Christine M. Voss. LONG-TERM (> 10 YR) EFFECTS OF THIN-LAYER DREDGED MATERIAL DEPOSITION ON AN IRREGULARLY FLOODED *JUNCUS ROEMERIANUS*-DOMINATED BRACKISH MARSH (Under the direction of Dr. Joseph J. Luczkovich) Department of Biology, April 2006.

The application of thin-layer dredged material onto marsh habitat was at first proposed as an economic and environmentally acceptable means of dredged material disposal; in recent years this technique has been under consideration as a management tool for the augmentation of eroding coastal marsh habitat. This study assessed the 12-year ecological impact of various thicknesses (0, 2, 4 and 10 cm) of thin-layer dredged material applied once to *Juncus roemerianus*-dominated marsh habitat near Wysocking Bay, North Carolina. My objectives were to experimentally determine the effects of thin-layer treatments on specific physical parameters (water level, plot elevation and bulk density and percent organic matter of substrate) and biological parameters (biomass and density of flora and fauna) and to describe temporal changes in marsh habitat following a thin-layer dredged material depositional disturbance. At 12 years, marsh elevation ($P < 0.001$), substrate organic matter ($P < 0.001$) and fish (of which 89% were *Fundulus heteroclitus*) biomass ($P=0.003$) and density ($P= 0.005$) remained significantly altered by thin-layer treatments. At 12 years, *J. roemerianus* biomass and density remained depressed in 2-cm, 4-cm and 10-cm treatment groups and a greater abundance of senescent *Juncus* shoots and upland vegetative species were observed in the 10-cm treatment group. My results contrast with results of other studies in which thin-layer amendments of 2 cm -10 cm depth were applied to eroding *Spartina alterniflora*-dominated marsh habitats. At Wysocking Bay, the deposition of 4 cm and 10 cm

thicknesses of thin-layer treatment altered marsh elevation significantly for 12 years and elevation appeared to prevent the recovery of marsh biota.

LONG-TERM (> 10 YR) EFFECTS OF THIN-LAYER DREDGED MATERIAL DEPOSITION ON AN IRREGULARLY FLOODED *JUNCUS ROEMERIANUS*-DOMINATED BRACKISH MARSH

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LONG-TERM EFFECTS OF THIN-LAYER DREDGED MATERIAL DEPOSITION
ON AN IRREGULARLY FLOODED
JUNCUS ROEMERIANUS-DOMINATED
BRACKISH MARSH

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INTRODUCTION

Coastal marshes are highly productive habitats that provide nursery functions for fishes and aquatic invertebrates, serve as keystone sites for biogeochemical processes and act as transition zones between terrestrial and estuarine environments. In North Carolina, marsh environments can be generally divided into two main types based on flooding regime: 1) tidal (regular high and low diurnal tides) and 2) irregularly-flooded (wind-driven water level changes). As true for much of the U.S. Atlantic and Gulf coasts, N.C. tidal marshes are dominated by *Spartina alterniflora*, whereas irregularly-flooded marshes are dominated by *Juncus roemerianus*. Both of these marsh types occur in estuarine waters along Pamlico Sound, however the irregularly-flooded marshes occur away from the inlets and on the western side of the sound.

Disturbance of marshes can occur naturally due to storm events and overwash, wrack deposition, and fires (Knowles et al. 1991; Schmalzer et al.1991; Courtemanche 1996; Mitsch and Gosselink 2000). However, disturbances also result from anthropogenic activities such as land development, mosquito control, and dredging and filling activities (Kuenzler and Marshall 1973; Deegan et al. 1984; Swenson and Turner 1987; Anderson 1989; Wilber 1993; Kennish 2001). In this study, I experimentally investigate the impacts to the physical and biological environments of an anthropogenic disturbance (dredged material additions) to an irregularly flooded *Juncus roemerianus*-dominated marsh in Wysocking Bay, NC over a 12-year period.

Ecology of Irregularly-Flooded Marshes

Irregularly-flooded marshes are those with an inundation regime controlled predominantly by meteorological forces and less so by astronomical forces. I have chosen to refer to this inundation regime as "irregularly-flooded" rather than "non-tidal", because muted astronomical patterns often remain to be observed. Irregularly-flooded marsh habitats differ from tidal marsh habitats in some important ways: 1) wave and current energy is reduced compared to diurnally-flooded ecosystems, 2) extent and duration of inundation is not regular nor readily predictable and 3) sustained inundation or lack of inundation can occur for weeks at a time, although sustained seasonal flooding typically occurs in spring and fall in North Carolina.

Juncus roemerianus is the dominant macrophyte found among brackish and saline irregularly-flooded marsh habitats along the southeastern U.S. coasts from southern New Jersey to Texas (Eleuterius 1975; Stout 1984; Mitsch and Gosselink 2000). *Juncus roemerianus* is tolerant to a wide range of environmental conditions (Eleuterius 1975; Eleuterius 1984; Stout 1984; Christian et al. 1990; Knowles 1991; Woerner and Hackney 1997; Brinson and Christian 1999). Pennings et al. (2005) found that the lower elevation limit of *J. roemerianus* was likely due to environmental factors (inundation and salt stress) in both field transplant and glasshouse experiments and that competition from *S. alterniflora* was not a significant factor in *J. roemerianus* growth. *Juncus roemerianus*-dominated marshes characteristically retain a greater proportion of brown (dead/senescing) leaves to green (live) leaves as compared to adjacent *S. alterniflora*-dominated marshes. At Cedar Island NWR, North Carolina, the leaves of *J. roemerianus* were observed to grow for an average of 259 days, tended to remain standing as they senesced (312 days on average) and standing dead leaves decomposed slowly at a rate of 0.29 leaves per year (Christian et al. 1990). This standing dead *Juncus* biomass may baffle wave and current energy throughout *Juncus*-dominated communities. Dominance of *J. roemerianus* has been found to be stable despite temporary declines caused by disturbances from wrack deposition and fire events (Stout 1984; Knowles 1991; Brinson and Christian 1999). Another characteristic feature of *J. roemerianus*-dominated marshes is the relatively high organic content of marsh substrate as compared to tidally-flooded systems; these peat-based soils are believed to be responsible for the bioaccretion which allows some coastal marshes to maintain elevation in the face of sea level rise (Moorehead and Brinson 1995).

Faunal utilization and nursery function of irregularly-flooded marshes is generally the same as in tidal marshes however, because the marsh is flooded for longer intervals, aquatic organisms have opportunity for more distant migrations across marsh platform (Marraro et al. 1991).

Marsh Disturbance Ecology

Coastal marshes are habitats often characterized by frequent disturbances. Marsh plant communities are low in species diversity and the typically few dominant species tend to be notably stress tolerant; the balance of the plant community includes random patches of secondary species (Stout 1984; Knowles 1991; Bertness and Ellison 1997; Mitsch and Gosselink 2000). Species site position is generally dictated by marsh hydrodynamics and salinity gradient, often varying in elevation by only a few centimeters; however, biotic interactions have been found to also influence community composition and distribution (Stout 1984; Bertness 1991; Hook 1991; Knowles 1991; Woerner and Hackney 1997; Bertness and Ellison 1997). Some of the natural disturbances experienced by marshes occur gradually, such as elevation changes due to sedimentation, accretion and sea-level rise; while others are more abrupt such as those caused by wrack deposition, fire or storm overwash. Depositions of wrack (rafted mats of dead vegetation) have caused shifts in community structure among marsh vegetation; however dominant vegetation is believed to recover after several years (Reidenbaugh et al. 1983; Hartman 1988; Knowles et al. 1991; Brinson and Christian 1999; Tolley and Christian 1999). After a Florida marsh habitat was submitted to disturbance by fire, Schmalzer et al. (1991) found that species composition of a *J. roemerianus* and *Spartina bakeri* marsh was similar to pre-burn conditions at 1 year post-burn; however, there was less live biomass of these two species when compared to pre-burn conditions. Courtemanche (1996) found that dominant marsh vegetation had a 98% recovery rate after an overwash event, caused by Hurricane Andrew. Many of the organisms in the marsh habitat community are well adapted to stress conditions. It may be that marsh habitat communities are capable of responding to some anthropogenic stresses, such as thin-layer dredged material deposition, much as they do when natural stresses occur.

Dredging and the Disposal of Dredged Materials

Our coastal waterways serve as an essential mode of transportation; pertinent for commerce, fisheries access, transportation and recreation. The U.S. Army Corps of Engineers (USACE) is responsible for maintaining secure and navigable waterways within U.S. federal waters (USACE 2001). Sustaining adequate channel and canal depth is foremost among the waterway maintenance duties of USACE. Waterway depth is most often maintained by periodic dredging. Approximately 300 X 10^6 cubic meters of sediment are dredged each year in the U.S. [National Research Council (NRC) 1985]. Nearly 75% of this amount is disposed at sites proximal or adjacent to the project (NRC 1985).

Several options exist for dredged material disposal: 1) discard into adjacent waters, 2) discard onto near-by land sites, 3) transport to inland sites as fill or compost and 4) transport to ocean disposal sites. The deposition of dredged material into water has well-established negative ecological impacts on pelagic and benthic communities at disposal site and mine site (Onuf 1994; Gibson and Looney 1994; Long et al.1996; Kennish 2002; Thrush et al. 2004). The deposition of dredged material onto land has been executed at various types of sites, for instance small islands created from dredged material can provide nesting habitat for colonial shorebirds (Parnell and Shields 1990) and dredged material applied to ocean beach berm can augment eroded beaches [The National Academy Press (NAP) 1995]. Disposal of dredged material onto ocean beach berms as "beach renourishment" has been shown to negatively impact beach infauna and associated littoral food webs (Peterson et al. 2000a; Peterson et al. 2000b), inhibit sea turtle nesting (Milton et al. 1997; Steintz et al. 1998) and elicit other ecological concerns (NAP 1995). Engineering concerns associated with these shoreline disposal options involve sediment grain-size incompatibility (Finkl et al. 1997; Milton et al. 1997; Steintz et al. 1998); additionally, sediment resuspension is likely in especially high-energy areas.

Transporting dredged material to distant locations is known to be relatively expensive. According to Lund (1991) the increase in cost of transporting and placing spoil material inland is approximately \$0.38 m^{-3}. Transportation costs alone would translate into \$28.5 million to relocate the 25% of dredged material not deposited near the dredge site, as reported by the NRC (1985) above.

