Material Disposal

A Final Report Submitted to

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Submitted by

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Abstract

This goal of this project was to determine if the placement of dredged material can be used to offset elevation losses in deteriorating marshes without decreasing productivity and/or diminishing functionality in adjacent non-degraded areas. Approximately 8 m^3 of dredged material was taken from dredged material disposal banks adjacent to the Atlantic Intracoastal Waterway and manually placed in deteriorated and non-deteriorated marsh plots behind Masonboro Island, NC (North Carolina National Estuarine Reserve). The material was distributed in wedges ranging from 10 cm in thickness to 0 cm in each of four study plots; 2 in deteriorated marsh and 2 in non-deteriorated marsh. Additional control plots were also established in deteriorated and non-deteriorated areas. The response of vascular plants, benthic microalgae (BMA), benthic infauna, and sediment redox potential to sediment additions were monitored between May 2000 and October 2001. Short-term sediment deposition rates, surficial flow attributes, and changes in sediment composition and granulometry were also examined.

The results of this study suggest that the addition of dredged material on the surface of deteriorating marshes led to increases in vascular plant stem densities and increased microalgal biomass. While the total thickness of sediment added to each plot did not significantly affect stem densities or BMA, 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. Sediment additions had little to no impact on these parameters in non-deteriorating sites. Further, sediment additions resulted in higher eH values (e.g. higher oxygen levels) in both deteriorating and non-deteriorating marshes. The increase in stem densities in the deteriorated sites seems to have increased flow baffling in these environments and led to decreases in remobilization of surface sediments especially in the deteriorated sites. Benthic infaunal data suggest that while sediment placement may have had a short-term affect on community structure, that recovery occurred quickly following sediment addition. Further, these data indicate that over the long-term, sediment additions did not negatively affect benthic infaunal diversity or abundance.

Further experimentation is necessary to further constrain "tolerable" levels of sediment additions. Our data indicate that thicknesses of 2 to 10 cm is practical for management practices, especially if distributed in a manner that enhances edge effects and includes both "thick" and "thin" regions. Obviously the addition of too much sediment to the marsh surface could be deleterious, but this study has not been able to detect that threshold. The next logical step for resource managers to pursue is a large scale (several kms) application of dredged material that includes incremental sediment additions and multiple patches/plots with varying spacing between the plots to evaluate potential effects on recovery time, especially in areas where large areas (km) of degraded marsh threaten habitat complexity of overall function.

Keywords: Marsh restoration, dredged material, sediment, *Spartina alterniflora*, Masonboro Island, NC

Introduction

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 the relative mean sea level allowing vegetation to take hold. 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). Fine sediments and halophytic vegetation characterize these marsh systems, which are formed and maintained through a combination of physical and biological processes (Reed, 1990). Marshes have a complex zonation and structure of plants, animals, and microbes, all tolerant to the stresses of salinity variations, tidal fluctuations, and extreme daily and seasonal temperature changes that occur in these intertidal areas (Mitsch and Gosselink, 1993).

Marsh systems are critical natural resources that provide significant ecological, economical 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 upon coastal fisheries, which rely heavily on a healthy functioning marsh to provide habitat and nursery areas. It has been estimated more than 90 percent of the commercially important fish and shellfish of the southeast Atlantic and Gulf coasts are either estuarine or salt marsh dependant at some point in their lives (Mitsch and Gosselink, 1993). Socially, these areas are valued 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 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 (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 increased sea level 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 have 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 sediment budgets of marshes (Reed, 1989; Friedrichs and Perry, 2001). Nonetheless, in many storm-impacted systems accumulation still remains insufficient to maintain an elevation in equalibrium with sea level rise.

The continued existence of a 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 these 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 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 channeling of the Mississippi River and the transport of its sediments into deep offshore areas, thus starving these systems of their sediment supply, exacerbate the situation. Within a Louisiana marsh, Delaune et al (1983) estimated through ^{137}Cs dating and artificial maker horizons that accretion rates averaged 0.8 cm/yr, whereas coastal submergence was averaged to be 1.2 cm/yr. In addition, Bauman et al. (1984) found marsh accretion insufficient to keep pace with apparent sea level rise in Louisiana inland marsh. In the southeastern U.S. system, marsh loss due to submergence is also an issue. Stevenson et al. (1986) documented accretion rates ranging from 1.7 to 3.6 mm/yr within a Chesapeake Bay marsh, and concluded that these where not sufficient to keep pace with sea level rise (3.9 mm/yr) in that region. In other southeastern U.S. marshes the rate of sediment accretion is approximately 1.2 mm/yr, which is less then the average rate of sea-level rise for the region (1.9 mm/yr) (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 (Mendelssohn and McKee ; 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; Mendelssohn and McKee, 1988). Studies have long documented such effects in the coastal marshes of southern Louisiana, as well as in the marshes bordering the east coast and Chesapeake Bay, where loss is occurring at an alarmingly high rate (DeLaune et al., 1983; 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 plants ability to baffle flow and, consequently, 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 especially be devastating when coupled with present and projected increases in rates of 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 the 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 two 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 typically may form (Hackney and Cleary, 1987). Moreover, sand that does accumulate in these inlets is more often than not 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 deteriorating marsh is one remedy that may mitigate current submergence problems. The rational 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 are 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 the dredged waterways significantly higher then 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®[2]) 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. The technology would allow for the operator to accurately dispose of 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 habitat (Cahoon and Cowan, 1988).

Although thin-layer disposal may reduce environmental impacts in several habitats, and improve marsh function in other habitats, few studies have evaluated its effect on the marsh (Cahoon and Cowan, 1988; Wilber, 1992a; Wilber, 1992b; Ford et al., 1999). 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.

In previous sediment addition studies, sediment additions have ranged from a few millimeters to over 30 centimeters in thickness (Cahoon and Cowan, 1988; Wilber, 1992; Ford et al., 1999). DeLaune et al. (1990) documented 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 millimeters of dredged material was applied to the surface. Wilber's (1992a; 1992b) studies were conducted in healthy marsh systems, therefore, resulting in an undetectable change in biomass. Wilber (1992) 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 results of placing dredged material 10 cm thick were less clear (Wilber, 1992).

Sediment additions may also have a positive effect on the redox potential of the marsh soil. Some studies have shown that marsh surface elevation and biomass differences may affect oxygen levels within the sediments (Howes et al. 1981; DeLaune et al.1983; Mendelssohn and McKee , 1988). Mendelssohn and McKee (1988) demonstrated this by varying inundation depths on transplanted marsh communities. When the marsh was transplanted to a lower elevation the environment became more reduced due to increased hydroperiod. The environment became more oxygenated when the marsh was transplanted to a higher elevation with less (Mendelssohn and McKee, 1988). Similarly DeLaune et al. (1983) showed redox potentials to be higher in surface sediments of the more productive streamside vegetation when compared to the adjoining inland vegetation for a Louisiana *Spartina alterniflora* marsh. The difference appeared to be associated with the higher elevation of the streamside location as compared to the inland location. In an east coast *S. alterniflora* marsh, Howes et al. (1981) demonstrated 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* marsh's ability to oxygenate the sediment more efficiently then a short form. In most marshes, the tall form of *S. alterniflora* is found on creek levees that are also areas of higher elevations.

The addition of sediment to the surface of the marsh may affect other factors including benthic microalgal composition and biomass. Benthic microalgae (BMA) are important primary producers in estuarine systems (Cadee and Hegeman, 1977), contributing a substantial amount of primary production to these systems (Peterson, 1981). They are numerous and productive but they do not accumulate the highly visible biomass created by marsh grasses and sea grasses (Freeman, 1989). It has been shown that benthic microalgal biomass can be many times greater than water column biomass, accounting for >50% of the total system primary production in many coastal environments (Cahoon and Cooke 1992; Krom 1991; Sundbäck and Snoeijis 1991). Other studies have determined that benthic microalgal production may range from one-third to as much as 1.4 times that of marsh angiosperm production (Freeman, 1989). The availability of benthic microalgal biomass (BMB) to consumers may be enhanced by its spatial distribution (Freeman, 1989) which may be directly and indirectly affected by the placement of sediment on the surface of the marsh.

Mean sediment grain size, for example, has been cited as a possible determinant of BMA biomass (Freeman 1989; Kennett and Hargraves 1985; Amspoker and McIntire 1978; Davis and Lee, 1983). Numerous studies have shown that fine sand bottoms are typically associated with a higher amount of BMA biomass than areas composed primarily of coarse sand and/or high clay/silt sediment (Freeman 1989; Chester et al. 1983; Newell 1965). When the sediment cannot retain the water for sufficient lengths of time to strip the nutrients from the water column, the nutrients are lost from the system during the falling tide. Conversely, if water is retained in the sediment for too long, the system can become anoxic, standing biomass declines and the shedding of detrital particles may be reduced. Under these conditions, nutrient availability may decrease because less organic matter is available to be converted to organic and inorganic forms of nutrients; both of which may be taken up directly by planktonic and benthic autotrophs (Krom 1991; Sundback et al. 1991; Flint and Kamykowski 1984). This step is critical in marine systems as the decomposition and dissolution of accumulated organic material are the two major sources of dissolved nutrients both to and from marine sediments (Sundback and Snoeijis 1991; Lomstein et al. 1990; Zeitzschel 1979). Thus, changes in the sedimentologic attributes of the marsh substrate resulting from the placement of dredged material can affect phytoplankton abundance as the latter is regulated by the regeneration of sedimented nutrients, especially when water column concentrations have been depleted (Ragueneau et al. 1994; Conley and Malone 1992; Sigmon 1995). Further, increases to the surface elevation of the marsh resulting from sediment additions result in increased light flux to the marsh surface because the treated area is inundated for a shorter period of time. Increased light flux, especially when coupled with the availability of nutrients, could promote significant increases in benthic microalgal biomass.

Changes in microphytobenthos biomass and diversity are likely to exert secondary impacts on higher trophic levels, although untested with respect to engineered sediment additions (Posey et al., 1999). Benthic microalgae in the marsh systems have been shown to display taxonomic reorganization in response to sediment inputs associated with hurricane induced overwash (Hilterman, 1998). Changes in diatom composition, therefore, will impact those organisms and, ultimately higher organisms like juvenile fish and crustaceans that rely on benthic invertebrates as a food source. Negative impacts to the infaunal community could cascade upward and affect juvenile fish and crustaceans that rely on this habitat, thus altering the value of these areas as a nursery habitat, at least on the short-term. Conversely, if executed properly, sediment additions can indirectly enhance higher level consumers by benefiting primary producers such as benthic microalgae and vascular plants. Thus, 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.

