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Sediment subsidy: effects on soil–plant responses in a rapidly submerging coastal salt marsh

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Abstract

The tidal energy subsidy hypothesis postulates that the high primary productivity of coastal salt marshes is the result of an energy subsidy provided by the tides. The sediment component of this subsidy is especially important in contributing to the elevation increase of the marsh surface, a process essential for the sustainability of salt marshes during periods of sea level rise. This research tested the hypothesis that sediment subsidies have an ameliorating effect on sea level rise-induced impacts to salt marsh vigor. We assessed the plant structural and soil physico-chemical responses to different intensities of sediment subsidy in a salt marsh experiencing a high rate of relative sea level rise. Sediments were hydraulically dredged with a high fluid to solids ratio (85%:15%) from the Gulf of Mexico and dispersed into a *Spartina alterniflora* dominated salt marsh. Approximately 2 years after this fluid-sediment-enrichment, maximum sediment elevation did not exceed 30 cm above ambient and both plant cover and aboveground biomass responded positively. Sediment subsidy increased soil mineral matter, and, in turn, soil fertility and marsh elevation, and thereby reduced nutrient deficiency, flooding, and interstitial sulfide stresses. Thus, sediment subsidy generated a more favorable environment for plant growth and potentially, marsh sustainability.

Keywords: Bulk density; Flooding; Fluid-sediment-enrichment; Nutrients; Salt marsh restoration/enhancement; Soil physico-chemistry; Submergence; Subsidence

1. Introduction

Coastal salt marshes are documented as some of the most productive natural ecosystems in the world, and they are well recognized for their important ecological functions and societal values (Costanza et al., 1998; Dawes, 1998; Mitsch and Gosselink, 2000). The high primary productivity of this ecosystem has been attributed to the "energy subsidy" provided by the tides (Odum and Fanning, 1973; Odum et al., 1983). Odum and his colleagues postulated that the daily tidal ebb and flow counteracts the stresses imposed by the saline and water saturated soil environment. Although the actual mechanisms for this subsidy were not originally identified, it is now generally accepted that it results from (1) the introduction of sediments and associated plant nutrients and (2) the drainage of soil pore water and the resulting removal of potential phytotoxins,

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like hydrogen sulfide (see Mendelssohn and Morris, 2000). In addition, the sediment component of this subsidy is especially important in contributing to the elevation increase of the marsh surface (Cahoon et al., 1995; Cahoon and Lynch, 1997) through both a direct effect of sediment volume and an indirect effect of fertility-enhanced organic matter production (Nyman et al., 1990; DeLaune et al., 1992).

The increase in marsh surface elevation that may result from sediment accretion is essential for the stability and long-term sustainability of coastal wetlands experiencing sea level rise (Baumann et al., 1984; Churma et al., 1992; Cahoon et al., 1995). The rate at which the marsh surface elevation increases over time must keep pace with the rate of water level rise, or salt marshes will become excessively inundated (Baumann and DeLaune, 1981; Baumann et al., 1984; Reed and Cahoon, 1992; Reed, 1995), resulting in the death of the marsh vegetation due to reduced substrate aeration (Mendelssohn et al., 1981) and the accumulation of hydrogen sulfide to toxic levels (Ingold and Havill, 1984; Mendelssohn and McKee, 1988; Koch and Mendelssohn, 1989; Webb et al., 1995). Louisiana's Mississippi River Delta Complex, where 80% of the US coastal wetland loss in the past 70 years has occurred, is presently experiencing such effects (Mendelssohn et al., 1983; Penland et al., 1990; Boesch et al., 1994). The significance of this sediment subsidy will likely become increasingly important if the present rise in sea level accelerates as predicted (IPCC, 2001; Twilley et al., 2001). Our research tested the hypothesis that sediment subsidies have an ameliorating effect on sea-level rise-induced impacts to salt marsh vigor. Additionally, the potential biogeochemical mechanisms responsible for this effect were evaluated. The sediment subsidy treatments were generated through the addition of various depths of hydraulically dredged sediments over a 43 ha area of rapidly submerging and deteriorating intertidal salt marsh (Penland and Ramsey, 1990; Dunbar et al., 1992). Although some researchers have investigated sediment addition to marshes (Reimold et al., 1978; DeLaune et al., 1990; Wilber, 1992; Ford et al., 1999), to our knowledge this is the first large-scale test of the sediment subsidy hypothesis in the context of sea level rise remediation. If we are to sustain intertidal wetlands as sea level rises, a test of the sediment subsidy hypothesis and an understanding of the mechanisms controlling salt marsh response to sediment addition are essential. These are the goals of this research.

2. Methods

2.1. Study site

The sediment subsidy hypothesis was tested within the Mississippi River Delta Complex, which comprises 40% of the coastal wetlands of the conterminous United States and is presently experiencing rates of water level rise and resulting wetland degradation equivalent to that predicted worldwide from global climate change toward the end of the 21st century. The high rate of sea level rise in the Mississippi River Delta Complex is primarily due to the compaction of unconsolidated Holocene alluvial sediments, as well as tectonic faulting (Morton and Purcell, 2001), which in the absence of sufficient sediment input result in high rates of surface subsidence (see Boesch et al., 1994). Rates of relative sea level rise (the combined effects of eustacy and isostacy) for the Mississippi River Delta Complex (Delta and Chenier Plains) range from 0.36 to 1.77 cm per year, based on tide gauges maintained by the US Army Corps of Engineers (Penland and Ramsey, 1990).