Coastal marsh habitat has been previously used as sites of dredged material disposal; this practice was considered the least expensive option for maintaining the canals that traverse marshes and near-by estuarine channels (Boesch et al. 1994). This disposal method ceased as the ecological role of marsh habitat became evident. Marsh habitats fulfill a variety of critical ecosystem functions, such as: 1) essential fish habitat (Weinstein 1979; Boesch and Turner 1984; Minello et al. 2003) and habitat for other nekton (Zimmerman and Minello 1984), 2) a source of primary production (Teal 1962; Weisberg and Lotrich 1982; Kneib 1984; Peterson and Turner 1994), 3) sites of biogeochemical cycling (Pomeroy et al. 1972; Valiela et al. 1978; Valiela and Teal 1979; Odum et al. 1979; Nixon 1980; Morris and Bowden 1986), and 4) storm buffers as well as storage sites and filter mechanisms for storm water (Mitsch and Gosselink 2000). Marsh habitats and adjacent sub-tidal habitats are important for the trophic transfer of primary production from the marsh to the estuarine and coastal ecosystems (Nixon and

Oviatt 1973; Kneib and Stiven 1978; Nixon 1980; Weisberg and Lotrich 1982; Boesch and Turner 1984; Kneib 1997; Komarow et al. 1999; Kneib 2000; Paterson and Whitfield 2000; Peterson et al. 2000b; Smith et al. 2000; Castellanos and Rozas 2001; Currin et al. 2003). Dredging and the disposal of dredged materials have been found to alter aquatic, marsh and associated sub-tidal habitats (Burdick 1967; Kuenzler and Marshall 1973; Deegan et al. 1984; Swenson and Turner 1987; Wilber 1993; Wulff et al.1997; Kennish 2001). Physical and chemical effects from dredging practices include mechanical damage to benthos, altered hydroperiods of marshes, changes in local circulation patterns of estuaries, increases in turbidity of estuarine waters, alterations to ecosystem chemistry and possible exposure to toxins (Kuenzler and Marshall 1973; Deegan et al. 1984; NRC 1985; Swenson and Turner 1987). Subsequent habitat alterations result in biological changes to the marsh food web, marsh utilization by fauna and to the adjacent sub-tidal ecosystem (Kuenzler and Marshall 1973; Wulff et al. 1997; Kennish 2001; Kennish 2002).

Coastal wetlands are currently protected by provisions in the Federal Water Pollution Control Act, enacted in 1972 and amended in 1977 and the Clean Water Act of 1982. Any dredging or filling of wetlands must be permitted by the USACE as per Section 404 of the Clean Water Act. The main objective of the Clean Water Act Section 404 is to avoid, minimize and/or mitigate dredging and filling impacts on wetlands.

Thin-Layer Dredged Material Applied to Marsh Habitat

Thin-layer technique is being evaluated as a method of applying dredged material onto marsh habitat. This contrasts with the traditional bucket method of dredged material disposal. Thin-layer application technique has been suggested to have minimal negative ecological impacts or possibly, positive impacts to salt marsh plant growth and recolonization (Ford et al. 1999; Leonard et al. 2002). This method (Jet Spray® system) suspends the dredged sediments into an aqueous slurry, and then expels this slurry via a high-pressure hose. The Jet Spray® system is able to spray the dredged-material slurry up to a distance of 80 meters (Ford et al. 1999). Typically, this results in a somewhat thin and evenly-dispersed layer across the marsh surface. In 1981, thin-layer technique was used to apply dredged material onto marsh habitat for the first time in the United States at Lake Landing canal near Wysocking Bay, North Carolina (Figure 1). These first exploratory sites are adjacent to the experimental site described in this paper.

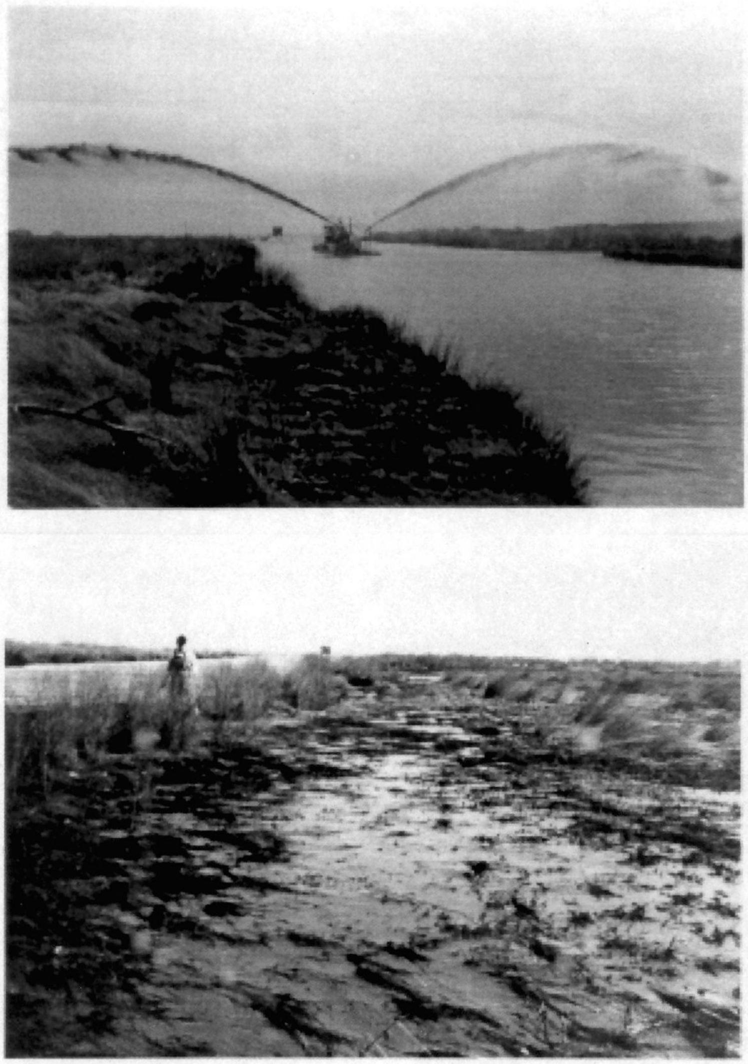

Figure 1. (top) Jet Spray® system applying dredge spoil and water slurry to the marsh at Lake Landing Canal, near Wysocking Bay, NC in 1981. (bottom) The associated *Juncus roemerianus*-dominated marsh after thin-layer dredge-spoil application.

The thickness of a "thin" layer of dredged material applied to marsh habitat has been reported to range from 2 - 91 cm, although shallower depths (2-10 cm) have been favored in recent years (Wilber 1993; Ford et al. 1999; Leonard et al. 2002). In a review paper, Wilber (1993) summarizes four case studies of thin-layer dredged material applied to marsh habitat which offer varied results. In the three, non-experimental studies, it was difficult to discern if changes observed in marsh vegetation were caused by the deposition of thin-layer dredged material or if variances were patterns which naturally occur within marsh habitat. One of these studies, conducted by Reimold et al. (1978), was experimentally designed with dredged material depth (8, 15, 23, 30, 61 and 91 cm) and character (sand, clay and sand/clay mix) as variables. Here, *S. alterniflora* demonstrated recovery in depths ≤ 23 cm; vegetation density was greater in sandy material and vegetation biomass was greater in clay material (Reimold et al. 1978). Results of this study were likely impacted by an artifact effect of the pipes used to contain the dredged material; varying heights of the pipes varied the amount of sunlight exposure and flooding regimes among treatments (Reimold et al. 1978). Wilber (1993) suggested that the two ways in which excess dredged material may kill or alter marsh vegetation composition was physical effects (smothering) and chemical effects (hydrogen sulfide toxicity). Overall, thickness of the dredged-material layer was suggested to be the most significant factor in marsh recovery (Wilber 1993). Ford et al. (1999) suggest that thin-layer deposition (2.3 cm -12.9 cm) of dredged material may also be effective in restoring and maintaining marsh elevation among subsiding marshes, as in the case of trials in Louisiana (see Discussion for details). Leonard et al. (2002) reported a two-fold increase in *S. alterniflora* stem densities and an increase in benthic microalgae, two years after dredged material had been applied at a 10-cm thickness in a deteriorating North Carolina tidal marsh habitat; a rapid recovery among benthic infauna was also observed (Leonard et al. 2002).

Wysocking Bay Dredging and Experiments

In 1981, the U.S. Army Corps of Engineers (USACE) first explored the use of thin-layer dredged material along Lake Landing Canal using Jet Spray® technology, near Lake Mattamuskeet, North Carolina (Figure 1). At this site, three dredged-material thicknesses were approximated at 5 cm, 20 cm and > 50 cm in depth when applied to marsh platform. Ten years later (1991), Wilber et al. (1992) found a general decrease in *J. roemerianus* biomass associated with increased thickness of dredged material depth and as compared to reference sites; however, no inference could be clearly stated due to high variability among sites. In 1992, Luczkovich and Knowles (pers. comm.) established an empirical study to determine the ecological effects of thin-layer dredged material (0, 2, 4, 10 cm in depth) sponsored by the USACE and North Carolina Sea Grant. After one year, Luczkovich and Knowles (pers. comm.) found a significant decrease in the amount of *J. roemerianus* biomass in only the 10-cm treatment group; this thesis is a continuation of the empirical study initiated by Luczkovich and Knowles in 1992.

The long-term (> 10 year) impact of thin-layer dredged material (up to 10 cm in depth) has not been determined for marsh habitat. My objectives were to experimentally determine the effects of various thicknesses (0, 2, 4, and 10 cm) of dredged material on marsh habitat over a 12 year period, to monitor specific physical, chemical and biological transformations and to describe temporal changes in a *J. roemerianus*-dominated marsh habitat following a thin-layer dredged material depositional disturbance.

Hypotheses tested were:

H_1: There will be differences in the physical and biological parameters within marsh habitat among experimental dredged material thickness levels in the short-term (1 year).

H_2: The differences in the physical and biological parameters within marsh habitat among experimental dredged material thickness levels will persist over the long-term (> 10 years).

MATERIALS AND METHODS

Site Description

The experimental study site was located along a canal on a *Juncus roemerianus*-dominated marsh near Wysocking Bay, North Carolina; in the vicinity of the original thin-layer-deposition exploratory sites selected by the USACE in 1981 (Wilber et al. 1992). Located on the southeastern edge of the Albemarle-Pamlico peninsula (N 35° 26' 11.3", W 76° 04' 10.7"), the irregular flooding regime of this brackish marsh is largely dictated by wind-driven events, with little lunar (M_2) tidal influence (Figures 2 and 3). This site was chosen because *J. roemerianus*-dominated marshes represent approximately 60% of the coastal marsh habitat in North Carolina (Wilson 1962) and because of its proximity to the original exploratory sites for comparison (Wilber et al. 1992).

Figure 2. Map of the North Carolina coast showing the location of Wysocking Bay

Figure 3. Map of Wysocking Bay (a) exploratory and (b) experimental marsh study sites.
Dotted-line box denotes study area.