Objectives

The goal of this study was to examine how the placement of dredged spoil material can be used to offset elevation losses in deteriorating marshes without decreasing productivity and/or diminishing functionality. The specific objectives of this study were:

1. To determine maximum sediment addition depths in tidal marshes that optimize elevation maintenance without compromising microphytobenthos and vascular plant biomass, or community structure of resident fauna (benthos);

2. To determine the effects of sediment placement on standing biomass, microalgal biomass, sedimentation rate, soil redox conditions in deteriorating and non-deteriorating *S. alterniflora* marsh sites;

3. To evaluate temporal changes in granulometry, standing biomass, geochemistry, sediment accumulation, and benthic infaunal community structure for treated and non-treated marsh areas;

4. To constrain optimal thickness of sediment placement that yield positive benefits for treated deteriorated sites without being deleterious to treated healthy sites; and

5. To disseminate project results to a range of potential users through multimedia and on-site experience.

Methods

Study Area and Experimental Design: 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 strands 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. The sediments in the study area consist mainly of sandy muds (approximately 50 percent fine sands and 50 percent muds). 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 this study, marsh areas were classified as non-deteriorated or deteriorated. Non-deteriorated marsh exhibited dense strands of *S. alterniflora* (> 350 stems/m^2) while deteriorated areas were characterized by sparse strands (fewer than 200 stems/m^2). Sediment deficits are probably the controlling factors leading to areas of sparse vegetation as evidenced by the generally more 'soupy' 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, although the fickleness and periodic nature of strong storms can make overwash a less reliable source of inorganic sediments in this system. For these reasons, the sediment-starved marshes behind Masonboro Island were ideal for this study.

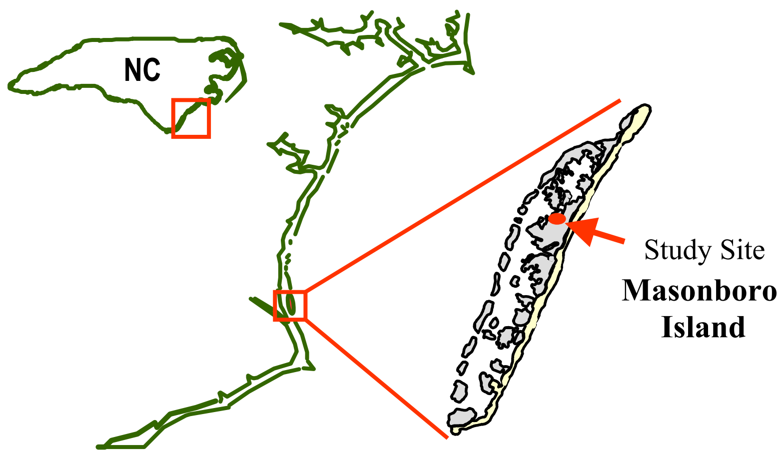

Figure 1. Masonboro Island

The experiment consisted of four treated sample plots each measuring 6.4 meters by 6.4 meters (two deteriorated (DET) and two non-deteriorated (ND)) and two non-treated (control) areas (one deteriorated (DET) and one non-deteriorated (ND)). The non-deteriorated sites were characterized by healthy stands (> 350 stems/m^2) of *S. alterniflora,* whereas the deteriorated sites were characterized by *S. alterniflora* die back (fewer than 200 stems/m^2). Preliminary elevation surveys taken showed a difference between non-deteriorated and deteriorated sites of approximately 23 cm, with the deteriorated sites being lower. During the initial phase of site preparation, approximately 8.2 m^3 of material was taken from a dredge spoil mound less than 1km from the proposed study site. Boardwalks were constructed over sample sites to help eliminate impacts of human disturbance during the study. Manual application was used to apply the sediment to all the study areas. 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.

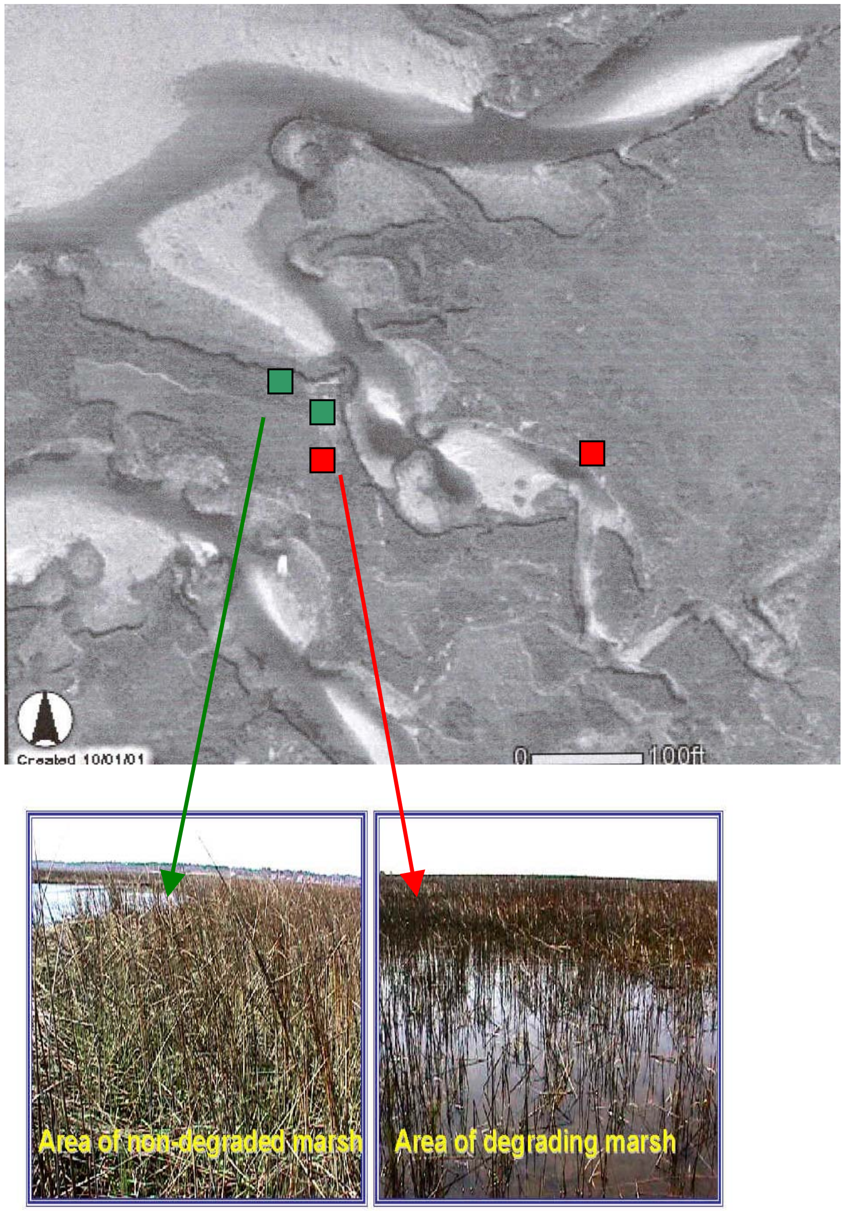

Figure 2. Aerial photograph of study site (non-deteriorated sites shown in green, and deteriorated sites shown in red). Insets show conditions of non-deteriorated and deteriorated sites prior to sediment additions.

Sediment application occurred during May 2000. 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 was mix fraction of medium to coarse-grained material. The fill material was transported from a small spoil 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 done to verify thickness of deposited material. For the purpose of the study, each plot wasdivided 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. The elevations were as follows: ND control (0 cm), ND site A (8.2 cm), ND site B (9.1 cm), DET control (-22.5 cm), DET site A (-13.4), and DET site B (-7.3 cm).

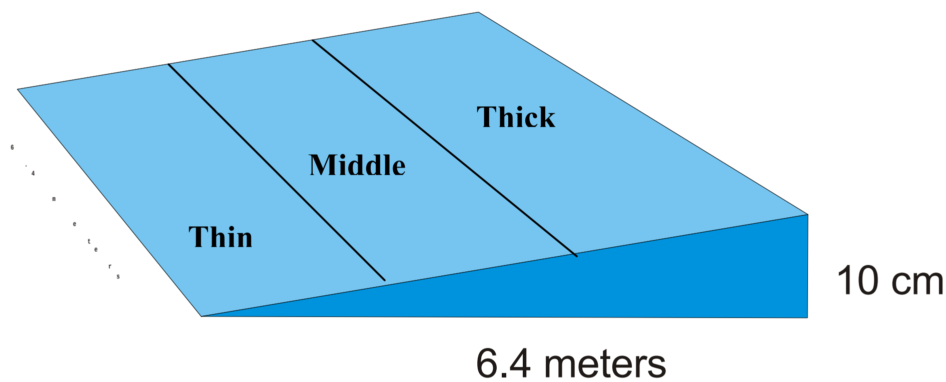

Figure 3. Diagram showing geometry of sediment fill.

Physical parameters: Tidal water levels were monitored 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 measure water level every 23 minutes. The water level recorders were programmed to measure tidal inundation every 23 minutes. Sediment characteristics, including dry bulk densities and organic content were monitored annually to determine any return to pre-addition conditions. Dry and wet sieving was used to determine the weight percent of material larger than 62.5 micrometer in diameter. The coarse fraction was then dry sieved at whole phi intervals. A LS 230 Beckman-Coulter particle sizer 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 then dried at 60^{o}C, weighed, and given as gram dry weight per unit volume. Surface samples were also combusted at 500 ^{o}C to determine the percent organics.

A SonTek hand-held acoustic velocity recorder was used to measure tidal inundation speeds within the 4 treated sites and 2 non-treated control areas. Velocity transects were also determined for a non-deteriorated and a deteriorated site. These measurements were included later in the experiment in hope of explaining some unexpected sedimentation results occurring within the sites.

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 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. In addition, an additional set of traps was deployed in each of the non-treated (control) areas. 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. Traps were in place for approximate 24 hour period, in which both deployment and retrieval occurred at low water. Upon retrieval, the traps were then oven dried at 60 ^{o}C and weighed to determine total accumulation in (mg cm^{-2} day^{-1}). Organic content was determined by combusting sediment traps at 450 ^{o}C for 4 hours.

Chemical parameters: Soil redox potential (eH) was examined using a portable voltmeter, platinum electrode, and reference electrode (Faulkner et al., 1989). 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 sediment thickness at each treated site and at both control sites between August 2000 and November 2001 (Table 1).

Benthic Microalgae: Sediment samples for BMA analysis were taken from all sites (control, deteriorated, and non-deteriorated) for both pre and post sediment addition. Samples for analysis were taken using a 20 mm (diameter) coring tube. Pre-sediment addition samples were taken once a month from January 2000 until the final addition of dredged sediment material in June 2000. A total of eight samples were taken at each of the four sites in which sediment was to be added. Both deteriorated sites and both non-deteriorated sites were analyzed together and classified as deteriorated control and non-deteriorated control.

Following the final sediment addition in June 2000, sediment samples were taken from every site the week immediately following sediment addition. For the next year (June 2000- July 2001) eight samples were taken within each of the individual sediment addition heights, within each of the four amended locations, on a monthly basis. Eight samples were also taken at each of the control sites during all sample collections. Samples were collected and taken back to the laboratory where they were analyzed for mean benthic chlorophyll a using the Whitney and Darley (1979) method.

Benthic Infauna: Replicate core samples were collected thick, medium, and thin sediment addition plots 6-8 weeks (July 2000) after sediment addition. This timing allowed for stabilization of the actual sediment additions, and was sufficient to allow initial responses by the plant community, and the infaunal community (i.e. organisms that were negatively impacted by the act of sediment addition would not interfere with sampling results) to subside. Additionally we feel this time period was short enough so that immigration from adjacent unimpacted marsh areas was minimal. A second set of samples was collected 10 months after the initial samples (May 2001). This sampling period follows a peak spring recruitment period for many infaunal taxa and allowed us to evaluate recovery of these areas.