The specific study site is a degrading, intertidal salt marsh located 106 km southeast (29°12.31′N, 89°26.23′W) of the city of New Orleans, near Venice, LA. This salt marsh is located within the Modern (Birdsfoot) delta, which experiences rates of relative sea level rise (0.94 cm per year at Port Eads from 1944 to 1988; Penland and Ramsey, 1990) approximately nine times that of world-wide eustacy. This high rate of relative sea level rise, due to high land subsidence, has resulted in some of the highest rates of land loss for coastal Louisiana (5.7–11.4 km^2 per year since 1958; Dunbar et al., 1992). Although other natural and anthropogenic factors likely contribute to wetland loss in this area, and the rest of coastal Louisiana (Mendelssohn et al., 1983), the high flooding duration (47%, see Fig. 4) in this wetland provides support for relative sea level rise as a major factor determining wetland health. This salt marsh and those nearby exhibit low resilience to disturbance and low vigor (Mendelssohn and Slocum, 2002), and based on the degradation and loss of immediately adjacent wetlands over the last 50 years (Dunbar et al., 1992), these facts suggest that those remaining wetlands are highly stressed and have a limited lifespan. Because these subsiding marshes have experienced water level increases that fall within the range of predicted increases in sea level (9–88 cm; IPCC, 2001) by 2100, the Mississippi River Delta Complex provides an ideal natural laboratory in which to test the sediment subsidy hypothesis with respect to the remediation of sea level rise-induced impacts to coastal wetlands.

2.2. Methodology

In January 1992, a sediment slurry (85% liquid and 15% solids), which as mitigation was hydraulically dredged from the Gulf of Mexico to fill a gas pipeline canal, accidentally overflowed into an adjacent submerging salt marsh located 106 km southeast (29°12.31′N, 89°26.23′W) of the city of New Orleans, near Venice, Louisiana. This activity initially resulted in a gradient in depth of added sediment from trace amounts to as much as 60 cm of sediment above the natural marsh surface over a 43 ha area. We utilized this sediment addition gradient to evaluate the sediment subsidy hypothesis as a function of different intensities of sediment addition on plant community structure and soil condition as described below.

A rotary laser level was used to measure the depth of added sediment in the marsh. Based on the results from this survey, the study site, a deteriorating, intertidal, saline marsh dominated by *Spartina alterniflora* Loisel, was divided into five areas for purposes of this study:

(1) Reference: marsh which received no hydraulically dredged sediment;
(2) SAR-I (sediment affected region I): marsh receiving trace amounts of sediment, but not quantifiable with standard elevation survey techniques;
(3) SAR-II: marsh with measurable sediment burial not greater than 15 cm;
(4) SAR-III: marsh with 15–30 cm of sediment burial;
(5) SAR-IV: marsh with more than 30 cm of sediment burial.

Five permanent sampling sites were randomly located within each of the five sediment addition areas for a total of 25 permanent sampling plots. Each sampling plot was a 2 m × 2 m area. Vegetation data were collected on 12 November 1993 and 24 August 1994 to represent end of the growing season responses. Live and dead plant biomass and total live and dead cover were sampled on both sampling dates while species percent cover was measured only in 1994. Percentage cover was visually determined. Also, a 0.25 m^2 area adjacent to each permanent sampling plot was clipped at the soil surface of all vegetation. Clip-plots were haphazardly located within an area representative of the permanent plot's species assemblage. In the laboratory, plant material collected from each sampling plot was separated into live and dead portions by species. This plant material was dried to a constant weight in a forced-air oven at 65 °C and weighed to the nearest 0.1 g. From this vegetative data, standing crop and species composition were determined for the five study areas.

In addition, various soil physico-chemical characteristics were measured. However, unlike the vegetative data, different soil parameters were measured in November 1993 and August 1994. Exchangeable soil elements and organic matter content were measured in 1993, while interstitial elemental concentrations and all other soil variables were measured in 1994. Soil oxidation–reduction potential (Eh) was measured within the permanent plots using six bright platinum electrodes and a calomel reference electrode with three subsamples taken at the soil surface and three subsamples at 15 cm below the surface at each site (1994). An average of the three subsamples from each depth was used for statistical analysis. Soil samples were collected from the same area from which plant clip-plots were collected for extractable NH_4–N, P, Ca, Mg, K, Na, Fe, Mn, Cu and Zn analyses (1993). Soil extraction of these elements followed standard techniques (NH_4–N extraction following Bremner and Keeney, 1966; P extraction following Byrnside and Sturgis, 1958; Ca, Mg, K, Na extraction following Thomas, 1982; Fe, Mn, Cu, Zn extraction following Baker and Amacher, 1982). After extraction, NH_4–N samples were filtered through a 0.45 μm syringe filter and concentrations were measured using the Colorimetric, Automated Phenate Method (US Environmental Protection Agency (EPA), 1979). The remaining elemental concentrations were measured using a Fisher inductively coupled argon plasma emission spectrometer (ICP). Data from soil extractions are presented on a soil-volume basis since this method is preferable in situations where there is a wide variation in bulk density between study sites (DeLaune et al., 1979). Soil cores for measuring bulk density were collected in 1994 from the same area from which the plant clip-plots were collected. The bulk density cores (5 cm diameter × 9.5 cm height) were dried at 65 °C to a constant weight, and mass determined.