Experimental Design

A randomized-block design was used to control for minor pre-existing elevation differences at this site. The 16 experimental plots encompassed a 10 m X 200 m area of marsh habitat delineated parallel to and 30 m from the canal edge. The study site was divided into 4 blocks, each containing four 4-meter by 4-meter plots with similar pre-existing elevations; each plot received one of four randomly assigned treatments varying in dredged material thickness. The four experimental dredged material treatment depths were: no dredged material (control), 2 cm of dredged material, 4 cm of dredged material and 10 cm of dredged material. The dredged material treatments were applied once in October 1992 using a 2-hp gasoline-powered diaphragm pump which created a sediment-and-water slurry similar to that of the Jet Spray® system. This portable gasoline-powered pump allowed for control of deposition amounts. Wood barriers of appropriate height were placed around the perimeter of the 2 cm, 4 cm and 10 cm depositional treatment plots. Figure 4 is an aerial photo of the Wysocking Bay marsh study site with a diagram depicting the layout of experimental plots.

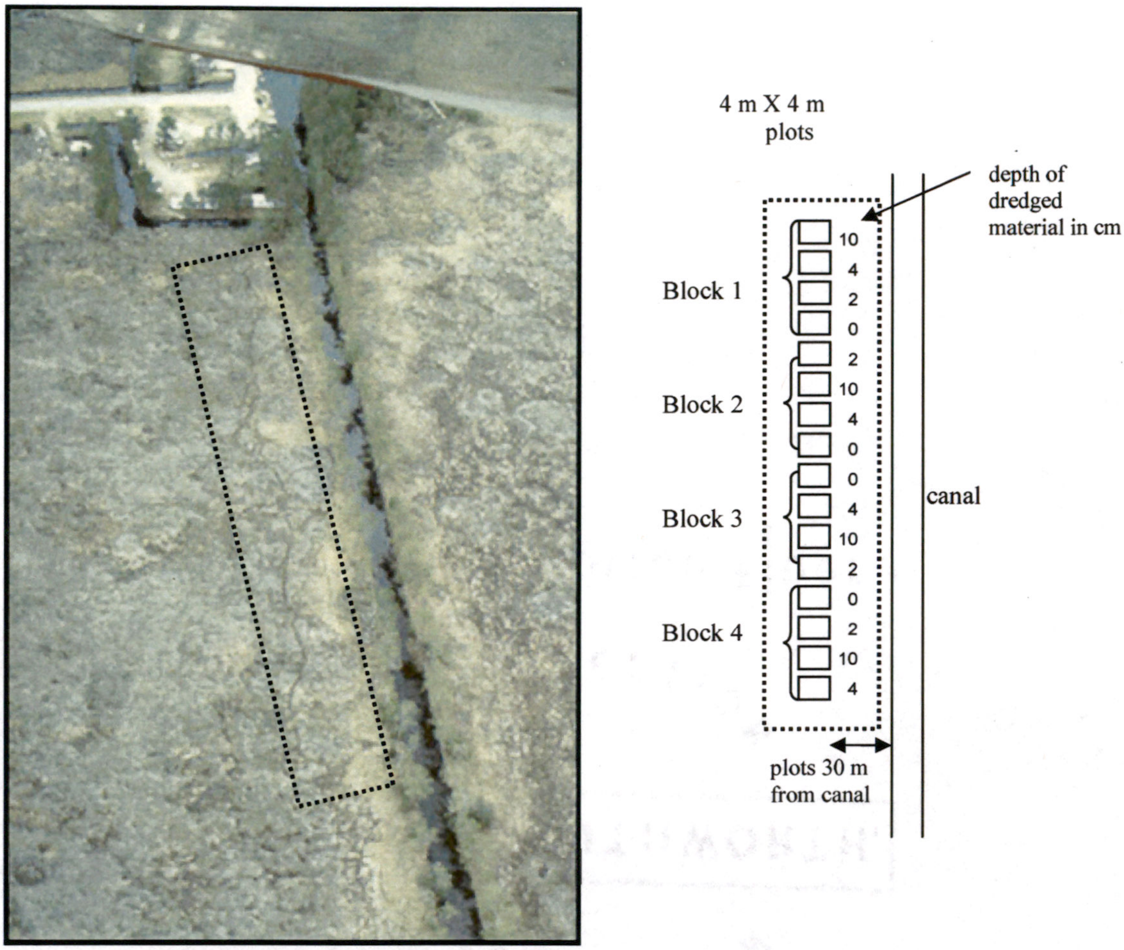

Figure 4. Wysocking Bay Marsh experimental study site aerial photo and experimental design layout. Dotted-line box denotes study area.

Sampling

This study monitored physical parameters (water level, plot elevation and bulk density and percent organic matter of substrate) and biological parameters (biomass and density of flora and fauna) of the marsh habitat in the study area over a period of 12 years.

Physical parameters

A Stevens® water-level recorder (Model 71) equipped with a 1:1 gear ratio was used to continuously monitor water level from a central well location; from this, hydroperiod data was calculated for each plot daily. Plot elevation was determined by levelling a minimum of 12 random points per plot with a Topcon (Model AT-F2) automatic level equipped with 40-power scope in 1992 and 1993, and with a Topcon (Model RL-50A) rotating-laser system in 2004; an on-site benchmark was maintained over the duration of the study. Sediment traps (3 per plot) were used to determine the exact depth of dredged material placed onto each plot.

Soil cores were obtained in 1993 (year one) and 2004 (year 12) to determine soil bulk density and percent organic matter. Four replicate 385-cm^3 cores (10-cm depth) were taken randomly from an undisturbed quadrant of each plot; soil samples were dried at 90 °C, homogenized via mortar and pestle, and then dried at 105°C until a consistent weight was maintained as per Allen (1989) and Klute (1986). Sample weights were recorded and averaged to determine the soil bulk density of each plot. Two sub-samples of each core replicate were then ashed at 500 °C for 3 hours to determine soil loss on ignition which serves as a proxy for percent organic content of organic soils (Allen 1989; Nelson and Sommers 1996).

Biological Parameters

Vegetation. Vegetation samples were harvested in September of 1992-1996 and 2003-2004. These samples were harvested from four replicate 20 cm X 20 cm sites per plot, by clipping plant material at the soil surface. Sample sites within each plot were selected at random using a plot sampling grid of 20 cm X 20 cm ($0.04m^2$) cells over laid on each plot; previously sampled cells were not used. Live shoots from each species and all dead vegetation were placed into individual labeled paper bags, and then dried at 85 ^oC until a consistent weight was maintained; weights of dried vegetation was recorded and averaged for each plot. Shoot densities within the 0.04 m^2 cells were recorded and averaged for each plot in 1992, 1993 and again in 2003 and 2004. In June 2004, all vegetative species in plots were identified and the percent cover of each was estimated visually. In October 2004, a visual assessment of *Juncus roemerianus* condition was conducted where shoots were classified as either: all live (green), all dead (brown) or senescent (part green and brown); a proportion of each classification was determined for each entire plot.

Fauna. Faunal biomass and density data was obtained by employing pit traps (Kneib and Stiven 1978; Kneib 1984; Marraro et al. 1991; Yozzo and Smith 1998; Able and Agan 2000), and simulated aquatic microhabitats (SAMs) (Kneib 1997), both passive collection devices. Pit traps were fashioned using opaque, plastic paint buckets (20-cm diameter, 14.5-cm height) with three 2.5-cm holes drilled through the bottom of each plastic bucket which was then covered by a precisely-fitted piece of nylon window screen (1.4-mm mesh) that was glued securely to bucket bottom. Additional adhesive was used to seal screen around drilled holes and along screen periphery to prevent the escape of organisms. Two pit traps were sited within each plot, by random placement within the 20 cm X 20 cm plot sampling grid, so that the top edge of the trap was just below the marsh substrate surface; water level within pit traps corresponded with that of the marsh. Aquatic and terrestrial organisms that swam or crawled into traps were stranded in pit traps as water level receded. Samples from pit traps were collected generally bimonthly after dredged material deposition treatment, October 1992 through September 1993 and October 2003 through November 2004; two pre-treatment samples were also obtained in September 1992. An effort was made to collect faunal samples after water level had flooded, and then soon receded from the study site; on occasions, samples were collected when the study site was submerged by water and also when the site was mostly dry.

Simulated Aquatic Microhabitats (SAMs) were used in addition to pit traps during the 2003-2004 sampling year to investigate faunal utilization of marsh on a smaller size-scale. SAMs are designed to simulate small pooled-water areas that occur as flood waters recede from the vegetated marsh surface and which offer an aquatic refuge to young nekton (Kneib 1997).

All organisms collected from pit traps and SAMs were preserved in a 10% formalin solution, most were stained with Rose bengal and all were contained in sealed, plastic Whirl-pak® bags until they could be enumerated and identified to the lowest taxonomic level practical in the laboratory. Specimens from pit traps were identified, enumerated, measured for total length and weighed, by species. Faunal biomass and density data from both pit traps (2 pit traps per plot) were summed for each plot for statistical analysis. Fauna from SAMs was identified, enumerated and summed for each plot by sample date; this data was analyzed separately from pit trap data.

Statistical Analyses

SYSTAT® for Windows® statistical software versions 10.2 (SYSTAT 2002) and 11.00.01 (SYSTAT 2004) were used for data analyses. Data were first tested for assumptions of normality; the Shapiro-Wilks test was used to evaluate distribution of data sets (normal distribution = $W \geq 0.85$) and a normal probability plot of residuals was used to assess homogeneity of variances visually. Data collected on water level, plot elevation, soil character and vegetation (except percent cover of atypical vegetative species) fit the assumptions of normality, thus allowing one-way analysis of variance (ANOVA) parametric analyses, with treatment thickness as the independent variable. The data from the fauna and percent cover of atypical vegetative species did not fit assumptions of normality (even with transformations); therefore, for these data I utilized the Kruskal-Wallis one-way ANOVA non-parametric analysis, with treatment thickness as the independent variable. The Boneferroni method of adjustment was used for a posteriori means testing of all data. Repeated measures univariate and multivariate

ANOVA tests were performed on all data to analyze treatment and time as a factor explaining variance and to determine if interactions between treatment and time occurred.

Linear regression analysis was used to consider the relationships between dependent variables such as elevation versus 1) *J. roemerianus* biomass, 2) standing dead vegetation biomass, 3) percent cover of atypical vegetative species, and 4) abundance of fishes; standing dead vegetation biomass versus percent cover of atypical vegetative species were also analyzed by linear regression. Plots were treated as the smallest, discrete experimental unit to avoid concerns of pseudoreplication.

RESULTS

The deposition of dredged material had immediate and lasting effects on marsh plots. All depositional treatments altered physical and biological parameters of plots to varying degrees, with some effects observed only seasonally.