All infaunal core samples (using a standard benthic core 10 cm diameter X 15 cm deep) were collected 1 m within each plot to reduce potential edge effects. Core samples were preserved in 10% buffered formalin with rose Bengal dye added to stain the organisms. The samples were later transferred to 50% isopropanol and sieved through a 500 micron screen. All organisms retained were removed from the sediment and identified to the lowest possible taxonomic level (species in most cases).

Vascular plants: Stem density, plant height and numbers of live versus dead shoots were examined for vascular plant response to sediment addition. Preliminary measurements were obtained prior to application, and subsequent measurements were compiled bi-monthly between Jun. 2000 to Oct. 2001 (Table 1). Five replicate 10 x 10 cm quadrats were used to measure plant biomass for each treatment thickness within the 4 study sites and the 2 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. The experimental sampling components and collection dates are shown in Table 1.

Table 1. Sampling schedule for vascular plants, soil chemistry, sediment deposition, grain size, and hydroperiod.

	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

Statistical Analyses: Paired t-tests were first conducted on eH, plant, and sedimentation data to determine if Non-deteriorated (ND) sites A and B were similar, and Deteriorated (DET) sites A and B were similar. This was done to avoid any psuedo-replication problems with the ND sites or the DET sites. A one, two, and/or three way Analysis of Variance was then used to determine where statistical difference occurred between the ND treated, DET treated, ND control, and the DET control. The independent variables used for the ANOVAs were marsh type (ND and DET), treatment thickness (high, medium, low, and control), and season (summer 2000, winter 2001, and summer 2001). Three way ANOVAs were conducted first to determine if any interaction was occurring among main factors. If significant interactions were seen, tests were broken down to two and then one way Analysis of Variance until no significant interactions were determined. A post hoc LSD test was also performed if the Analysis of Variance tests showed significant interaction between 2 or more variables. Benthic microalgal results were statistically analyzed using a two way analysis of variance for pre-sediment addition results and an a posteriori Tukey-Kramer analysis for post sediment addition. Benthic infaunal response to sediment addition treatments and pre-addition marsh condition were analyze with ANOVA on log-transformed abundances. Analyses were conducted only on dominant fauna and/or faunal groups.

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 3 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 - Oct 2000) were 149 stems m^{-2} and 256 stems m^{-2} in the deteriorated and non-deteriorated sites, respectively. 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 ($p<0.0001$) lower in the deteriorated sites as compared to the non-deteriorated sites. When mean stem densities are compared between 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.

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 first winter (November 2000-April 2001), mean stem densities for sites that received sediment additions were significantly ($p = 0.0074$) higher in the non-deteriorated (212 stems m^{-2}) sites than 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. 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.

During the second growing season (May 2001 – Oct 2001), stem densities in the control sites were 227 stems m^{-2} for the non-deteriorated and 134 stems m^{-2} for the deteriorated site. These means were not significantly different from the mean stem densities reported for the first growing season; $p = 0.7495$ and $p = 0.8231$ for deteriorated and non-deteriorated controls, respectively. Mean stem densities increased for all sites that received sediment additions. The increase was most profound in the deteriorated plots (Figure 4). Seasonal density whisker 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 $p = 0.0005$ and $p<0.0001$, respectively. 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} (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 ($p = 0.2420$). Further, all of the deteriorated plots that received sediment additions showed significant increases in stem density (Figure 6) while stem densities in the control sites did not significantly change ($p = 0.7495$).

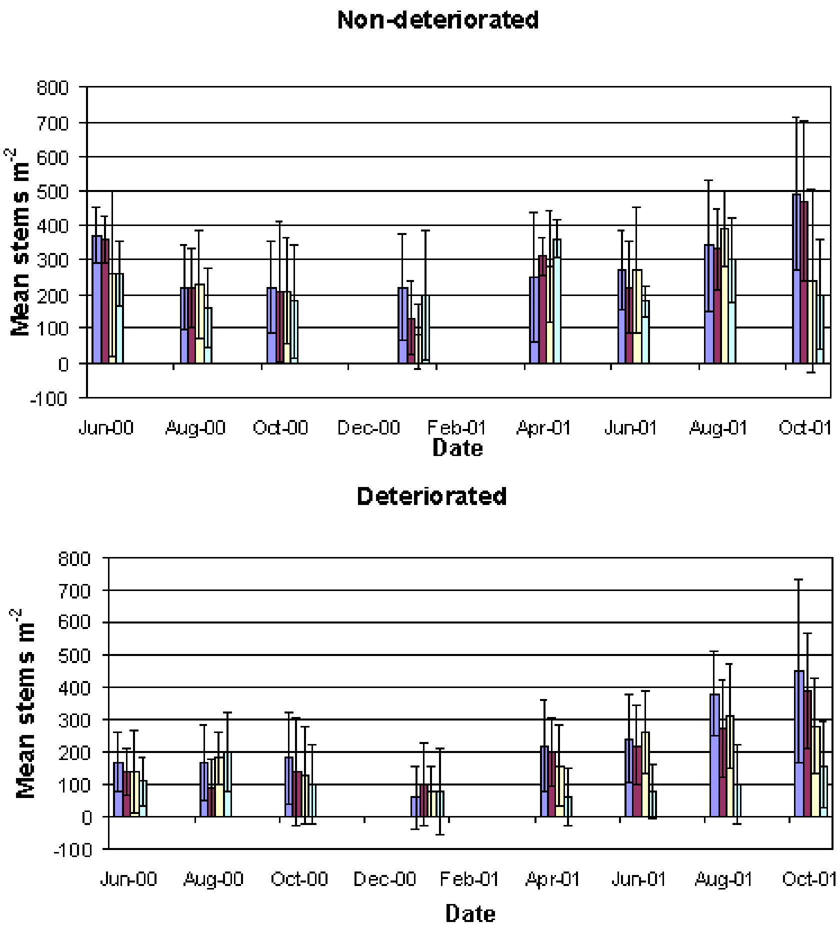

Figure 4. Mean stem densities of Non-deteriorated and Deteriorated sites. Thick treatments are shown in blue, medium treatments in red, thin treatments in yellow, and controls in aqua. Error bars indicate ± one standard deviation.

When mean stem densities for individual non-deteriorated and deteriorated treatment levels (i.e. thick, middle, thin) were compared, all cases were insignificant with the exception of the non-deteriorated-thick treatment which had significantly more *Spartina* shoots than the deteriorated-thin ($p = 0.0229$). Nonetheless, 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).

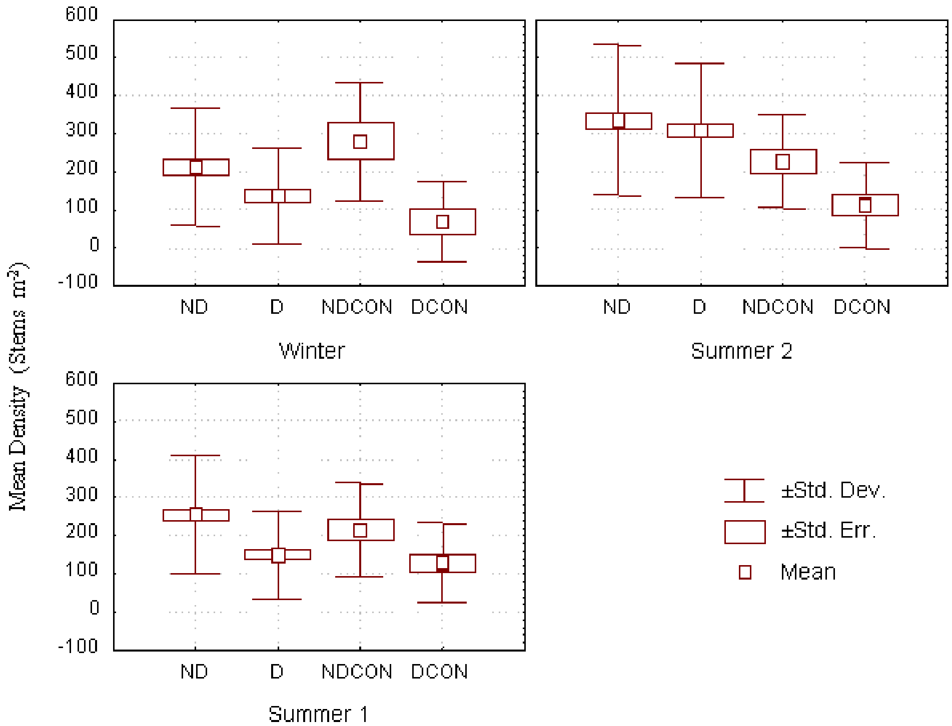

Figure 5. Box plots showing mean seasonal stem densities for non-deteriorated, deteriorated, and control sites.

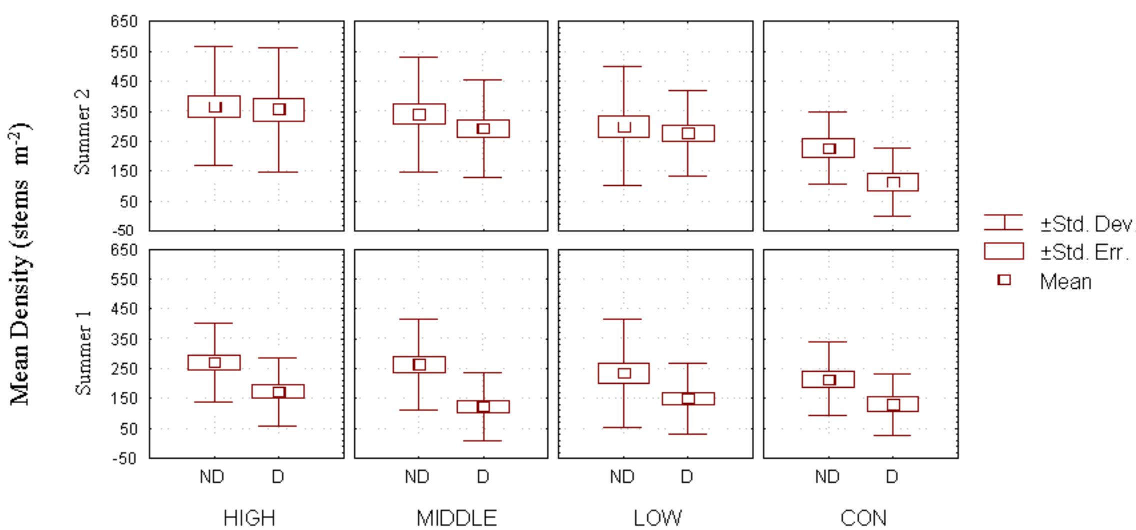

Figure 6. Whisker box plots of mean stem densities by treatment level in Summer 1 and Summer 2.