Additional soil cores (5 cm diameter × 25–30 cm length), collected in 1994 to measure the chemical constituents of the soil water, were immediately sealed in 500 ml centrifuge bottles in the field, purged with nitrogen gas through an air-tight septum for 2 min, and then placed on ice. In the laboratory, these soil samples were centrifuged (5000 rpm at 5 °C for 12 min) under nitrogen, and interstitial water was immediately removed from them (1994). One subsample of this water was used to measure salinity using an ATAGO S/Mill refractometer. Another subsample was placed in antioxidant buffer and analyzed for total soluble sulfide concentration (Sulfide Electrode, Lazar Research Laboratories, Los Angeles, CA, USA). A third subsample was filtered through a 0.45 μm syringe filter and used to measure interstitial NH_4–N concentrations (Colorimetric, Automated Phenate Method) (US EPA, 1979). This subsample was also analyzed for nitrate–N, but concentrations were so low (≈8 μM), even at the highest elevations (SAR-IV) where nitrification could have occurred, that they are not included here. The final subsample was also filtered through a 0.45 μm syringe filter, preserved with nitric acid (US EPA, 1979), and analyzed for a suite of other elements (i.e. P, Ca, Mg, K, Na, Fe, Mn, Cu, Zn) using the ICP. A problem arose with the ICP readings for Na that resulted in their exclusion from analysis. However, because of the close relationship between salinity and Na, we developed a regression equation ($[\mathrm{Na}] = 21.93 + 8.63 \times [\mathrm{salinity}\,(\mathrm{g\,l^{-1}})]$, adjusted $R^2 = 0.90$, $n = 75$) to estimate Na concentrations for these sampling periods from data collected later at the site (Robert P. Gambrell, personal communication, Wetland Biogeochemistry Institute, Louisiana State University). Na concentrations measured at a later date using the ICP very closely resembled those estimated by the regression equation.

After the above analyses had been completed, the soils were used to determine percent organic matter content and soil texture for each treatment area. Percent organic matter was estimated using a modification of the Walkley Black method presented in Nelson and Sommers (1982) (data from 1993). Soil texture was analyzed in 1996 for each site using the method of Gee and Bauder (1986).

We estimated the hydroperiod of the study sites for 1992 and 1993 from hourly water level measurements that were available in 1998 from a continuously recording water level gauge. To estimate tidal conditions in 1992 and 1993 for each permanent plot, the average change in elevation in each treatment area from 1992 to 1998 and from 1993 to 1998 was added to respective elevation measurements taken for each permanent plot in 1998 (note that due to sediment compaction elevations were lower in 1998 than in 1993). These 1992 and 1993 elevation estimates were used along with the 1998 tidal data to estimate the number of hours above/below the soil surface, percent time flooded, and average water level for each of the 25 plots in 1992 and 1993, assuming a similar tidal regime for these time periods. Because these data are only rough estimates of the actual tidal regime for these time periods, we just present the overall means for tidal levels within each of the four SAR areas for 1992 and 1993 and present no error terms or statistical test results. These data are only provided to furnish a semi-quantification of the different hydroperiods resulting from the sediment additions.

2.3. Statistical analyses

Data were analyzed using repeated measures ANOVA. All significant main effects were further investigated using Fisher's protected LSD. In order to improve normality and homogeneity of variance, certain variables had to be transformed. Log transformations were done on sulfide, interstitial NH_4–N, and interstitial iron and manganese. Arcsine square root transformations were done to the percent cover, percent organic matter, and soil texture data (i.e. percent sand, silt, and clay). General trends within the data were revealed by plotting the mean and S.E. of the raw data for each treatment. For data collected in both 1993 and 1994, this paper presents the overall treatment means averaged over time to emphasize the overall sediment-subsidy treatment effects and because time effects were minimal and not ecologically significant.

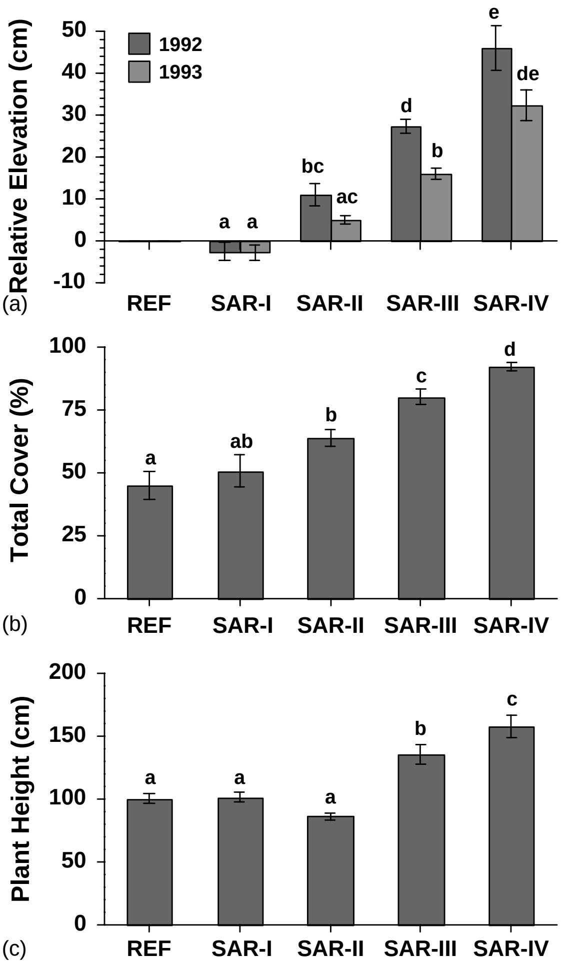

Fig. 1. The effect of sediment addition on (a) relative elevation in 1992 and 1993 ($n = 8$, 24, 14, and 9 for SAR-I, -II, -III, and -IV, respectively), (b) total vegetative cover ($n = 10$), and (c) plant height ($n = 10$) in the five treatment areas. Data are means, averaged over 1993 and 1994 for vegetative parameters, and standard errors. Shared letters indicate no statistical difference between treatment means (Fisher's protected LSD, $P < 0.05$).