Effects on Physical Parameters

Mean elevation of all plots prior to depositional treatment was 32.6 cm (± 2.01 SD) above sea level. Elevation of plots did not differ significantly prior to depositional treatment ($F=1.143$, $P= 0.371$). The actual depth of dredged material applied to experimental plots differed slightly from that targeted; mean plot depths were 0 cm (controls), 3.9 cm (±1.3 SD), 4.9 cm (±0.9 SD), and 9.7 cm (±0.6 SD), for 0-cm, 2-cm, 4-cm and 10-cm treatments groups, respectively. The deposition of dredged material altered the elevation of plots significantly for the duration of this study; ($F= 20.79$, $P<0.001$) at one year and ($F=15.48$, $P< 0.001$) at 12 years. Mean elevation of control plots was 31.5 cm (± 2.1 SD) and 33.4 cm (± 1.0 SD) in 1993 and 2004, respectively. In 1993, mean plot elevations were 36.6 cm (± 1.5 SD), 36.9 cm (± 1.8 SD) and 41.2 cm (± 1.9 SD) for the 2-cm, 4-cm and 10-cm treatments, respectively. In 2004, mean plot elevations were 35.8 cm (± 0.71 SD), 36.9 cm (± 1.2 SD) and 39.5 cm (± 1.9 SD) for the 2-cm, 4-cm and 10-cm treatments, respectively. Repeated measures ANOVA of elevation prior to deposition 1992 through 2004 showed that treatment ($F= 9.98$, $P= 0.001$) and time ($F=60.57$, $P< 0.001$) were significant factors affecting elevation and that a significant interaction between treatment and time ($F=12.50$, $P< 0.001$) occurred.

Repeated measures ANOVA of plot elevations in 1993 and 2004 showed that that treatment ($F=26.7$, $P<0.001$) was a significant factor explaining elevational differences, however time was not ($F=0.149$, $P=0.071$), no interaction between treatment and time occurred (Figure 5).

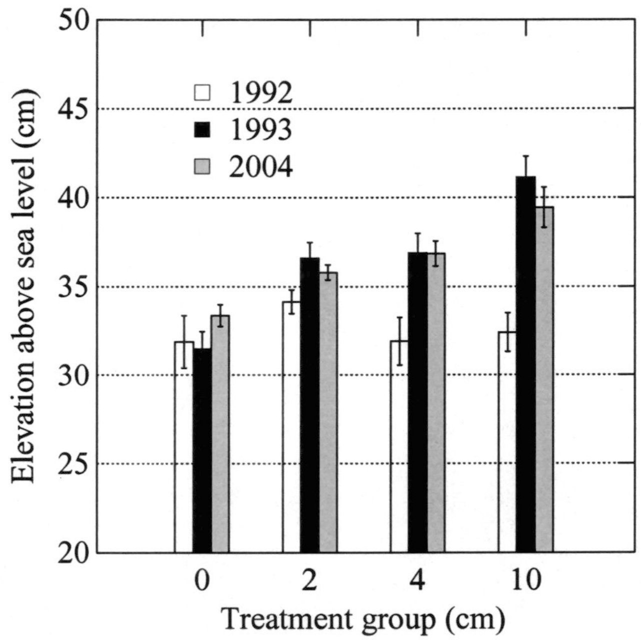

Figure 5. Elevation of plots by treatment in1992 (pre-treatment), 1993 and 2004 (bar denotes 1 SEM)

As expected, water level on the marsh varied greatly in the frequency and depth of inundation. Generally, a sustained period of little estuarine inundation occurred during June, July and August of each year, followed by a sustained period of estuarine flooding throughout September and October of each year (Figure 6). Mean water level for 26 September 1992 – 30 September 1993 and 26 September 2003 – 30 September 2004 was 8.2 cm (±6.24 SD) and -0.75 cm (±9.67 SD), respectively, with respect to the marsh surface. The difference in water levels between these two periods was significant ($F=225.0$, $P<0.001$).

In 1992, the dredged material deposited to treatment plots had a mean bulk density of 0.54 g/cm^3 (±0.06 SD) and mean percent organic matter of 14.38 % (±0.2.3 SD). The bulk density of the top 10 cm of marsh substrate was significantly altered in 1993 ($F= 32.76$, $P< 0.0001$) by the depositional treatments of dredged material. In 1993, bulk density values ranged from a mean of 0.181 g/m^3 (±0.008 SD) in the control plots to 0.621 g/m^3 (±0.067 SD) in the 10-cm treatment plots (Figure 7). The difference in bulk density among treatments was not significant in 2004 ($F= 2.66$, $P = 0.095$), with means ranging from 0.197 g/m^3 (±0.031 SD) in control plots to 0.437 g/m^3 (±0.255 SD) in 10-cm treatment plots. Repeated measures ANOVA of post-treatment samples (1993 and 2004) showed that both treatment ($F=12.74$, $P= 0.005$) and time ($F= 4.72$, $P = 0.051$) were significant factors explaining the variation in bulk density; however there was no interaction between treatment and time ($F= 1.95$, $P= 0.175$). Bulk density values of treatment groups were essentially inversely proportional to percent organic matter values and both parameters reflected the character and depth of the original dredged material applied (Figure 7).

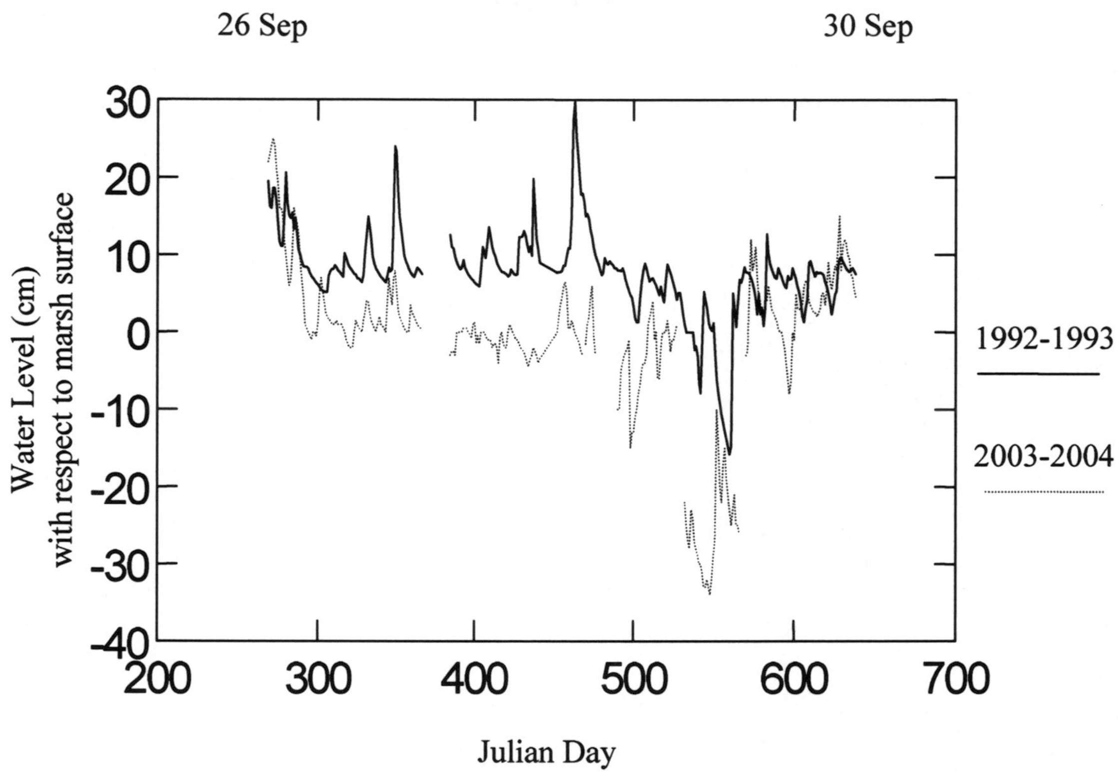

Figure 6. Hydrograph of Wysocking Bay Marsh study site 1992-1993 and 2003-2004

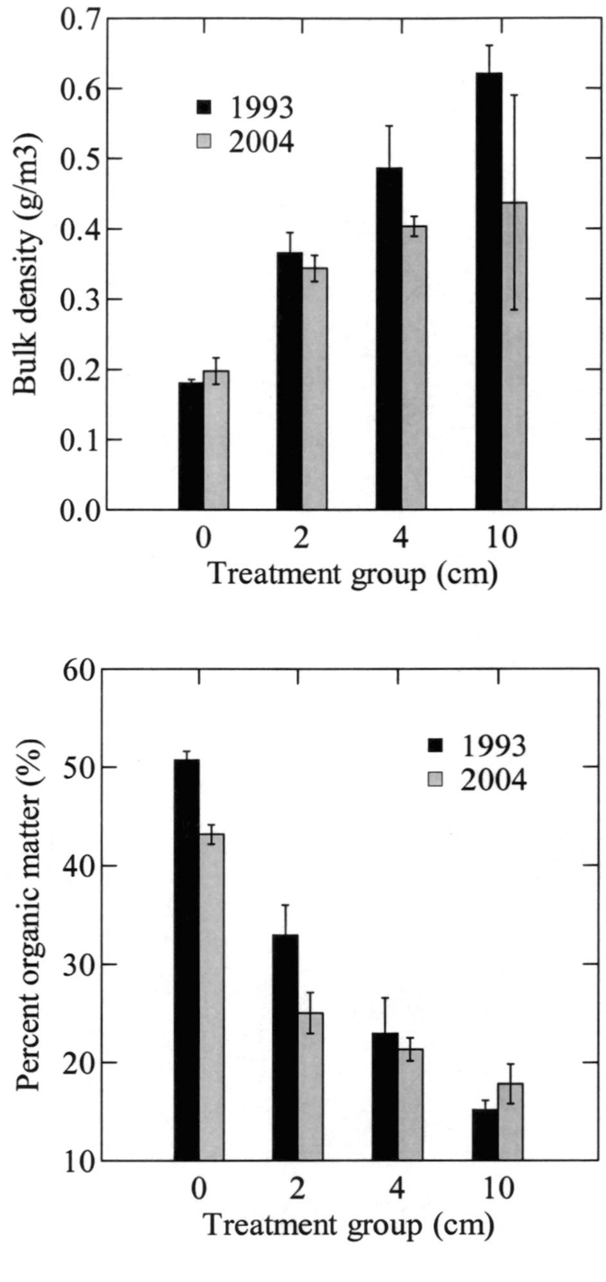

Figure 7. Mean bulk density and percent organic matter of top 10 cm of marsh substrate by treatment (bars indicate 1 SEM)

Percent organic matter of the top 10 cm of marsh substrate was significantly altered in 1993 ($F= 56.49$, $P<0.001$) and in 2004 ($F= 67.40$, $P< 0.001$) by the depositional treatments of dredged material. In 1993, mean percent organic matter values ranged from 51.7 % (±1.5 SD) in the control group to 15.2 % (±1.6 SD) in the 10-cm treatment group. In 2004, these values ranged from 43.2 % (±1.6 SD) in the control group to 17.8 % (±3.4 SD) in the 10-cm treatment group. Repeated measures ANOVA of post-treatment samples (1993-2004) showed that for percent organic matter, treatment ($F= 85.65$, $P< 0.001$) and time ($F= 13.29$, $P= 0.003$) explained variation among treatment groups and an interaction between treatment and time ($F= 6.48$, $P= 0.007$) occurred.