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 and Table 1). Paired LSD tests indicate significantly greater plant heights for both summers in the non-deteriorated sites than 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 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 ($p = 0.02$) in mean height between the first (49.8 cm) and second (28.1 cm) growing seasons. For both summer seasons, non-deteriorated and deteriorated control mean plant heights were significantly different from one another, $p = 0.01$ and $p = 0.0001$, for Summer 1 and Summer 2, respectively.

Table 2. Seasonally mean plant heights in treated and control areas for both deteriorated and non-deteriorated marsh types.

Site	Season	Mean (cm)	Control (cm)
ND	Summer 1	69	72.7
ND	Winter	46.3	46.9
ND	Summer 2	66	69.7
DET	Summer 1	45.6	49.8
DET	Winter	23.5	15.3
DET	Summer 2	49.5	28.1

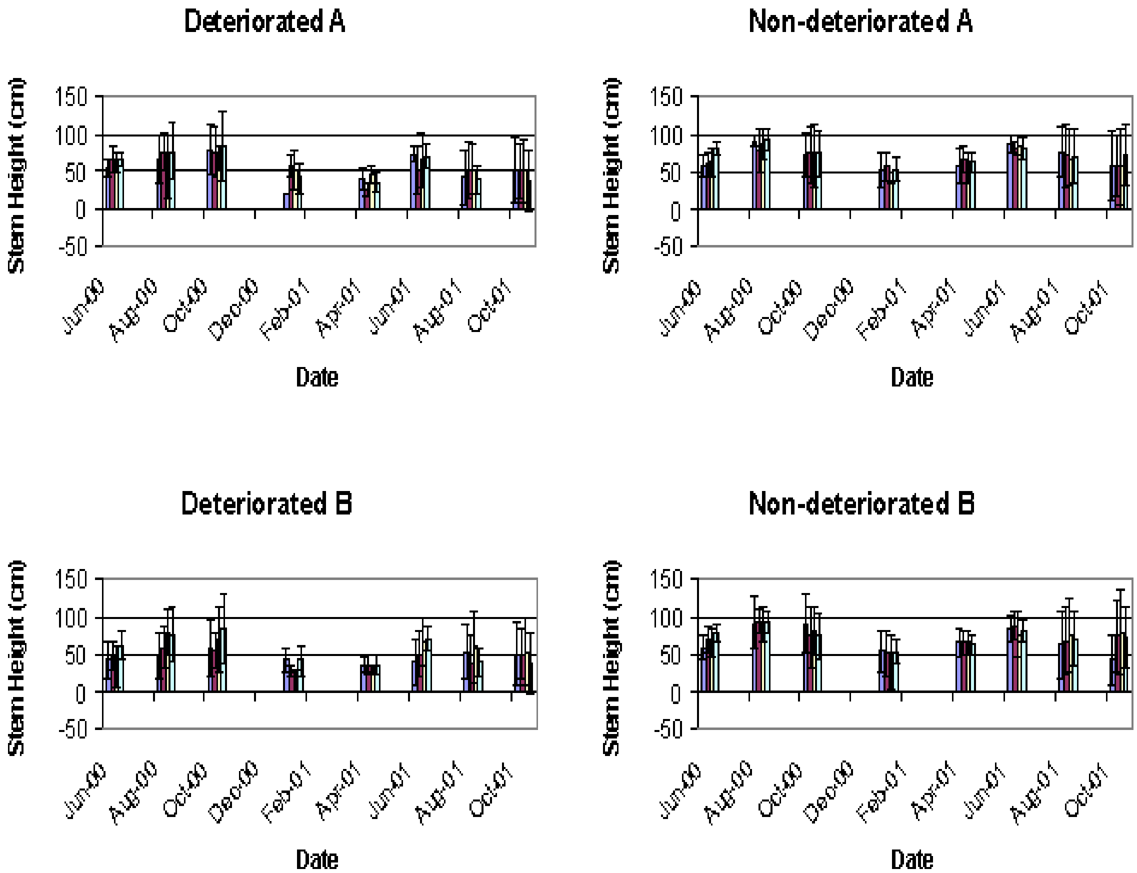

Figure 7. Monthly mean plant heights for various treatments in deteriorated and non-deteriorated marsh types. Thick treatments are shown in blue, medium treatments in red, thin treatments in yellow, and controls in aqua. Error bars indicate $\pm$ one standard deviation.

Chemical Parameters: The redox potential of the marsh sediments was recorded bi-monthly from August 2000 – November 2001. Eh profiles were generated for each treatment (i.e. thick, middle, and thin) in each of the 4 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 ($p<0.0001$). 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 sediments 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 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 ($p < 0.0001$) among the treatments in both the deteriorated and non-deteriorated 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). A two-way ANOVA of mean eH showed no clear interaction between the effects of treatment and season ($p = 0.1479$).

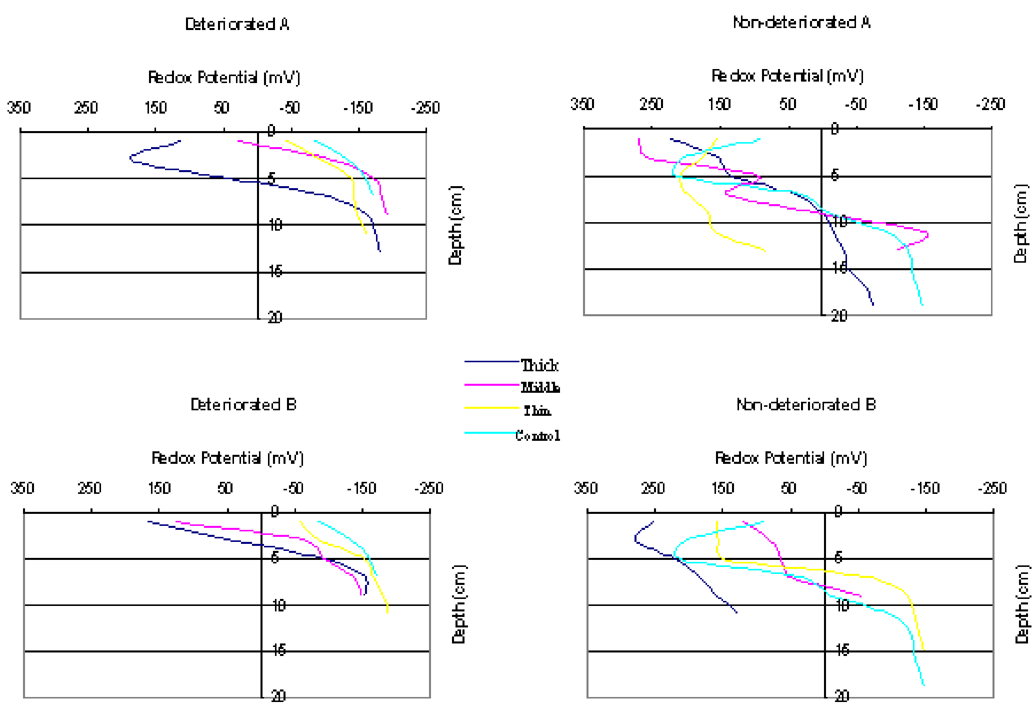

Figure 8. Averaged eH profiles for each study site (Aug 2000 – Nov 2001).

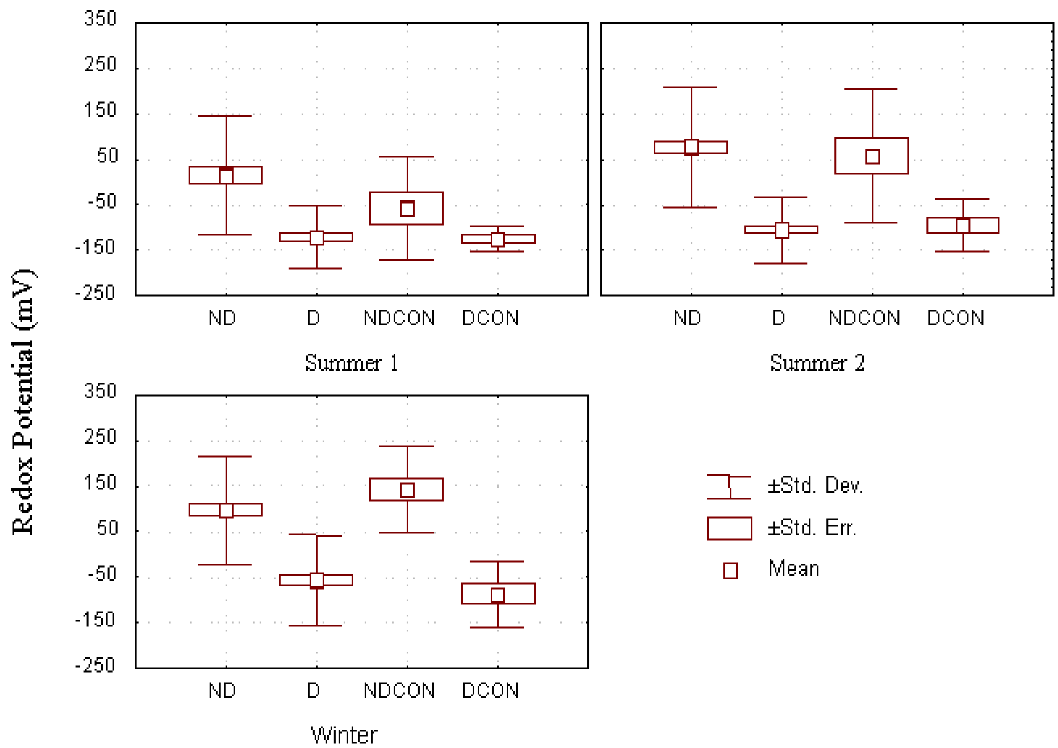

Figure 9. Seasonal box plots for treated (averaged) and control non-deteriorated and deteriorated sites.

Additional eH analyses were made using eH levels measured only within the introduced sand wedge layer and comparing them 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.

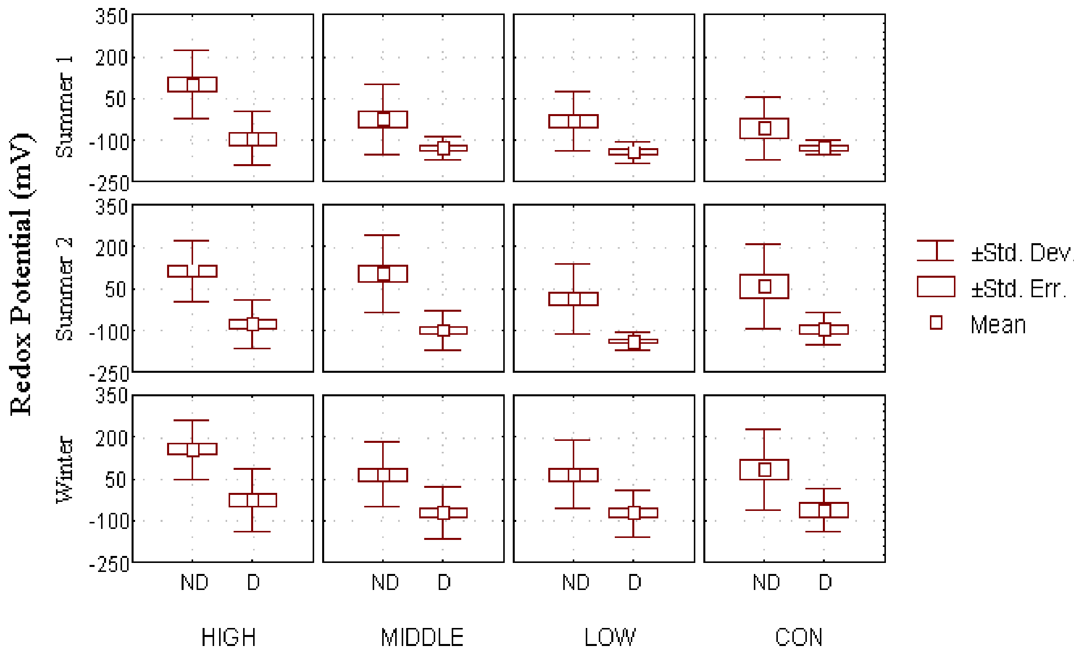

Figure 10. Box plots – Treatment versus seasonality versus marsh type.