3. Results

3.1. Vegetative parameters

Plants in the sediment subsidized areas showed a positive response to increasing depths of added sediment (Fig. 1a and b). With increasing levels of sediment subsidy, there was a significant increase in total cover ($P < 0.0001$) with levels in the SAR-IV sites more than twice those in the Reference area (Fig. 1b). This pattern is reflected in the cover estimates for *S. alterniflora*, the marsh dominant (Table 1). Cover values for *Spartina patens* (ait.) Muhl and *Distichlis spicata* (L.) Greene were low (Table 1), and their frequency of occurrence was negligible (data not shown). Only the increase in *S. alterniflora* cover with increasing sediment addition was significant ($P < 0.002$).

Similarly, plant height was 30–60% greater with increasing sediment subsidy (Fig. 1c). Sites that received the most sediment (SAR-III and -IV) had plants that were significantly taller ($P < 0.0001$) than those in sites receiving little or no sediment (Reference, SAR-I and -II).

Plant biomass also increased as the amount of sediment subsidy rose (Fig. 2). Total live + dead biomass

Table 1
Percent cover in 1994 of *Spartina alterniflora*, *Spartina patens*, and *Distichlis spicata* as a function of sediment subsidy

Treatment	Percent cover		
	Spartina alterniflora	*Spartina patens*	*Distichlis spicata*
Reference	47 ab (7)	0 (0)	0 (0)
SAR-I	53 ab (7)	0 (0)	1 (1)
SAR-II	44 a (3)	6 (5)	1 (1)
SAR-III	59 b (2)	0 (0)	0 (0)
SAR-IV	76 c (4)	0 (0)	1 (1)

Note. Shared letters within the same column indicate no statistical difference between means (Fisher's protected LSD, $P < 0.05$). No significant differences existed between treatments for *Spartina patens* and *Distichlis spicata*. Data are means with standard errors in parentheses ($n = 5$).

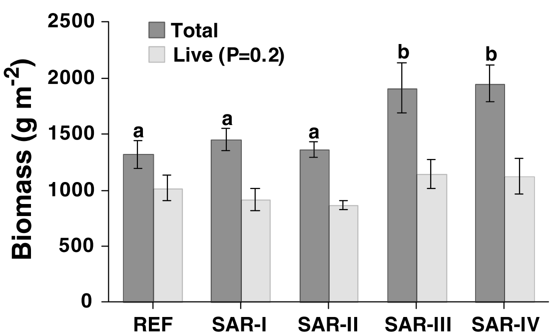

Fig. 2. Total biomass and total live biomass in the five sediment subsidized areas. Data are means, averaged over 1993 and 1994, and standard errors ($n = 10$). Shared letters indicate no statistical difference between treatment means (Fisher's protected LSD, $P < 0.05$).

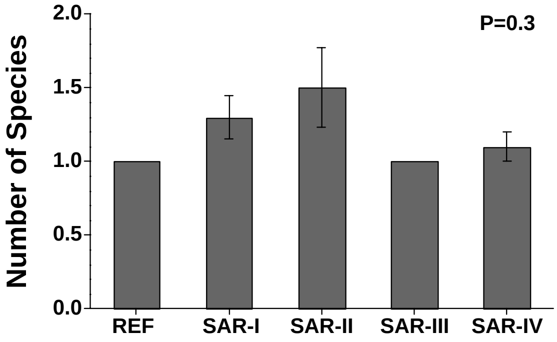

Fig. 3. Number of species in the five treatment areas. Data are means, averaged over 1993 and 1994, and standard errors ($n = 10$).

significantly increased with sediment addition ($P = 0.004$), but the pattern was not as clear for live biomass alone ($P = 0.2$). Total live + dead biomass was between 32 and 48% greater in SAR-III and -IV than in the remaining treatment areas. Live biomass followed a similar pattern.

No change in species composition occurred at the sites with increasing sediment additions. The number of species found in all five areas was similar. SAR sites had an average of between one and two species and a range of one to three species (Fig. 3). Typically, *S. alterniflora* was dominant at all SAR sites with *S. patens* and/or *D. spicata* as minor residents. However, *S. alterniflora* was the only species present in the Reference plots.