Effects on Biological Parameters

Vegetation

Juncus roemerianus biomass averaged 34.17 g/0.04m^2 (±3.08 SEM) among all study plots prior to dredged-material treatments (September 1992) and the control group remained roughly at this level throughout this study (Figure 9). In September 1992, *J. roemerianus* density averaged 36.63 shoots/0.04m^2 (±3.10 SEM) and the control group ranged from 24.0 – 62.0 shoots/0.04m^2 in other years. *Juncus roemerianus* biomass and shoot density decreased significantly ($F= 3.78$, $P= 0.04$ and $F= 5.76$, $P= 0.01$, respectively) among dredged-material treatments during the first year (1993) of this study. In 1993, *J. roemerianus* biomass averaged 24.58 g/0.04m^2 (±5.01 SEM) in 2-cm treatments, 30.57 g/0.04m^2 (±6.10 SEM) in 4-cm treatments and 9.00 g/0.04m^2 (±5.06 SEM) in 10-cm treatments (Figure 8). In 1993, *J. roemerianus* shoot density means were 44.25 shoots/0.04m^2 (±8.72 SEM) for the control group, 23.0 shoots/0.04m^2 (±4.42 SEM) for 2-cm the treatment group, 29.0 shoots/0.04m^2 (± 5.20 SEM) for the 4-cm treatment group and 8.75 shoots/0.04m^2 (±5.25 SEM) for the 10-cm treatment group (Figure 9). There were significant differences in *J. roemerianus* biomass among treatments overall in 1994 (F=11.14, P = 0.001), 1995 (F=7.52, P= 0.004) and 1996 (F= 3.98, P= 0.035), driven by low biomass values in the 10-cm treatment plots (Figure 8). Pair-wise comparisons (with Bonferroni adjustment) of *J. roemerianus* biomass in the 2-cm and 4-cm depth treatment groups showed that these treatments were not significantly different from controls in any year after dredged material treatments were applied (Table 1). *Juncus roemerianus* biomass in the 10-cm depth treatment group continued to decline for the first 3 years post-treatment, and then demonstrated some recovery by the fourth year; this group had significantly less biomass than the control group in Bonferroni-corrected pair-wise comparisons in 1993-1996 (Table 1). Plot vegetation was again sampled in 2003 and 2004 (years 11 and 12) wherein an overall decrease was observed (relative to 1993-1996) in *J. roemerianus* biomass among treatment groups (including controls) except in the 10-cm depth treatment group, in which biomass had remained relatively stable (Figure 8). In 2003, differences in *J. roemerianus* biomass among treatments were not significant (F= 2.65, P= 0.096) and were marginally significant for *J. roemerianus* shoot density (F= 3.41, P= 0.053). In 2003, *J. roemerianus* shoot density means were 29.5 shoots/0.04m^2 (±2.90 SEM) for the control group, 11.5 shoots/0.04m^2 (±8.59 SEM) for 2-cm the treatment group, 9.0 shoots/0.04m^2 (±1.58 SEM) for the 4-cm treatment group and 9.50 shoots/0.04m^2 (±5.30 SEM) for the 10-cm treatment group (Figure 9). In 2004, differences in among treatments were not significant for *J. roemerianus* biomass

(F= 1.86, P= 0.19) or *J. roemerianus* shoot density (F= 1.46, P= 0.28) and no group differed significantly from controls. In 2004, *J. roemerianus* shoot density means were 39.5 shoots/0.04m^2 (±2.63 SEM) for the control group, 19.5 shoots/0.04m^2 (±14.9 SEM) for 2-cm the treatment group, 13.0 shoots/0.04m^2 (±6.42 SEM) for the 4-cm treatment group and 18.75 shoots/0.04m^2 (±9.87 SEM) for the 10-cm treatment group (Figure 9). Repeated measures ANOVA (including pre-treatment plot means) showed that treatment (F= 7.12, P= 0.005) and time (F= 4.51, P= 0.001) explained variance in *J. roemerianus* biomass and an interaction between treatment and time occurred (F= 2.07, P=0.02). The repeated measures ANOVA (post-treatment only: 1993-2004) showed that treatment (F= 10.15, P = 0.001) and time (F= 2.98, P = 0.018) still explained variance in *J. roemerianus* biomass and no interaction between treatment and time occurred (F= 1.17, P= 0.11).

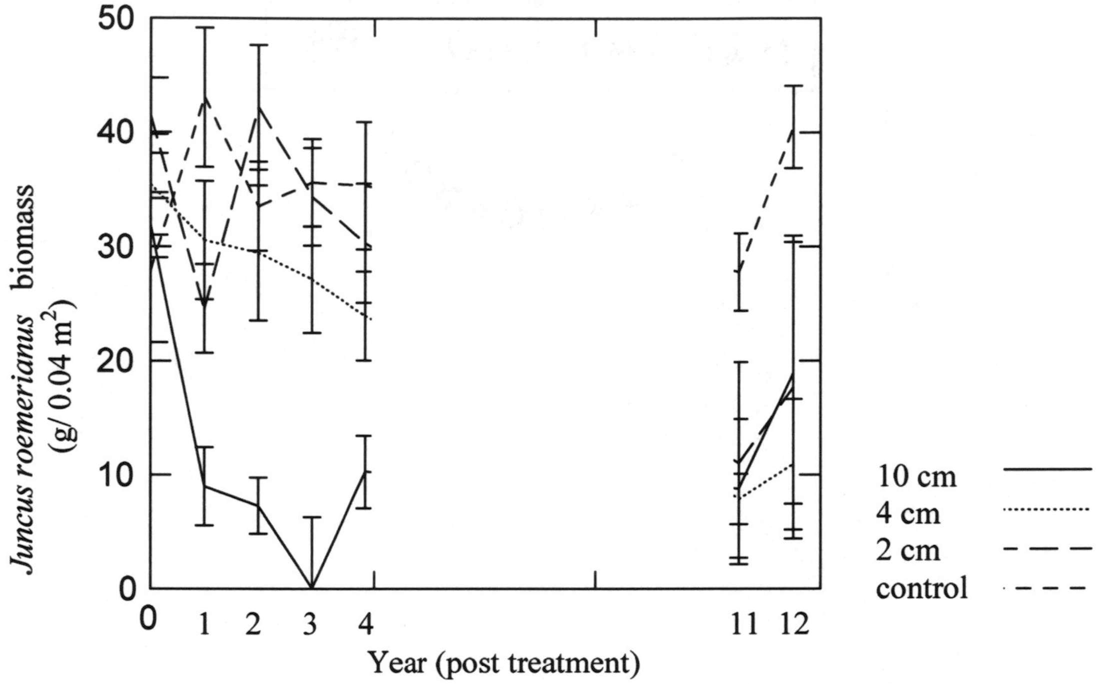

Figure 8. *Juncus roemerianus* biomass (g/0.04m^2) of treatment groups over a 12-year period [1992 (year 0) - 2004 (year 12)] after receiving a single treatment of 0, 2, 4 or 10 cm dredged material (bars indicate 1 SEM)

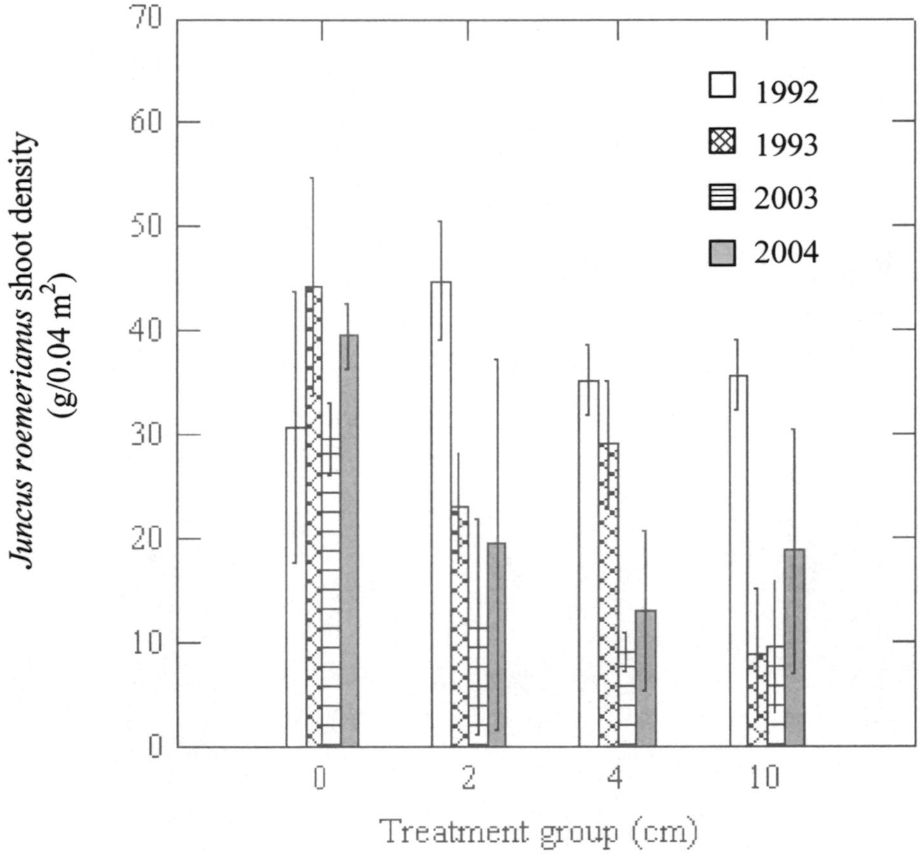

Figure 9. *Juncus roemerianus* shoot density (g/0.04m^2) of treatment groups over a 12-year period [1992 (year 0)-2004 (year 12)] after receiving a single treatment of 0, 2, 4 or 10 cm dredged material (bars indicate 1 SEM)

Table 1. Matrices of ANOVA probability values for differences occurring among *Juncus roemerianus* biomasses, by treatment-groups, in pair-wise comparisons (adjusted using Bonferroni Multiple Comparison Tests) by sampling year. Group 1 = control group, Group 2 = 2-cm treatment group, Group 3 = 4-cm treatment group and Group 4 = 10-cm treatment group.