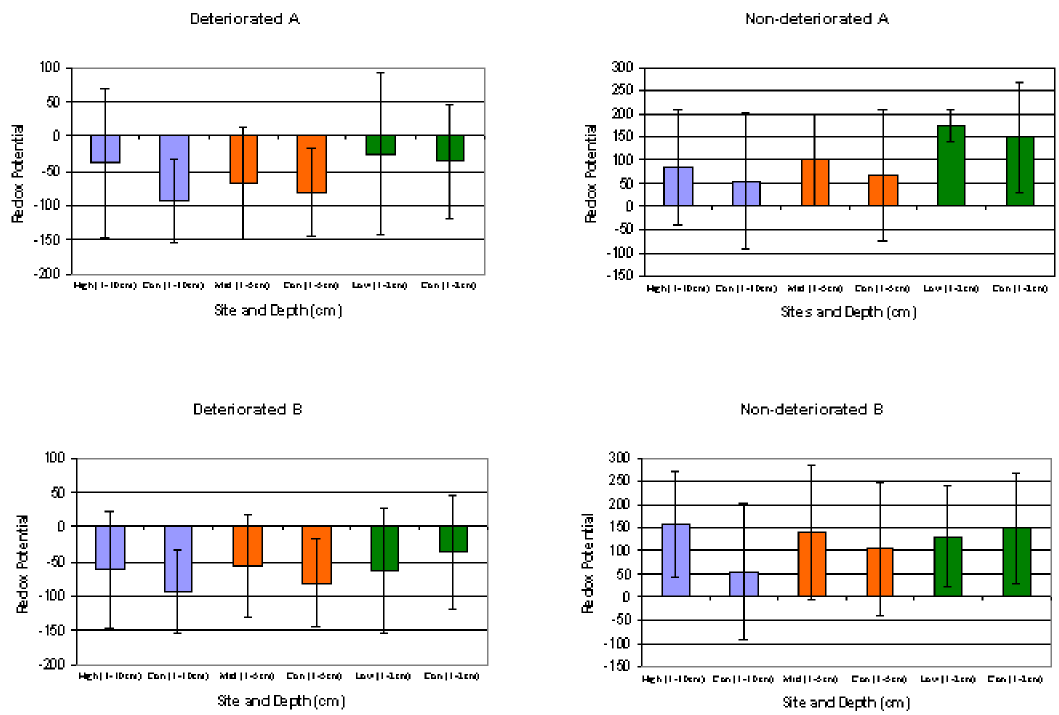

Figure 11. Mean depth-integrated Redox potentials measured within the sediment addition layer and an equivalent depth in the associated control. Error bars indicate ± one standard deviation of the mean. A high degree of seasonal variation and the decreasing exponentially shape of the profiles account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

Physical parameters: Particle deposition was monitored quarterly from October 2000 through September 2001 (Table 1). 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). A three-way ANOVA indicated significant interaction between marsh type (i.e., deteriorated or non-deteriorated), treatment (i.e., sediment thickness), and season ($p = 0.0204$). Total deposition 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 in the control areas did not significantly differ between the first growing season and the second growing season in either control areas. Further, mean deposition rates in Summer 1 were not significantly different from mean deposition rates measured during Summer 2 (Figure 12) in the non-deteriorated sites that received sediment additions.

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, middle, 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, middle, and thin treatments, respectively. For both the deteriorated thick ($p = 0.0001$) and the thin ($p < 0.0001$) the difference between Summer 1 and Summer 2 means was significant. Although deposition rates in the treated deteriorated sites remained significantly lower than deposition rates in corresponding non-deteriorated rates (Figure 12) in Summer 2, the difference in mean deposition rate appears to have decreased between the two marsh types since sediment placement.

Grain size was determined for the dredge fill material (June 2000), and for surficial sediments of all the non-deteriorated and deteriorated treatment and controls (Jan 2001 and June 2002). The dredge fill consisted of medium to coarse sand with no fine sands or muds (mean diameter of 0.57 mm.) Deteriorated and non-deteriorated control grain-sizes from Jan 2001 were used as a proxy for pre-fill conditions, because pre-addition grain sizes for these areas were unavailable. These materials consisted of 50 percent medium to fine sands and 50 percent muds, (mean diameter of 0.11 mm) for both marsh types. Mean grain size of surface samples has decreased over the duration of the study in all treatments for both the deteriorated and non-deteriorated sites (Figure 13). These data suggest that the surface sediments are returning to pre-fill conditions over time but that the return to pre-fill conditions is occurring most rapidly in the non-deteriorated sites (Figure 13).

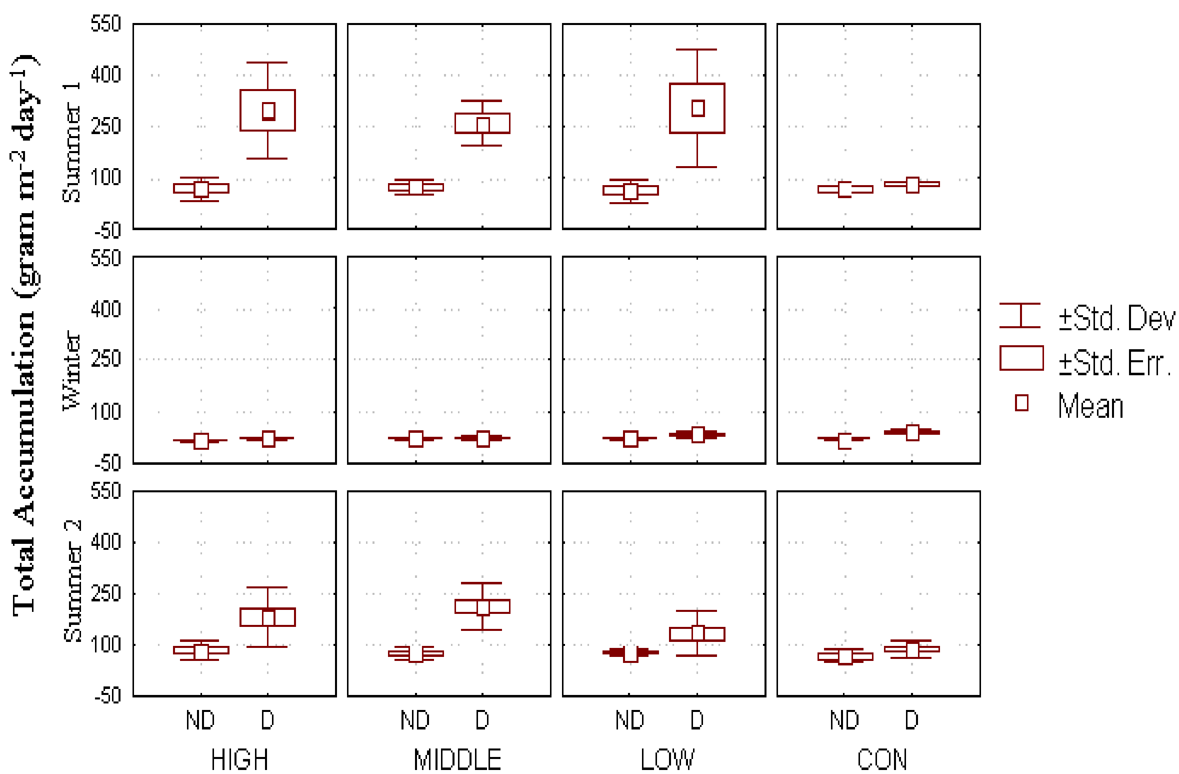

Figure 12. Surficial sediment deposition measured in deteriorated (D) and non-deteriorated (ND) experimental marsh sites.

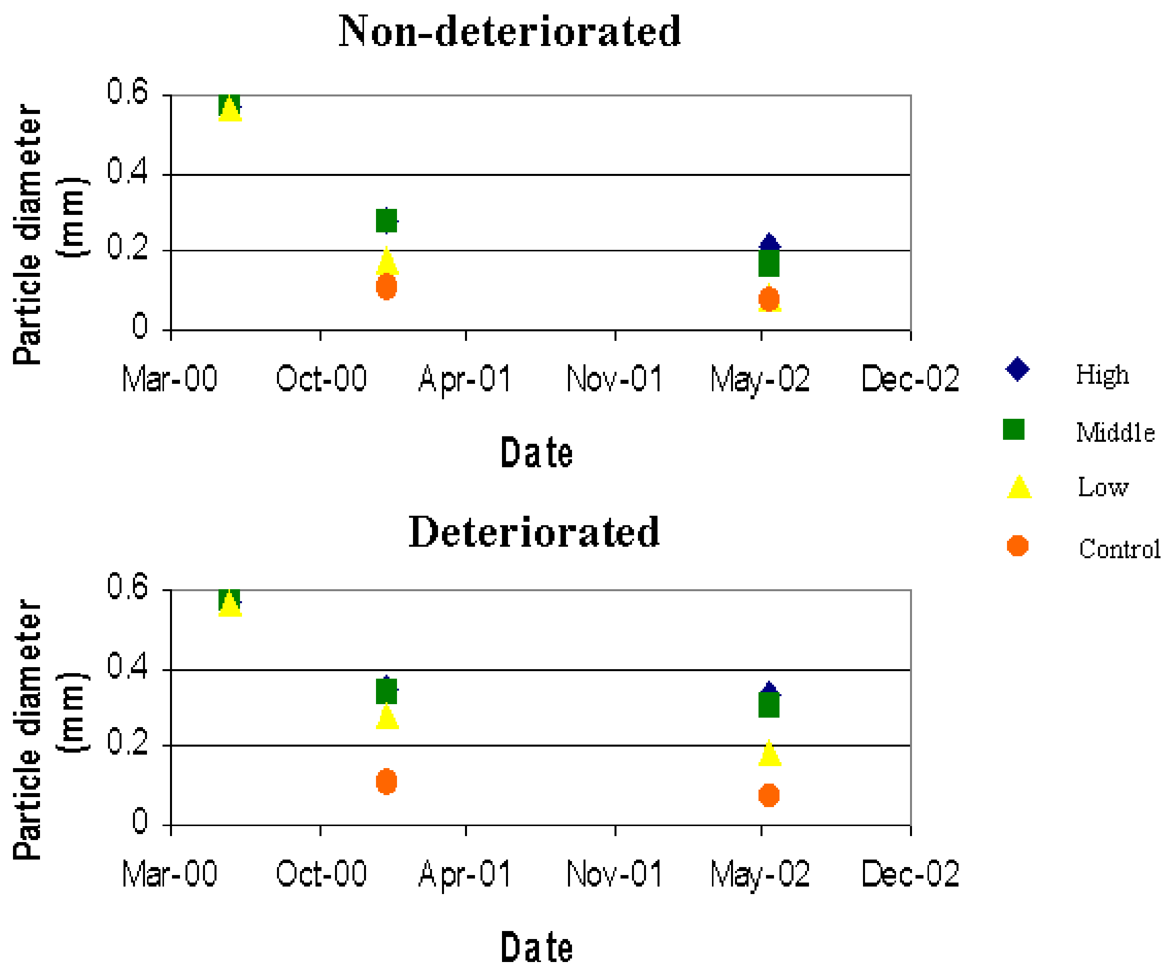

Figure 13. Surficial mean grain size for the deteriorated and non-deteriorated sties.