3.2. Soil physico-chemical status

Most physico-chemical parameters also showed some form of response to differential sediment subsidy. With progressively more sediment added, redox potential rose at the soil surface and at depth (15 cm) ($P = 0.008$ and 0.005, respectively) (Table 2). SAR-III and -IV sites had much higher redox potentials ($P \leq 0.05$) at the surface than areas with little or no sediment addition (Reference and SAR-I). Likewise, redox potential at depth (15 cm) followed the same pattern as sediment deposition increased with SAR-III and -IV sites again having significantly higher Eh ($P \leq 0.04$) when compared with the Reference and SAR-I. These Eh data agree well with the tidal regime estimates. Estimates of the number of hours above/below marsh surface, percent time flooded and average water level all decreased with increasing sediment additions (Fig. 4). Also, flooding depths and durations increased markedly at the higher elevation sites from 1992

Table 2
Redox potential at the soil surface (0 cm) and at rooting depth (15 cm) for the five sediment subsidized areas for 1994

Treatment	Redox potential, Eh (mV)	
	Surface (0 cm)	Depth (15 cm)
Reference	−73 a (46)	3 ab (20)
SAR-I	−21 a (26)	−37 a (17)
SAR-II	5 ab (29)	44 bc (27)
SAR-III	67 b (8)	74 c (22)
SAR-IV	76 b (24)	91 c (28)

Note. Shared letters within the same column indicate no statistical difference between means (Fisher's protected LSD, $P < 0.05$). Data are means with standard errors in parentheses.

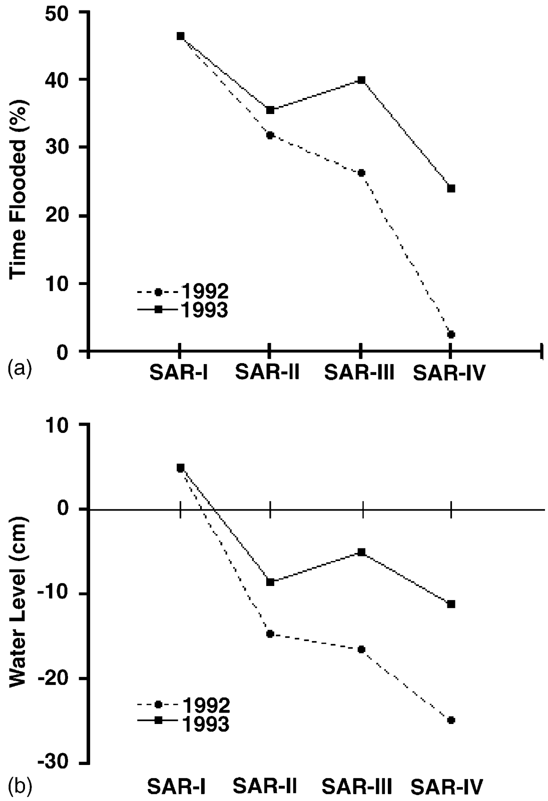

Fig. 4. The effects of sediment addition on (a) percent time flooded, and (b) average water level for 1992 and 1993. Data are overall means for each SAR area with no error bars or statistical differences shown since data were estimated from 1998 tidal measurements (see text).

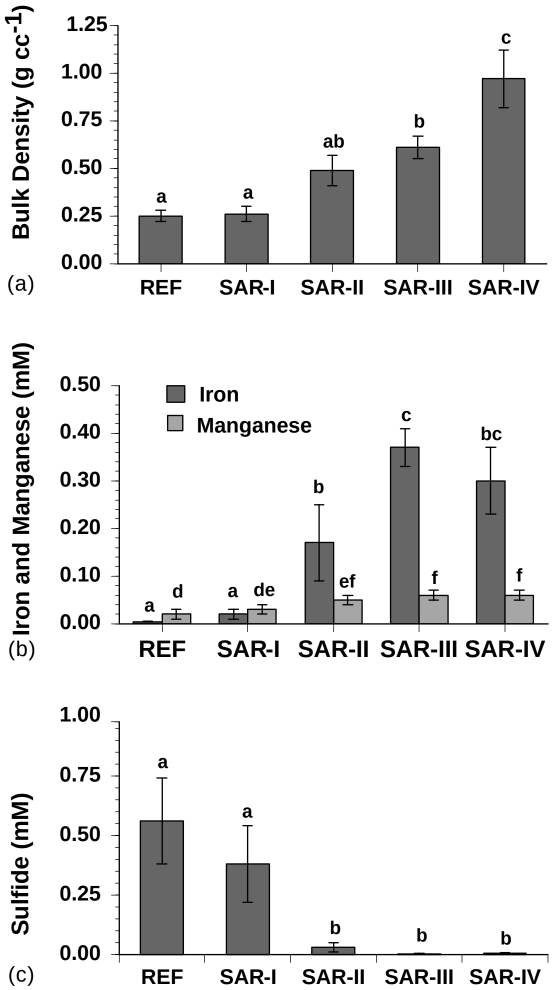

Fig. 5. The effect of sediment addition on (a) bulk density ($n = 5$), (b) interstitial iron and manganese ($n = 5$), and (c) interstitial sulfide ($n = 5$) in the five treatment areas in 1994. Data are means and standard errors. Shared letters indicate no statistical difference between treatment means (Fisher's protected LSD, $P < 0.05$).

to 1993 as the substrate in these areas compacted (Fig. 4).

Bulk density showed a steady and pronounced increase from Reference to SAR-IV sites ($P < 0.0001$) (Fig. 5a). SAR-IV bulk densities were roughly four times higher than in the Reference and SAR-I areas. Only the Reference and SAR-I sites appeared to have similar bulk densities (Fig. 5a).