1993 (Year 1 post-treatment)

	1	2	3	4
1	1.000			
2	0.587	1.000		
3	1.000	1.000	1.000	
4	**0.037**	0.937	0.348	1.000

1994 (Year 2 post-treatment)

	1	2	3	4
1	1.000			
2	1.000	1.000		
3	1.000	0.395	1.000	
4	**0.008**	**0.001**	**0.025**	1.000

1995 (Year 3 post-treatment)

	1	2	3	4
1	1.000			
2	1.000	1.000		
3	1.000	1.000	1.000	
4	**0.008**	**0.010**	0.066	1.000

1996 (Year 4 post-treatment)

	1	2	3	4
1	1.000			
2	1.000	1.000		
3	1.000	1.000	1.000	
4	**0.044**	0.118	0.552	1.000

Linear regression analysis indicated that elevation accounted for 29 % (P=0.017) and 14.9 % (P=0.083) of the variance in *J. roemerianus* biomass in 2003 and 2004, respectively; as well as 89.6 % (P=0.29) and 0.4 % of percent dead vegetation (by weight in $0.04m^2$ replicates) in 2003 and 2004, respectively.

In 2004, a significant (KW=7.98, P= 0.046) percent cover (based on visual assessment) of atypical vegetative species, seldom found within *J. roemerianus*-dominated marshes, were observed in the 10-cm treatment plots; atypical species observed were: *Amaranthus cannabinus, Pluchea purpurascens, Iva fructens, Solidago sempervirens, Scirpus americanus, Eleocharis flavescens, Typha angustifolia* and *Salicornia europaea*. These species typically occur in slightly higher elevation areas and sporadically occur among or associated with *J. roemerianus*-dominated marshes. Species that commonly occur within *J. roemerianus*-dominated marshes, *Spartina patens* and *Distichlis spicata*, were observed in plots of all treatment groups. Linear regression analysis revealed that elevation explained 26.1% (P=0.01) of the variance in observed percent cover of atypical species in 2004. When observed percent cover values were arcsine transformed, elevation still explained 36.9% (P=0.01) of the variance among plots; however these values did not significantly differ by treatment (KW=6.39, P=0.09). The abundance of 2003 dead vegetation explained only 29.3% (P=0.16) of the difference in atypical species observed in 2004.

The assessment of various stages of *J. roemerianus* condition revealed that a significantly greater percent of *J. roemerianus* leaves were senescent in the 10-cm treatment group (F=3.86, P= 0.038) in September 2004 as compared to that of controls.

Linear regression analysis showed that elevation explained only 35.7 % ($P=0.058$) of the difference observed in *J. roemerianus* senescence. No other trends were observed in the assessment of *J. roemerianus* condition among treatments or blocks.

Fauna

Pit traps. Comparisons of the abundance of all fauna (fishes and invertebrates) collected in pit traps, summed across all sampling dates within each plot (summed biomass of all fauna and summed density of all fauna in Table 2), yielded no significant differences among treatment means for the first post-treatment sampling period (1992-1993) (summed biomass of all fauna, KW= 3.29, P= 0.35; summed density of all fauna, KW= 1.53, P= 0.68) (Table 2). In year 12 (2003-2004), summed density of all fauna differed significantly (KW= 12.81, P= 0.005) among treatments, being inversely proportional to dredged material depth (Table 2). Control plots in year 12 had significantly higher mean density (180 individuals/two pit traps) than all other treatments (ANOVA, Bonferroni multiple comparison test, $P< 0.0002$, Table 2). Summed biomass of all fauna in year 12 differed (KW= 7.57, P= 0.056) among treatments, with the greatest biomass in controls, and significantly less in the 2-cm treatment (but not significantly less in the 4-cm and 10-cm treatments, Bonferroni multiple comparison test, Table 2). Repeated measures ANOVA of both sampling periods (post-treatment) showed that treatment was a significant factor in the differences observed in summed biomass of all fauna (F= 3.56, P= 0.047), but not in summed density of all fauna (F= 1.68, P= 0.223). There was a general increase in the summed biomass of all fauna collected among plots during year 12 as compared to year 1 of this study; the lower biomass values collected in year 1 may have been a consequence of the initial disturbance caused by the deposition of dredged material.

Table 2. Summary of abundance of total fauna collected in pit traps at Wysocking Bay Marsh, NC in two post-treatment periods (1992-1993 and 2003-2004). Mean [±1 standard error (SEM)] biomasses (g wet biomass/two traps) and densities (number of individuals/two traps) of fishes, invertebrates and all fauna by dredged materials treatment group and sample period. Means within each sampling period for each variable that do not differ significantly in Bonferroni Multiple Comparison Tests are indicated by the same letter superscripts (a, b, etc.).

Sample Period	Treatment group (cm dredged material)	Summed fishes biomass (g/plot) (SE)	Summed fishes density (no./plot) (SE)	Summed invertebrate biomass (g/plot) (SE)	Summed invertebrate density (no./plot) (SE)	Summed biomass of all fauna (g/plot) (SE)	Summed density of all fauna (no./plot) (SE)
1992-1993	0	2.5^{a} (1.0)	49^{a} (11)	22^{a} (4.5)	130^{a} (11)	25^{a} (5.5)	179^{a} (13)
"	2	0.25^{b} (0.1)	10^{b} (4.0)	27^{a} (13)	152^{a} (3)	27^{a} (13)	162^{a} (35)
"	4	0.25^{b} (0.1)	4.5^{b} (1.7)	29^{a} (20)	217^{a} (6)	29^{a} (20)	222^{a} (62)
"	10	0.003^{b} (0.003)	0.25^{b} (0.25)	16^{a} (11.0)	225^{a} (37)	16^{a} (11)	225^{a} (36)
2003-2004	0	11^{a} (1.7)	40^{a} (9.9)	215^{a} (41)	140^{a} (23)	226^{a} (39)	180^{a} (22)
"	2	4.4^{b} (0.5)	22^{a} (2.5)	108^{b} (12)	43^{b} (11)	112^{b} (12)	65^{b} (12)
"	4	1.5^{b} (0.21)	8.5^{b} (2.1)	159^{a} (18)	37^{b} (4.0)	161^{a} (18)	45^{b} (2.7)
"	10	0.59^{b} (0.11)	4.8^{b} (0.85)	153^{a} (12)	33^{b} (1.7)	153^{a} (12)	38^{b} (2.3)

When only the data for summed fish were compared, the difference among means of treatment groups was significant for both summed fish biomass and summed fish density in year 1 (KW= 12.31, P= 0.006 and KW= 12.96, P= 0.005, respectively) and in year 12 (KW= 14.12, P= 0.003 and KW= 12.23, P= 0.007, respectively); both measures were inversely proportional to dredged material depth during both sampling periods (Figure 9). In year 1, fish biomass and density in all depositional treatment groups differed significantly from controls (Table 2). In year 12, this pattern was repeated with the exception of the 2-cm treatment group not differing significantly from controls (Table 2). Elevation could explain 58.7% (P<0.001) and 41.0% (P=0.004) of the variation for summed fish biomass and density, respectively in the 12 year sampling period. Repeated measures ANOVA of both sampling periods showed that treatment was a highly significant factor in the differences observed in summed fish biomass (F= 18.82, P< 0.001) and in summed fish density (F= 13.17, P< 0.001) and the abundance of fishes did not differ significantly between sampling periods.

Summed invertebrate biomass (KW= 3.19, P= 0.362) and density (KW= 4.46, P= 0.216) did not differ significantly in year one. In year 12, summed invertebrate biomass did not differ significantly (KW= 6.86, P= 0.077), however summed invertebrate density did differ significantly (KW= 8.99, P= 0.025). Repeated measures ANOVA of summed invertebrate data showed that neither treatment nor time explained the difference in invertebrate density.

When faunal biomasses and density data of individual sample dates were analyzed, significant differences between treatments occurred among fishes in May, July and September of 1993, October and November of 2003 and January, September, and November of 2004 (Table 4). And of these dates, fish biomass and density values in October 2003 and November 2004 were not proportional to depth of dredged material (Table 3). Only in January 2004, was there also a significant difference among means of invertebrate biomass (KW= 8.14, P= 0.043). Fish species recovered from pit-trap sampling were: *Fundulus heteroclitus, F. confluentus, F. luciae, Gambusia affinis*, and *Cyprinodon variegatus*. Fish were most often recovered in the juvenile stage. The marsh fiddler crab, *Uca minax*, was the dominant invertebrate organism found at this site; amphipods, isopods, arachnids and insects were also present in pit trap samples.

Simulated aquatic microhabitats. The contents of the simulated aquatic microhabitats (SAMs) were collected on the same 2003-2004 dates on which pit trap samples were collected, unless all SAMs were empty. The individuals recovered from two SAMs were summed for each plot. Only during January, March, August and November 2004 did the SAMS collect an adequate sample size (n > 10 individuals) for reliable statistical comparison. Summed faunal density (all dates included) differed marginally among treatment groups (KW= 7.69, P= 0.053); this difference was driven by a large number of organisms collected from two control plots only in March 2004. Means (all dates included) among treatment groups were 191 (±44.6 SEM) individuals in controls, 55.5 (±40.1 SEM) individuals in the 2-cm treatment group, 31.5 (±16.4 SEM) individuals in the 4-cm treatment group and 14.3 (±4.11 SEM) individuals in the 10-cm treatment group. Organisms found in the SAMs were *Gammarus spp*., other amphipods, *Eulimnadia sp*., cyclopoid copepods, isopods, gastropods, polychaetes, arachnids, insects and larval *Rana sp*.

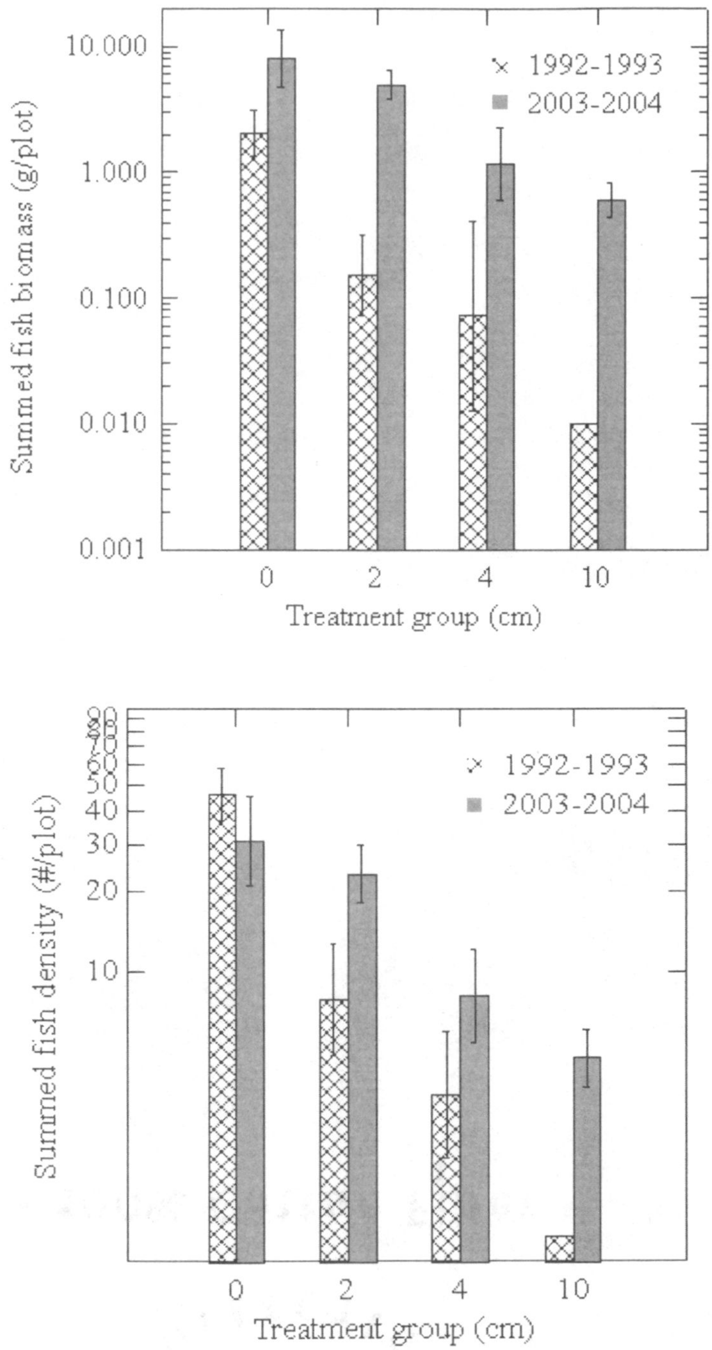

Figure 10. Summed biomass and density of fishes by treatment group during 1992-1993 (post-treatment) and 2003-2004 sampling periods (bars indicate 1 SEM)