Velocities of over-marsh flows were recorded during a rising tide in each of the four-study sites and in both controls (Sept 2002). The mean velocities were higher in the treated deteriorated sites (4.05 cm s^{-1}) than in the treated non-deteriorated sites (1.75 cm s^{-1}). Deteriorated control velocities (5.68 cm s^{-1}) were also higher than non-deteriorated control velocities (1.79 cm s^{-1}) (Figure 14).

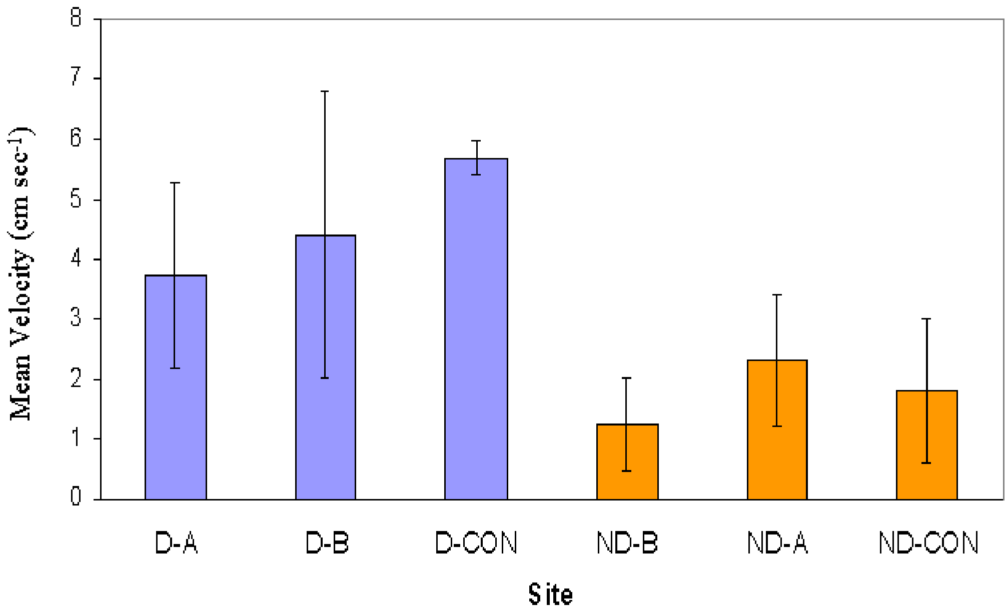

Figure 14. Mean flow velocities for the deteriorated and non-deteriorated sites. Each mean is calculated from three sampling bursts. Error bars indicate $\pm$ one standard deviation for the three sampling bursts.

Benthic Microalgal Biomass: Significant differences in benthic microalgae biomass, measured by chlorophyll *a*, were found both spatially and temporally for samples taken both pre-sediment addition and post sediment addition in all sites sampled. Samples taken at all sites, pre-sediment addition, were averaged according to the pre-treatment condition of the marsh. Samples taken from the two sites designated non-deteriorated were combined to find the average benthic microalgal biomass, as was done for the two sites designated deteriorated. Two way analysis of variance (Table 3) showed a significant difference in benthic microalgal biomass between non-deteriorated and deteriorated sites prior to sediment addition with temporal quantities as well ($p<0.0001$). Non-deteriorated sites, which were characterized by healthy *Spartina alterniflora* growth, showed significantly higher benthic microalgae biomass than in the deteriorated sites (Figure 15) both spatially (from site to site) and temporally (over a span of 5 months). Higher values of benthic microalgae biomass were associated with non-deteriorated sites during winter months and in the late spring. One way analysis of variance of non-deteriorated sites versus deteriorated sites showed that non-deteriorated control sites had a significantly higher mean *chl a* of 66.222 milligrams/m^2 while deteriorated control sites had a mean *chl a* of 13.332 milligrams/m^2 ($p<0.006$).

Table 3. Summary of all Effects1-VAR1 (non-deteriorated control), 2-VAR2 (deteriorated control).

Variable	df Effect	MS Effect	df Error	MS Error	F	p-level
1	1	72981.70	84	230.6608	316.4027	0.0000
2	5	6464.82	84	230.6608	28.0274	0.0000
12	5	5142.14	84	230.6608	22.2931	0.0000

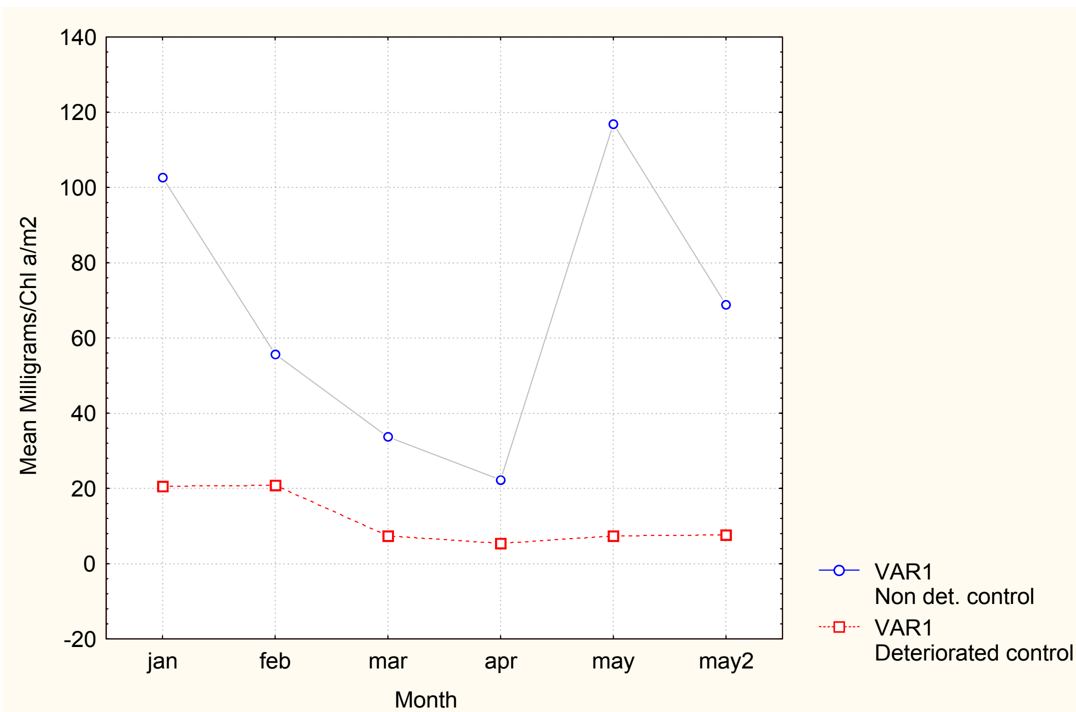

Figure 15. Mean Chl *a* of pre-sedimented sites/month.

Post sediment addition, mean benthic microalgae biomass was found to be both spatially and temporally significantly different between sediment amended sites and control, or no sediment addition, sites (Table 4). Monthly mean chlorophyll *a* was significantly greater in amended sites compared to the control, or non-amended, sites (Figure 16). A Tukey Kramer means comparison analysis showed that there were significantly greater amounts of benthic microalgae biomass present in all amended sites compared to both control sites (Table 4). Overall, benthic microalgae biomass was neither greater nor less when comparing amended sites (Figure 18) and a Tukey Kramer means comparison analysis and one way analysis of variance ($p<0.166$) showed no significant difference between amended sites.

Table 4. Means comparison of amended sites versus non-amended (control) sites.

Effect	Control	Treatment	Estimate	Standard error	DF	T Value	Pr> |t|
Sediment	0 inch	1 inch	-49.1941	4.8514	7	-10.14	<0.0001
Sediment	0 inch	2 inch	-55.9639	4.8514	7	-11.54	<0.0001
Sediment	0 inch	4 inch	-54.4799	4.8514	7	-11.23	<0.0001

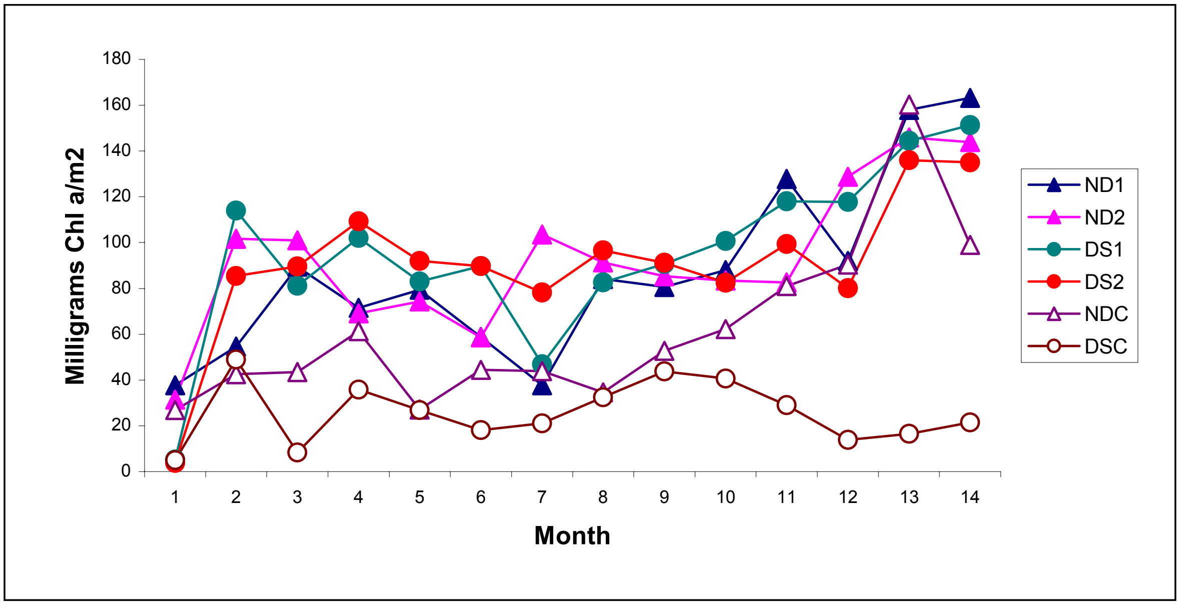

Figure 16. Comparison of mean monthly Chl *a* of amended sites versus control sites. ND indicates amended non-deteriorated sites, DS indicates amended deteriorated sites, and NDC and DSC indicate non-deteriorated and deteriorated control sites, respectively.