Interstitial iron and manganese concentrations significantly increased ($P < 0.0001$ and 0.004, respectively) with increased sediment subsidy and bulk density (Fig. 5b). Soil extractable iron and manganese showed similar trends with sediment addition ($P = 0.001$ and <0.0001, respectively) (Table 3). In contrast, interstitial copper and zinc levels were similar for the different treatment areas (data not shown). The same pattern was seen for extractable Cu and Zn concentrations (Table 3).

Interstitial sulfide concentrations fell sharply with increased sediment addition ($P = 0.004$) (Fig. 5c). Reference and SAR-I plots had sulfide concentrations much higher ($P \leq 0.02$) than the remaining treatment areas. Sulfide concentrations in the Reference and SAR-I sites were six times higher than those for SAR-II and roughly 2 orders of magnitude higher than those in SAR-III and -IV areas. Average sulfide concentrations measured in 1994 for Reference and SAR-I sites (0.6 ± 0.2 and 0.4 ± 0.2 mM, respectively) were not at levels known to be lethal to salt marsh plants, but were approaching concentrations high enough to impair plant growth ($\geq$1.0 mM; Koch and Mendelssohn, 1989).

Interstitial (Fig. 6) and exchangeable (Table 3) NH_4–N concentrations also decreased as added sediment depth increased, but only the decrease in interstitial NH_4–N concentrations was significant ($P = 0.002$). Interstitial NH_4–N concentrations in the SAR-II, -III, and -IV sites were 7–20 times lower ($P \leq 0.04$) than those in the Reference and SAR-I areas (Fig. 6a). Exchangeable NH_4–N concentrations in the high elevation sites (SAR-III and -IV) were roughly half those of the Reference, SAR-I and -II (Table 3).

Conversely, interstitial phosphorus concentrations significantly rose ($P = 0.008$) from the Reference to SAR-III plots but decreased to concentrations similar to the Reference in SAR-IV (Fig. 6b). Exchangeable P concentrations showed a similar pattern, significantly increasing from the Reference to SAR-IV sites ($P = 0.04$) (Table 3).

Interstitial soil salinity was higher with more added sediment ($P < 0.0001$) (Fig. 7). Individual comparisons indicate that areas receiving the most sediment (SAR-III and -IV) had significantly higher ($P \leq 0.0003$) salinities than those receiving the least (Reference and SAR-I). However, the absolute difference was only about 4 g l^{-1}.

Table 3
Exchangeable soil nutrient concentrations in response to sediment subsidy in 1993

Treatment	Exchangeable soil nutrient concentrations (μmol cm^{-3})									
	NH_4–N	P	Ca	Mg	K	Na	Fe	Mn	Cu	Zn
Reference	0.258 (0.054)	1.94 a (0.29)	16.38 a (1.39)	26.52 (2.19)	8.24 (0.90)	77.86 (5.02)	6.86 a (2.37)	0.28 a (1.2)	0.03 (0.01)	0.09 (0.02)
SAR-I	0.171 (0.038)	2.36 a (0.54)	22.68 ab (5.02)	30.38 (3.40)	10.38 (1.38)	94.44 (11.38)	10.84 a (3.50)	0.80 a (2.8)	0.02 (0.01)	0.07 (0.01)
SAR-II	0.171 (0.065)	3.97 ab (0.62)	40.98 bc (6.38)	33.38 (3.39)	11.86 (1.37)	98.82 (13.57)	35.74 b (3.72)	3.02 b (2.9)	0.03 (0.01)	0.11 (0.01)
SAR-III	0.115 (0.084)	2.77 a (0.43)	44.80 c (7.34)	33.98 (3.69)	11.10 (1.73)	98.90 (10.54)	28.66 b (3.80)	2.76 b (3.6)	0.04 (0.02)	0.12 (0.02)
SAR-IV	0.101 (0.040)	5.12 b (1.35)	58.84 c (8.69)	28.96 (4.30)	10.08 (1.53)	83.98 (11.99)	28.64 b (8.30)	3.20 b (6.2)	0.05 (0.01)	0.11 (0.02)

Note. Shared letters within the same column indicate no statistical difference between means (Fisher's protected LSD, $P < 0.05$). Data are means with standard errors in parentheses ($n = 5$).

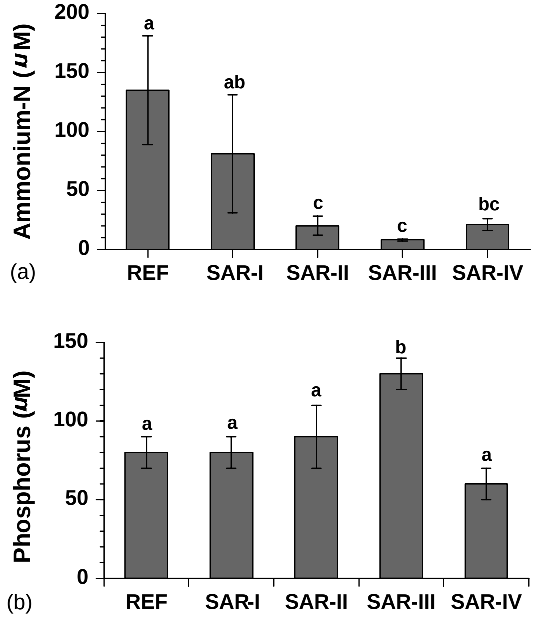

Fig. 6. The effects of sediment addition on (a) interstitial NH_4–N concentrations and (b) interstitial soil phosphorus levels in the sediment subsidized treatments in 1994. Data are means and standard errors ($n = 5$).