Table 3. Summary of faunal abundance collected in pit traps at Wysocking Bay Marsh, NC by sampling dates during two periods (1992-1993 and 2003-2004). Mean biomasses (g wet biomass/two traps) and densities (number of individuals/two traps) of fishes, invertebrates and all fauna (fishes and invertebrates) by dredged materials treatment group and sample date (* = sample contained individuals too small to weigh individually).

Sample Date	Treatment group (cm dredged material)	Fishes biomass (g/plot)	Fishes density (no./plot)	Invertebrate biomass (g/plot)	Invertebrate density (no./plot)	All fauna biomass (g/plot)	All fauna density (no./plot)
12 Sep 1992	0	0.85	1.4	14*	15	14*	16
	2	0.89	0.29	5.8*	13	6.7*	13
	4	1.4	0.24	5.6*	35	7.0*	35
	10	0.99	2.3	5.2*	8.2	6.2*	11
18 Sep 1992	0	1.4	9.8	14*	19	16*	29
	2	0.29	4.5	15*	43	13*	47
	4	0.24	5.3	13*	28	35*	33
	10	2.4	13	35*	16	11*	29
6 Nov 1992	0	0.51	2.0	0.79*	31	1.3*	33
	2	0.11	2.8	4.9*	16	5.1*	19
	4	0.02	1.8	0.85*	25	0.87*	26
	10	0.003	0.25	0*	33	0.003*	33
16 Jan 1993	0	0	0	0.85*	5.5	0.85*	5.5
	2	0	0	0.95*	5.3	0.95*	5.3
	4	0	0	0*	11	0*	11
	10	0	0	0.08*	6.5	0.08*	6.5
8 Mar 1993	0	0	0.25*	0*	39	0*	40
	2	0.003	0.25	0*	37	0.003*	37
	4	0	0	0	29	0*	29
	10	0	0	0	36	0*	36

Table 3 (continued).

Sample Date	Treatment group (cm dredged material)	Fishes biomass (g/plot)	Fishes density (no./plot)	Invertebrate biomass (g/plot)	Invertebrate density (no./plot)	All fauna biomass (g/plot)	All fauna density (no./plot)
6 May 1993	0	0.32	2.3	8.9*	23	9.2*	25
	2	0	0	6.4*	55	6.4*	55
	4	0	0	4.6*	115	4.6*	115
	10	0	0	5.1*	82	5.1*	82
29 Jul 1993	0	0.59	15	9.1*	8.8	9.7*	23
	2	0.10	4.5	6.0*	20	6.1*	2
	4	0.23	2.8	5.8*	20	6.0*	23
	10	0	0	2.1*	42	2.1*	42
26 Aug 1993	0	0.54	8.3	0*	9.3	0.54*	18
	2	0.02	0.50	5.8*	2.5	5.8*	3.0
	4	0	0	15*	5.5	15*	5.5
	10	0	0	5.5*	20	5.5*	20
16 Sep 1993	0	0.53	22	2.8*	14	3.3*	36
	2	0.02	2	2.8*	16	2.8*	18
	4	0	0	3.3*	13	3.3*	13
	10	0	0	2.9*	6	2.9*	6
18 Oct 2003	0	1.4	2.3	30	2.8	31	7.3
	2	0.71	5.5	17	2.3	18	7.8
	4	0.21	2.5	15	2.5	15	5.0
	10	0.27	2.5	22	4.8	22	7.3
15 Nov 2003	0	7.1	25	0.19	2.3	7.3	27
	2	0.76	5.3	0.10	1.3	0.87	6.5
	4	0.39	0.75	0	0	0.39	0.75
	10	0	0	0	0	0	0

Table 3 (continued).

Sample Date	Treatment group (cm dredged material)	Fishes biomass (g/plot)	Fishes density (no./plot)	Invertebrate biomass (g/plot)	Invertebrate density (no./plot)	All fauna biomass (g/plot)	All fauna density (no./plot)
30 Jan 2004	0	1.9	9.8	2.9	9.8	4.8	45
	2	0.91	4.3	0.44	4.3	1.3	9.3
	4	0.06	0.25	0.08	0.25	0.14	2.3
	10	0	0	0	0	0	0.25
12 Mar 2004	0	0.65	1.3	2.5	59	3.2	61
	2	0	0	0.13	4.5	0.13	4.5
	4	0	0	0.02	1.8	0.02	1.8
	10	0.05	0.25	0.12	2.0	0.17	2.3
21 May 2004	0	0	0	13	3.8	13	3.8
	2	0	0	5.8	1	5.8	1
	4	0	0	21	3.8	21	3.8
	10	0	0	13	3	13	3
5 Aug 2004	0	0	0	94	19	94	19
	2	0	0	30	6	30	6
	4	0	0	56	11	56	11
	10	0	0	70	13	70	13
12 Aug 2004	0	0	0	28	5.6	28	5.6
	2	0	0	15	3.5	15	3.5
	4	0	0	24	5.5	24	5.5
	10	0	0	14	3.0	14	3.0
20 Aug 2004	0	0	0	15	3.3	15	3.3
	2	0.07	1.5	14	3.0	14	4.5
	4	0	0	25	5.0	25	5.0
	10	0	0	19	3.8	19	3.8

Table 3 (continued)

Sample Date	Treatment group (cm dredged material)	Fishes biomass (g/plot)	Fishes density (no./plot)	Invertebrate biomass (g/plot)	Invertebrate density (no./plot)	All fauna biomass (g/plot)	All fauna density (no./plot)
18 Sep 2004	0	0	0	14	2.5	14	2.5
	2	0	0	4.1	0.88	4.1	0.88
	4	0.28	1.5	5.9	1.3	6.2	2.8
	10	0	0	6.6	1.5	6.6	1.5
24 Sep 2004	0	0	0	11	2.3	11	2.3
	2	0	0	15	3.0	15	3.0
	4	39	45	13	2.3	13	2.8
	10	0	0	7.1	1.5	7.1	1.5
2 Nov 2004	0	0.11	0.25	4.2	4.0	4.3	4.3
	2	1.9	4.5	5.5	13	7.4	18
	4	0.56	3.0	0.10	1.5	0.66	4.5
	10	0.27	2.0	1.2	0.25	1.5	2.3

Table 4. Summary of treatment effect among four dredged-material treatments (Kruskal-Wallis one-way ANOVA) in biomass and density of fishes at Wysocking Bay Marsh, NC by sample date during two sampling periods (1992-1993 and 2003-2004)

Sample date	Biomass of fishes		Density of fishes	
	Kruskal-Wallis statistic	Probability	Kruskal-Wallis statistic	Probability
12 September 1992 (pre-treatment)	2.14	0.543	2.65	0.449
18 September 1992 (pre-treatment)	0.898	0.826	1.19	0.754
6 November 1992	4.65	0.200	4.65	0.200
16 January 1993	0	1.00	0	1.00
8 March 1993	2.14	0.543	2.14	0.543
6 May 1993	10.3	**0.017**	10.3	**0.017**
29 July 1993	11.0	**0.012**	11.0	**0.012**
26 August 1993	3.00	0.392	2.15	0.542
16 September 1993	12.9	**0.005**	12.9	**0.005**

Table 4 (continued).

Sample date	Biomass of fishes		Density of fishes	
	Kruskal-Wallis statistic	Probability	Kruskal-Wallis statistic	Probability
18 October 2003	9.65	**0.022**	4.82	0.186
15 November 2003	12.5	**0.006**	13.5	**0.004**
30 January 2004	9.67	**0.022**	9.70	**0.021**
12 March 2004	2.15	0.524	2.15	0.542
21 May 2004	0	1.0	0	1.0
5 August 2004	0	1.0	0	1.0
12 August 2004	0	1.0	0	1.0
20 August 2004	3.0	0.392	3.0	0.392
18 September 2004	10.3	**0.017**	10.3	**0.017**
24 September 2004	6.40	0.094	3.86	0.277
2 November 2004	9.48	**0.024**	8.47	**0.043**

DISCUSSION

The application of thin-layer dredged material onto marsh habitat was at first considered as an economic and perhaps environmentally-acceptable means of dredged material disposal; in recent years this technique has been proposed as a management strategy to facilitate the persistence of eroding coastal marsh habitat. This experimental study indicated that a single, thin-layer, dredged-material application alters marsh habitat with significant and long-term impacts on marsh elevation and substrate characteristics; effects on marsh vegetation were initially significant then became more subtle over time, and impacts on faunal abundance (especially fish) remained significant for up to twelve years.

Short-Term Effects

Immediate effects resulting from the application of thin-layer dredged material were observed among all depositional treatment groups in this irregularly-flooded marsh habitat. The elevation and substrate character of marsh habitat were significantly altered in 2, 4 and 10 cm thin-layer dredged material treatments.

Initially, marsh vegetation was physically crushed by the application of dredged material, with the mean biomass of *J. roemerianus* inversely proportional to the depth of material applied after 1 year, although the mean in biomass was only significantly lower than controls for the 10-cm treatment group. *Juncus roemerianus* biomass in the 2-cm and 4-cm treatment groups did not differ significantly from the control group during this study and indicated that these thicknesses of dredged material applied to *J. roemerianus*-dominated marsh are adequately thin to permit recovery of marsh vegetation after one year. These *J. roemerianus* biomass patterns among 0, 2, 4, and 10-cm treatment groups, concur with Wilbur's (1993) synopsis of marsh vegetation recovery.