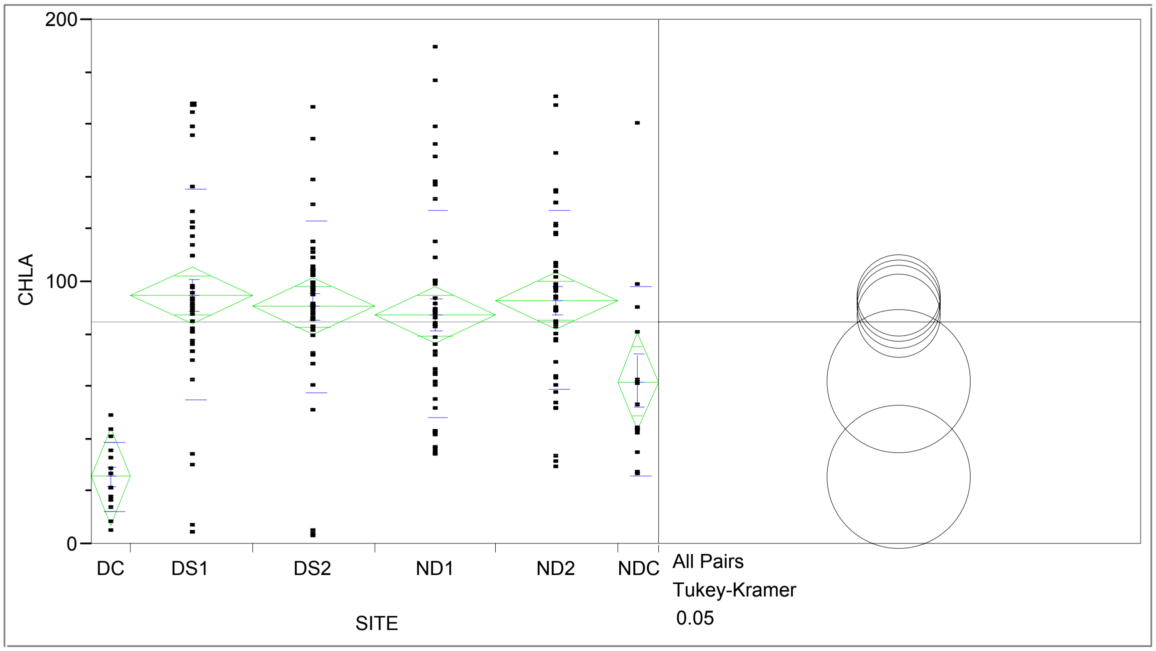

Figure 17. Comparison of Chl *a* mean and variability in amended and control sites. DC indicates the deteriorated control, NDC is the non-deteriorated control, DS indicates deteriorated, and ND is non-deteriorated.

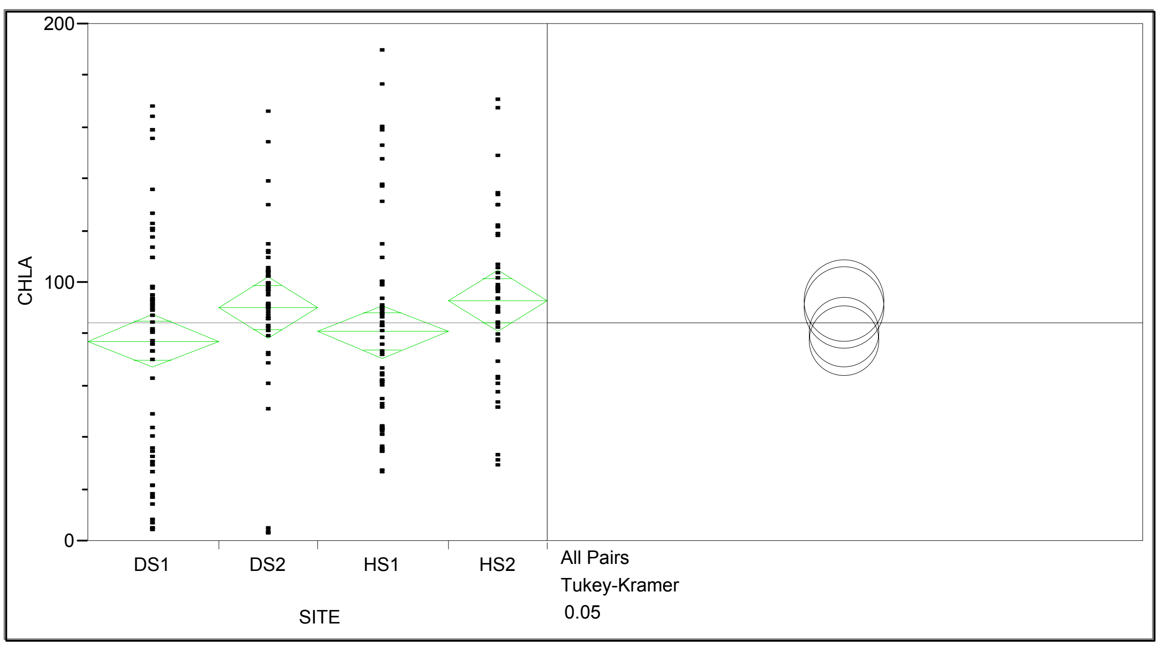

Figure 18. Comparison Chl *a* means and variability between deteriorated (DS) and non-deteriorated (HS) sites post sediment addition.

Analysis of sediment addition increments within sites also showed there to be no significant difference in benthic microalgae biomass between sediment additions (Table 5). Significant differences in mean benthic microalgae biomass were observed between the zero sediment addition (or control sites) and all of the three sediment additions (one inch, two inch, and four inch) but not within the additions themselves (Figure 19).

Table 5.- Means comparison of sediment increment additions.

Effect	Variable 1	Variable 2	Estimate	Standard error	DF	T Value	Pr> |t|
Sediment	1 inch	2 inch	-6.7698	3.9611	7	-1.71	0.1312
Sediment	1 inch	4 inch	-5.2857	3.9611	7	-1.33	0.2238
Sediment	2 inch	4 inch	1.4841	3.9611	7	.37	0.7190

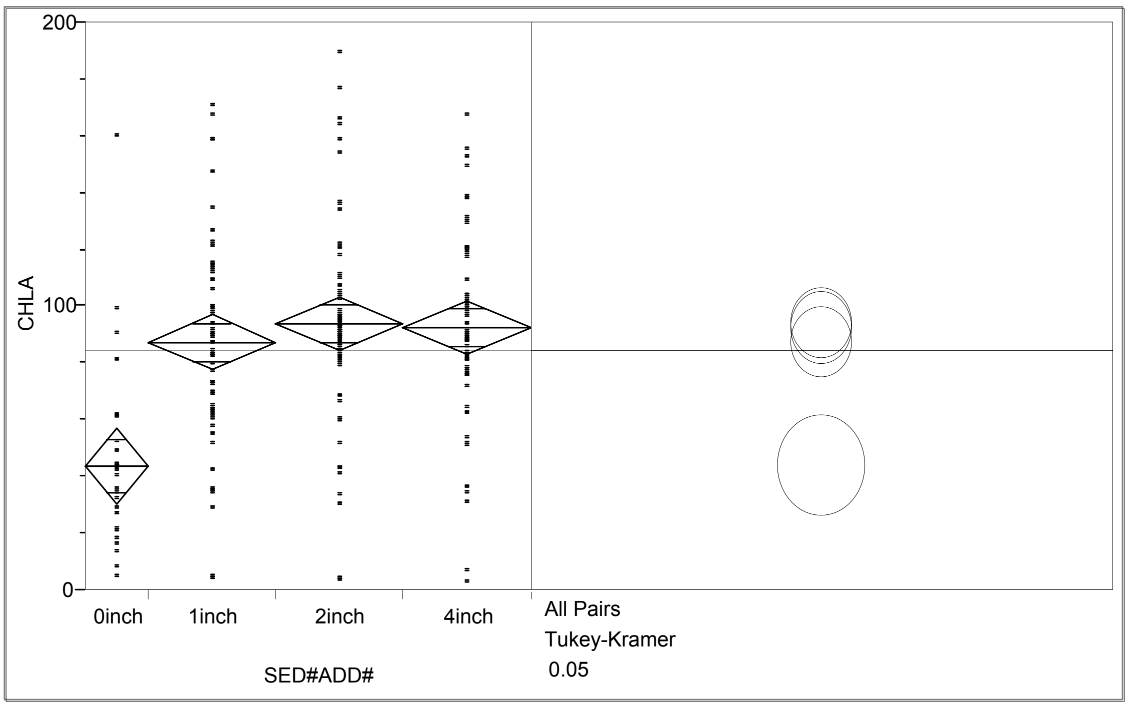

Figure 19. Comparison of Chl *a* means and variability among amended sites resulting from incremental sediment additions.

Benthic Infaunal Response: Twenty-two taxa were present in the initial infaunal sampling. Of these taxa, *Capitella capitata*, *Streblospio benedic*ti, *Nereis succinea* (all three ploychaetes), and oligochaetes were the numeric dominants. *S. benedicti*, *N. succinea*, and oligochaetes were all more common in non-degraded plots vs. degraded plots. Likewise *Tanais* sp. (a small crustacean that lives in and among the roots of marsh plants and detritus) was present only in the non-dregraded sites for this sampling period. *Gemma gemma* (a small brooding bivalve), *Heteromastus filiformis* (a capitellid polychaete), and *Laeonereis culveri* (a nereid polychaete) were only present in the degraded plots. Abundances tended to be higher in thin sediment addition plots vs. thick, however there was no significant difference.

After ten months there were 42 total taxa collected from both degraded and non-degraded plots. Most of these additional species had low abundances. During this second sampling period, the dominant taxa were *Capitella capitata*, sabellidae sp (juvenile sabellid polychaete), *Streblospio benedicti* (spionid polychaete), *Tharyx* sp. (cirratulid polychaete) and oligochaetes. There were some notable differences between degraded and non-degraded plots. *Leitoscoloplos* sp. (an orbinid polychaete) and *Laeonereis culveri* were again only present in the degraded plots, likewise the juvenile nereidae group (small nereid polychaetes too small to identify) were also present exclusively in the degraded plots and may represent young of the year of *Laeonereis*. *Streblospio benedicti* and *Tharyx* sp. were present in much higher abundances in the degraded plots. Small sabellid polychaetes (the second most abundant taxa) were found only in the non-degraded plots. Mites show the same pattern but this group are not true infauna and may be associated more with the vegetation than the marsh surface. Oligochaetes were the most numerous taxa found during this sampling and showed an even distribution between treatement types.

There was little difference between sediment thickness treatments within the plots, with no differences among thick and thin sediment additions. Differences among plot types had a greater impact on the infaunal community than actual sediment addition.