The interstitial water concentrations of some soil cations were also affected by sediment subsidy. With increased amounts of added sediment, interstitial Ca, K, Mg, and Na all significantly increased ($P \leq 0.01$) in concentration (Table 4). Calcium, magnesium and sodium levels in the SAR-IV plots were approximately

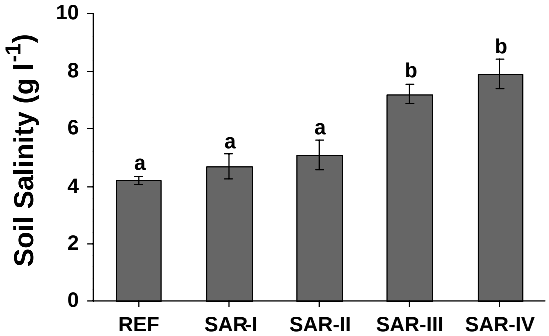

Fig. 7. Interstitial soil salinity in the sediment subsidized treatments in 1994. Data are means and standard errors ($n = 5$). Shared letters indicate no statistical difference between treatment means (Fisher's protected LSD, $P < 0.05$).

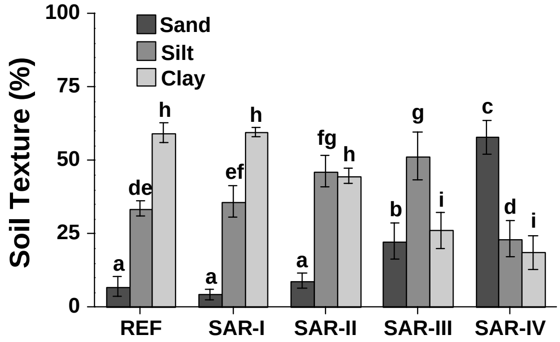

Fig. 8. Soil texture in the five sediment subsidized treatment areas in 1996. Data are means and standard errors ($n = 5$).

twice as high as in the Reference area. Interstitial potassium concentrations rose roughly 40% with increasing sediment additions to and including SAR-III, with concentrations in SAR-IV similar to those of the Reference area. In contrast, of the exchangeable elements, only Ca concentrations significantly increased with sediment addition ($P = 0.0007$) (Table 3). Soil exchangeable Mg, K, and Na concentrations were similar in all study areas.

Soil texture and organic matter content varied greatly along the sediment subsidy gradient. Sites that received more sediment had more sand and less clay than sites receiving less sediment burial (Fig. 8). Percent sand content was significantly higher ($P \leq 0.0001$) in SAR-IV (58% sand) than in all the other treatment areas (22–4% sand). Conversely, percent clay followed the opposite pattern with relative clay content significantly decreasing ($P = 0.0002$) from 59% in Reference and SAR-I sites to 19% in SAR-IV sites. Percent silt content significantly increased ($P = 0.0005$) from the Reference to SAR-III sites but was lower in the SAR-IV plots. However, of all size classes, silt content varied the least between treatment areas with levels at all sites between 23 and 51%. Sediment addition also caused a significant decline in soil organic matter content ($P = 0.004$). However, organic matter content in the soils at all study sites was relatively low (<1%).

4. Discussion

Sediment subsidy affected many of the vegetative and physical parameters measured. Plant height and

Table 4
Interstitial soil cation concentrations in the five sediment subsidized areas in 1994

Treatment	Interstitial soil cation concentrations (mM)			
	Ca	Mg	K	Na
Reference	2.925 a (0.122)	9.24 a (0.25)	1.581 a (0.137)	70.4 a (1.8)
SAR-I	3.323 ab (0.140)	10.50 a (0.79)	1.931 ac (0.082)	81.0 a (5.6)
SAR-II	3.860 bc (0.356)	11.21 a (1.33)	2.072 bc (0.128)	85.1 a (10.4)
SAR-III	4.360 c (0.113)	15.41 b (1.59)	2.245 c (0.149)	122.4 b (7.4)
SAR-IV	5.791 d (0.792)	19.50 b (2.76)	1.745 ab (0.144)	146.7 b (15.3)

Note. Shared letters within the same column indicate no statistical difference between means (Fisher's protected LSD, $P < 0.05$). Data are means with standard errors in parentheses.

biomass increased with increasing sediment deposition. The increase in marsh elevation associated with sediment addition may help explain this pattern. Marsh surface elevation was substantially greater in areas receiving more sediment subsidy, and this increase in elevation had a positive influence on a number of factors affecting plant growth.

First, increased elevation reduced the depth of flooding and improved soil aeration. Redox potential measurements showed that soils were more oxidized in the higher elevation sites than in the lower areas that received less sediment. With a reduction in flooding, the water table would more often fall below the soil surface allowing for soil drainage and the direct exchange of gases between the air and the soil. Also, reduced flooding would facilitate aeration of the rhizosphere by plant roots since more aboveground tissue that contains aerenchyma would be exposed to the air for a longer period of time. Plant growth can be reduced in poorly aerated soils because of an oxygen deficiency in the rooting zone which forces plants to rely more heavily on anaerobic metabolism for their energy production (Mendelssohn et al., 1981). However, since soils were less flooded and more aerated at the higher elevation sites, the roots of plants in these areas could likely respire aerobically and reduce their reliance on alcoholic fermentation (Mendelssohn et al., 1981; Mendelssohn and McKee, 1988) resulting in more growth (Wilsey et al., 1992).