Differences in plot elevation altered flooding regimes among treatment groups which resulted in aquatic organisms having restricted access to higher elevation plots when flooding was minimal. The greatest differences in faunal abundance among treatments occurred with fishes and these differences were most often significant during the fall and spring of the year (Table 2). The essential role of coastal marsh habitat has been established for many estuarine resident fishes (Lotrich 1975; Kneib and Wagner 1994; Kneib 1997; Komarow et al. 1999). *Fundulus* species have been found to be among the most abundant resident fishes of salt marsh habitats found along the Atlantic, Gulf and Pacific coasts of the United States (Able and Fahay 1998, Kneib 1997, Kneib 2000, Talley 2000). My findings concur, as *Fundulus* species comprised 89 % of all fishes recovered at this study site with *Fundulus heteroclitus* accounting for 77 % of the total faunal density and 89% of all fishes. The key role of *Fundulus* species in the trophic transfer of primary production from marsh habitat into estuarine waters has been well established (Kneib and Stiven 1978; Kneib and Wagner 1994, Kneib 2000, Smith et al. 2000). At Wysocking Bay, the greatest abundance of fishes occurred in the lower elevation plots; therefore plots of higher elevation were utilized less frequently than those of lower elevation, inferring some loss in ecological function. Teo and Able (2003) also found that elevation of marsh habitat was the main limiting factor in the abundance of *F. heteroclitus* on a restored diurnally-tidal marsh habitat in New Jersey. Along the western Pamlico Sound, the irregularly-flooded marsh habitat has a pulsed inundation pattern of erratic depths and there were period when all experimental plots were submerged for weeks at a time. Therefore, the time period over which marsh primary production is transferred, may be altered by the addition of thin-layer dredged material (higher elevations receive less frequent inundation); however, the quantity of energy transferred from this habitat may not be diminished, providing that the vegetation and benthic communities are not modified. Additional research would be required to determine such intricacies of trophic transfer in irregularly-flooded marshes.

Long-Term Effects

Upon examination of the effects of various thicknesses of thin-layer dredged material deposition onto marsh habitat after 12 years, I found that elevation in the 4-cm and 10-cm treatment groups, the substrate percent organic matter among all depositional treatments groups, and the seasonal abundance of fishes remained significantly altered from original thin-layer treatments. The long-term effects of this depositional disturbance on other biological parameters of marsh habitat were less apparent.

During 1994 - 1996, *J. roemerianus* biomass remained significantly lower in the 10-cm treatment group when compared to controls. An overall reduction in *J. roemerianus* biomass and shoot density was observed in 2003 (year 11) and although differences among treatment groups were not statistically significant, *J. roemerianus* biomass within the depositional treatment groups was clearly depressed when compared to the control group (Figure 8). The cause of this general reduction of vegetation biomass is unknown, however several major hurricanes were noted to have occurred in North Carolina during the 1996 – 2003 (no sampling) time period; it is assumed that these storms or some other stress occurred then to this ecosystem. It appears that all thicknesses of the thin-layer dredged material depositional disturbance rendered marsh vegetation less resilient to future stresses.

I was surprised to find a significant quantity of atypical vegetative species within the plots of the 10-cm treatment group during the 2004 growing season; this community shift was not previously observed at this site. Although some of the atypical species were found in each of the treatment groups, these species comprised a greater percent cover in the 10-cm treatment group; elevation was able to explain this difference. The cause of this sudden community shift in vegetation is unclear, however I note that Hurricane Isabelle made landfall in the general area of this study site (after 2003 harvest) and water levels were sufficiently high as to import seeds from upland locations. Interestingly, *J. roemerianus* biomass increased across all treatment groups in 2004, even though a significant proportion of *J. roemerianus* leaves in the 10-cm treatment group were senescent.

Utilization of marsh habitat by fishes remained significantly different among treatments when plot biomass and density was summed for each year-long sampling period; this was driven by differences that occurred in spring and fall of each year (Tables 3 and 4). This observation was likely due to remaining significant differences in elevation among treatment groups; most often, the control group (lowest elevations) retained the greatest fish biomass and abundance. During periods of deeper inundation, usually associated with storms and seasonal wind events (sometimes lasting for several weeks), higher elevation areas may have functioned better as refuge habitat for small (especially juvenile) organisms because larger aquatic predators were unable to access shallow areas. The premise of depth-dependent refuge habitats may explain occasions when differences among faunal biomass was significant at Wysocking Bay, however did not exhibit a monotonic function between depth of dredged material and abundance (see summed invertebrate and all fauna biomass 2003-2004 in Table 2). Kneib and Wagner (1994) found that nekton utilization of a diurnally-flooded marsh habitat varied with inundation depth and stage and by species; mobility of individuals was also believed to have affected faunal distribution over the flooded marsh. In brief, predator and prey species utilized different depth zones of the marsh habitat throughout the tidal cycle (Kneib and Wagner 1994). Therefore, varying marsh habitat topography may be as critical as actual elevation in considering the effects of thin-layer dredged material deposition on the ecosystem function of marsh habitat, especially those that are irregularly-flooded.

Management Implications

In recent years, the application of dredged material using thin-layer technique has been proposed for the augmentation of eroding marshes. It is thought that a direct addition of sediment could: 1) contribute to marsh accretion where natural sediment supply has been reduced, 2) supplement sediment to marshes that are eroding due to exposure of amplified wave-energy and 3) augment marsh habitat where relative elevation is decreasing due to subsidence and/or sea-level rise. The positive attributes of sediment addition require balance with concerns of impaired habitat function caused by a depositional disturbance and altered abiotic parameters of marsh habitat. On a Louisiana barrier island, Courtemanche (1996) observed a 98% recovery rate among *Spartina alterniflora* in a 10-cm-deep deposition zone of an overwash fan caused by Hurricane Andrew. Can a thin-layer of sediment applied mechanically to marsh habitat mimic a natural sediment depositional disturbance as from a storm?

Several studies have suggested that a thin-layer sediment deposition can result in increased plant biomass and density. Ford et al. (1999) reported a three-fold increase in *S. alterniflora* stem densities within one year of a 2.3-cm application of dredged material on a Louisiana deteriorating (where relative sea level rise outpaces accretion processes) marsh habitat. Here, these researchers found no significant difference in the bulk density (top 5 cm) and an increase in the percent organic matter (top 25 cm) of marsh substrate which had been treated with 2.3 cm of river sand one year earlier; this rise in organic matter was suggested to be the result of increased below-ground vegetative biomass (Ford et al. 1999). These data contrast with my findings at Wysocking Bay, a non-deteriorating marsh habitat where vegetative biomass decreased and substrate organic content was significantly lowered in response to all depositional treatments. Leonard et al. (2002) found that *S. alterniflora* stem densities increased somewhat proportionally with depth of dredged material (0 – 10 cm) when the homogenized, medium to coarse-grained material was applied along the periphery of deteriorated and non-deteriorated, diurnally tidal marsh habitat, on Masonboro Island, North Carolina. Increased vegetation density was found to cause increased drag on flood waters resulting in increased sedimentation upon the marsh surface at Masonboro Island and at other sites (Leonard and Luther 1995; Leonard et al. 2002; Mudd et al. 2004). Thus, the addition of an appropriate thickness of dredged material onto marsh surface may facilitate a positive feedback loop of marsh accretion (Leonard and Luther 1995; Mudd et al. 2004). Relationships between dredged material depth and stem abundance were different at Wysocking Bay; I found that *J. roemerianus* biomass was inversely proportional to the depth of dredged material applied. These biomass differences became more subtle over time, however all depths of dredged material had a negative impact on *J. roemerianus* biomass even after 12 years when a shift in community composition was also observed in plots that had received 10 cm of dredged material.

My results contrast with results of studies in which thin-layer amendments of 2 - 10 cm depth were applied to eroding marsh habitats. The marsh habitat in these studies differed from Wysocking Bay in several ways: 1) *Spartina alterniflora* was dominant in these marshes, 2) the marsh platform was deteriorating and 3) these marshes experienced regular, diurnal flooding regime. It is reasonable that a disturbance would elicit differing responses between marsh habitats with a regular, diurnal-tidal, high-energy flooding regime from those with an irregularly-flooded, low-energy environment and a meteorologically-driven flooding regime. The above findings highlight the importance of the flooding regime (including velocity of inundation) in addition to relative elevation when considering an appropriate depth of dredged material that may be applied to marsh habitat. The physiology of dominant marsh vegetation will also factor into resilience of these ecosystems. Rozas (1995) described the complexities of faunal utilization of marsh habitat based on hydroperiod and remarked that hydroperiod also alters marsh vegetation, which in turn, further impacts faunal utilization of marsh habitat. The tolerance ranges of flora and fauna may also vary in each marsh habitat type; therefore great care must be taken in applying experimental results across various marsh habitat types.

The physical and chemical characteristics of potential dredged material and its compatibility with existing marsh substrate will likely be a factor in habitat response and recovery. Due to variations in material character and associated nutrients, dredged material additions have been reported as having a "fertilizer effect" on marsh vegetation (Reimold et al.1978; DeLaune et al. 1990; Pezeshki et al. 1992; Courtemanche 1996). Nutrient levels were not measured at Wysocking Bay; however the mineral characteristics of the material applied remained evident for the 12-year duration of this study. The character (especially grain size) of dredged material has been considered as an important factor in the biological recovery of renourished beaches (NAP 1995; Finkl et al. 1997; Milton et al. 1997; Steintz et al. 1998; Peterson et al. 2000a; Peterson et al. 2000b) and is likely to impact marsh ecosystems in a similar manner.

Coastal marshes have been established as essential ecosystems within estuarine and coastal ecosystems. Their social, economic and ecological role is becoming even more self-evident as their surface area decreases. Even though marsh communities have evolved to be resilient to disturbances, expanses of coastal marsh habitat are losing elevation relative to sea level on a global scale. Thin-layer dredged material application has benefited marsh biota where it decreases environmental stressors (Ford et al. 1999; Leonard et al. 2002). To date, empirical evidence suggests that thin-layer dredged-material depositions up to 4 cm deep can augment marsh habitats with little to no negative impact on marsh vegetation or benthic microalgae (Ford et al. 1999; Leonard et al. 2002) and that thin–layer depositions of 4-10 cm in thickness can cause declines in vegetation and fish abundance (this study). The response of marsh biota to thin-layer dredged material application will likely vary by marsh habitat type which is largely defined by erosion rates and flooding regime. At Wysocking Bay, the deposition of 4 cm and 10 cm thin-layer treatments significantly altered marsh elevation for 12 years and elevation appeared to be the most critical factor in the recovery of marsh biota.

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APPENDICES

Appendices are enclosed in an electronic format on the attached compact disc (CD). A hard copy of all raw data is retained by author and a separate copy was submitted to Dr. Joseph Luczkovich (thesis advisor) at East Carolina University's Institute of Coastal and Marine Resources.