In these isolated marsh systems, where sediment starvation and circulation may be an issue (contributing to the degradation of the marsh plots in question), the recovery and/or recolonization of marsh plots is most likely to occur through the immigration and emigration of adults from adjacent marsh habitat or from local larval sources. Abundances were greater in the 10-month recovery compared to the initial samples (for both plot types) and there was some evidence that the plots were on similar development trajectories. Rapid colognization after sediment addition may refelct the oppourtunistic nature of the dominant infaunal taxa.

These results suggest that incremental sediment additions (even as much as 100 mm) may not have long-term impacts on the infaunal community. However it is important to note that scale of this study (10's of meters) and irregular addition sediment may have enhance colonization, allowing small areas within individual plots to persist and recolonize the rest of the plot area. This agrees with landscape mosaic theory suggesting that by limiting the size of the disturbance area or patch we increase the likelihood of recovery and quick colonization from adjacent undisturbed areas.

Table 6. Mean infaunal abundance and (stderr) for both degraded and non-degraded plots by treatment thickness. These means represent abundances 10 months after sediment addition (i.e., recovery).

	Non-Degraded		Degraded	
Species	**Thick**	**Thin**	**Thick**	**Thin**
amphipod sp.	0.25(0.25)	0	0	0.25(0.25)
Aricidea sp.	0	0	0	0.25(0.25)
Axiothella sp.	0	0	0	0.75(0.48)
Bezzia/palpomyia	0.5(0.29)	3.6(0.40)	2.67(2.19)	0.75(0.75)
Capitella capitata	36.75(8.82)	27.4(5.99)	13(5.69)	3(0.71)
Collembolla sp.	0.25(0.25)	0.8(0.37)	0.33(0.33)	0
Dolichopodid larvae	2.5(1.55)	4.8(0.49)	1(1.0)	0
Drilonereis sp.	0	0	0	0.5(0.5)
Gemma gemma	0	0	1.33(0.88)	0
Heteromastus filiformis	0	0	1.33(1.33)	1.25(1.25)
Hobsonia florida	0	0.6(0.6)	0	0
Illyanassa obsoleta	0	0	0.33(0.33)	1.25(0.75)
insect pupae	0.25(0.25)	1.2(0.73)	1(1.0)	0.5(0.5)
juv. bivalve	3(0.91)	4.8(1.16)	6.67(2.41)	7.25(3.09)
juv. gastropod	3.5(1.04)	6.6(2.77)	0.67(0.33)	0.75(0.49)
juv. nereidae	0.25(0.25)	0.4(0.40)	13(5.57)	18.25(1.03)
Laeonereis culveri	0	0	1.0(0.59)	3.5(0.87)
Leitoscoloplos sp.	0	0	0	6.25(2.78)
Mediomastus sp.	1.25(0.63)	1(1.0)	0	0.5(0.5)
Mite	35.25(25.92)	10(2.76)	0	0
Mulinia lateralis	0	0.2(0.2)	0	0
Mytilus sp.	1.25(0.95)	0.2(0.2)	0	0
Nemertean	0.75(0.48)	0.8(0.58)	0.33(0.33)	0.5(0.29)
Nereis falsa	0	0	0	0.5(0.29)
Nereis succinea	0	2.2(1.46)	1(1.0)	3.5(1.94)
oligochaete	49.75(17.57)	49.4(11.89)	53(19.31)	35.5(25.52)
Orchestia sp.	1(0.58)	1(0.63)	0	0.75(0.48)
platyhelminth	0	0.2(0.20)	0	0
Polypedilum sp.	0	0.6(0.60)	0	0
Prionopsio heterobranchia	0	0	0	0.25(0.25)
Prionspio sp.	0	0.2(0.20)	0	1(0.71)
Sabellidae	55.5(23.11)	55.4(15.31)	2.33(2.34)	0.25(0.25)
Scolelepis sp.	0	0	0	0.25(0.25)
Scoloplos sp.	0	0	0	0.25(0.25)
Streblospio benedicti	1(0.41)	7.6(4.36)	7.33(1.67)	26(7.15)
Syllidae sp.	0.75(0.75)	0.4(0.40)	1.33(0.88)	1.75(0.63)
Syrphidae sp.	0	0.2(0.20)	0	0
Tanais sp.	0.25(0.25)	7.4(2.62)	2(1.16)	0.25(0.25)
Leptochelia sp.	0	3.6(3.60)	0	0
Tharyx sp.	0.25(0.25)	0	35.67(9.69)	29.25(20.93)
Uca pugilatnor	0.75(0.48)	0	0	0

Discussion

The results of this study indicate that the addition of dredged material on the surface of deteriorating marshes led to a two-fold increase in vascular plant stem densities but had little to no effect on the overall height of *Spartina alterniflora*. While the total thickness of sediment added to each plot did not significantly affect stem densities, it is apparent that by the end of the project the convergence of stem densities in the non-deteriorated and the deteriorated sites was greatest for the areas that received the thickest additions. Sediment additions had little to no impact on stem densities or plant height in non-deteriorating sites.

Sediment additions also resulted in higher eH values (e.g. higher oxygen levels) in both deteriorating and non-deteriorating marshes and the highest eH values were associated with areas that received the thickest sediment additions. It is likely that these changes in eH improved soil conditions and led to the observed improvement of canopy in the deteriorated sites. One additional benefit of the increase in plant cover in the deteriorated sites is an increased potential for flow baffling. Velocities in the amended deteriorated sites were lower than velocities in controls, but still higher than in the non-deteriorated sites. However, the latter were more densely vegetated to begin with. The net effect appears to be that the soupy surface sediments in the deteriorated sites were stabilized by the addition of coarser material, additional plant growth, and reduction in flow velocity; thus, preventing further erosion of these sites.

The addition of dredged sediment material to the marsh surface appears to be beneficial to the production of BMA. Benthic microalgal biomass, measured as mean sediment chlorophyll *a*, was observed in significantly higher quantities in non-deteriorated pre-sediment addition sites and at all sites post sediment addition. The apparent amount of sediment added to the marsh surface seems to be a non-factor, however, some evidence suggests that the grain size of the sediment added may be important. The results further suggest that a healthy or non-deteriorated preexisting marsh habitat, which was determined by percent cover of healthy *Spartina alterniflora* growth, may affect benthic microalgae production. Over the duration of the study, the taxonomic diversity of benthic microalgae was essentially unchanged eventhough the biomass was affected. The addition of fill material to the marsh surface also led to an increase in marsh elevation which was beneficial to the production of BMA in sites that were not only previously deteriorated, but also to those that were previously classified as non-deteriorated.

Benthic infaunal data suggest that while sediment placement may have had a short-term affect on community structure, that recovery occurred quickly following sediment addition. Further, these data indicate that over the long-term, sediment additions did not negatively affect benthic infaunal diversity or abundance. Further experimentation is necessary to further constrain "tolerable" levels of sediment additions. Our data indicate that thicknesses of 2 to 10 cm may be beneficial, especially if distributed in a manner that enhances edge effects and includes both "thick" and "thin" regions. Obviously the addition of too much sediment to the marsh surface could be deleterious, but this study has not been able to detect that threshold. One future study that may be of interest to managers examining the process of restoring degraded marsh and preserving marsh function throughout the restoration process would be to conduct incremental sediment additions on a larger scale with multiple patches/plot with varying spacing between the plot to evaluate potential effects on recovery time, especially in areas where large areas (km) of degraded marsh threaten habitat complexity of overall function.

Technology Transfer and Management Application

The results of this project have been disseminated to a variety of audiences through a variety of venues. We have presented our results at regional, national and international scientific meetings and shared our results with local managers and state officials. In addition, we have participated in other dissemination activities supported by the CICEET program. Below is a list of these activities including activities resulting in publications.

Presentation of project overview and preliminary results as part of another CICEET funded project "Estuaries Live!". This was a joint effort facilitated by Dr. John Taggert of the NCNERR and Ms. Susan Lovelace also of NCNERR. We broadcast our overview live from the field via internet links. The North Carolina Coastal Resources Commission and several "on-line" elementary schools were the primary audience.

Results from this research were disseminated to local community members through a web paged developed and maintained for the project:www.uncwil.edu/people/lynnl/ciceet.htm Further, data and results from the project have been incorporated into educational curriculum including: Ecology of Coastal System (BIO434), Development of Wetlands (GLY591), Coastal Sediment Dynamics (GLY555); and Honors Environmental Geology (GLY120) at the University of North Carolina at Wilmington.

Results were presented to the scientific community through presentations at several regional, national, and international meetings. These included: Southeastern Geological Society of America (2000 and 2001); Benthic Ecology Meeting (2000); Estuarine Research Federation Meeting (2001); International Coastal Symposium (2002); and the annual meeting of the National Geological Society of America (2002).

Scientific and Academic Achievement

Thus far, this research has produced seven published abstracts (below)and two manuscripts in preparation: 1) "The effects of thin layer dredge disponsal on tidal marsh processes, Masonboro Island, NC" to be submitted to Estuaries, and 2) "Benthic microalgal response to sediment additions" to be submitted to a journal to be determined.

Published Abstracts:

Leonard, L.A. 2000. Tidal marsh restoration: recent efforts to improve design and function. In: The Geological Society of America Southeastern Section 49th Annual Meeting, Abstracts with Programs, p. 46.

Wren, Ansley and Leonard, Lynn 2000. A field study of fine scale hydrodynamics in marsh canopies and implications of the impacts on particle deposition and distribution in coastal marshes. In: The Geological Society of America Southeastern Section 49th Annual Meeting, Abstracts with Programs, p. 52.

Leonard, Lynn A., Laws Richard A., Cahoon, Lawrence, Posey, Martin; Alphin, Troy; Reesey, Heather, and Gina Panasik 2000. Sediment Recycling: Marsh renourishment through dredged material disposal. In: Benthic Ecology 29th Annual Meeting, Abstracts, p. 86.

Panasik, G.M. Deteriorated Marsh Ecosystems: Can the Addition of Inorganic Sediment Increase Benthic Microalgae Biomass? 16th Biennial Conference of the Estuarine Research Federation, Conference Abstracts, 4-8 November, St. Pete Beach, p. 105.

Croft, A. and L. Leonard. The Effects of thin Layer Disposal of Dredged Material on Tidal Marsh Processes. 16th Biennial Conference of the Estuarine Research Federation, Conference Abstracts, 4-8 November, St. Pete Beach, p. 30.

Croft, Alex L; and L.A. Leonard. 2001. The effects of dredged material disposal of tidal marsh processes. Geological Society of America Southeastern Section, Abstracts with Programs, A-71.

Croft, Alex L; L.A. Leonard. 2002. Thin layer dredge material disposal impacts on tidal marsh processes. National Meeting of the Geological Society of America, Abstracts with Programs, 27-30 October, Denver, Colorado.

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Masters of Science Theses

Croft, Alex L. The effects of thin layer disposal on tidal marsh processes. University of North Carolina at Wilmington, Department of Earth Science.

Panasik, Gina M. Deteriorated marsh ecosystems: Can the addition of inorganic sediment increase benthic microalgea biomass? University of North Carolina at Wilmington, Department of Marine Biology.

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