Second, improved soil aeration associated with increased elevation also reduced the concentration of phytotoxins that develop in more reduced salt marsh soils. The major plant toxin typically produced in reduced salt marsh soils is free sulfide (Gambrell and Patrick, 1978; Mendelssohn and McKee, 1988). In highly reduced salt marsh soils such as those in Reference and SAR-I sites, obligate anaerobic bacteria use sulfate from sea water as their terminal electron acceptor during respiration and convert it to sulfide. However, in more oxidized soils like those of SAR-III and -IV, aerobic and facultative anaerobic bacteria that do not produce toxic hydrogen sulfide during respiration are energetically preferred. Sulfide levels in the highest elevation sites (SAR-III and -IV) were far below those that negatively affect plants, but in the lowest elevation sites (Reference and SAR-I) sulfide concentrations did at least approach those which reduce the growth of *S. alterniflora* (≈1 mM; Koch and Mendelssohn, 1989). Sulfide reduces plant growth by impacting energy production, thereby inhibiting metabolic processes such as plant uptake of NH_4–N (Bradley and Morris, 1990; Koch et al., 1990), which is the primary limiting nutrient to salt marsh plant productivity (Valiela and Teal, 1974; Mendelssohn and Morris, 2000).

Third, differences in physical characteristics of the sediment may also help explain the increase in plant production as the sediment subsidy rose. Bulk densities indicate that mineral matter content was much higher in areas receiving the most sediment. Wetland soils with a higher mineral content have a greater ability to take up and sequester nutrients (Mitsch and Gosselink, 2000), and they also have been shown to provide more nutrients on a per volume basis when compared to organic salt marsh soils (DeLaune et al., 1979). Furthermore, the higher mineral content at higher sediment addition levels resulted in greater Fe and Mn concentrations in the interstitial water. These elements are important for their ability to precipitate sulfides and thereby reduce toxic soluble sulfide concentrations (Gambrell and Patrick, 1978). In addition to the mineral content per se, the texture of this fill material is another important factor to consider. Soil texture was not homogeneous throughout the sediment affected areas. SAR-IV sites had a much higher sand content than soils from the other sediment affected areas which were mostly silt and clay. This difference in texture may explain why some variables that had been increasing in value up to and including SAR-III declined in SAR-IV (e.g. K and P), since fine texture soils sorb more nutrients than coarser ones (Brady and Weil, 1996).

Fourth, increased plant growth could also be due to the effects of sediment additions on soil nutrient availability. Both interstitial and exchangeable NH_4–N concentrations were lower with greater sediment addition. However, high plant production in sites receiving the most sediment shows that the low NH_4–N concentrations in these areas did not negatively affect plant growth. In fact, extractable NH_4–N levels measured from the dredged material were 40 times higher than those for similar soil material collected within the study area in 1993 (e.g. 4.44 ± 0.94 μmol NH_4–N cc^{-1} soil in the dredged sediment versus 0.11 ± 0.08 μmol NH_4–N ml^{-1} soil in SAR-III). Thus, given the high plant biomass in areas receiving the most sediment addition plus the initially high NH_4–N content of the fill soil when first deposited in these areas, it appears that the current relatively low nitrogen status of soils in the greater sediment affected areas is at least partly due to plant uptake and removal of this nutrient. However, alternating periods of flooding and drying in sites receiving the most sediment might have also contributed to the low NH_4–N levels in these areas since these conditions would promote nitrogen loss through leaching and denitrification (Patrick and Wyatt, 1964; Brady and Weil, 1996).

In contrast, interstitial and extractable phosphorus concentrations both increased with sediment subsidy. Sites receiving more sediment had a higher soil mineral content than areas that received less fill, as witnessed by the higher bulk densities with increasing sediment addition. Soil P is usually closely associated with mineral matter because of the high retention capacity that the mineral fraction has for this plant nutrient (Syers et al., 1969; Brady and Weil, 1996). Therefore, sites receiving less sediment addition would be likely to have soils with a lower available P content. Thus, plant production may have been further increased in the highest sites because of the increased concentrations of plant available P in these areas, especially if the high initial soil N resulted in a P deficiency.

Finally, the concentration of a number of soil cations important for plant growth was also higher in the sites receiving the most sediment. Interstitial Ca, Mg and Na all increased in concentration as sediment additions rose up to SAR-IV levels, and interstitial K concentrations rose up to SAR-III but fell in SAR-IV. Extractable concentrations of these cations also increased with sediment addition, although only the increase for extractable Ca was significant. These four cations are all major chemical constituents of sea water, and salinity rose with increasing sediment additions. Therefore, as salinity increased, the cations comprising saltwater increased. The somewhat different pattern for interstitial K may be due to the difference in soil texture for SAR-IV. Clay minerals are a major source of K in the soil, typically providing K to plants via cation exchange processes on their charged outer surfaces and also more slowly through weathering processes (Brady and Weil, 1996). Thus, the greater clay content in the fill material for SAR-I, -II and -III sites may account for the slightly higher levels of interstitial K seen in these areas versus the sandy soils of SAR-IV.

A change in species composition was expected at high sediment subsidy levels, but no such change occurred. With the increase in elevation associated with sediment addition, it was thought that areas receiving the most sediment might convert to upland or high marsh habitats due to the removal of plant stressors such as salinity and inundation. However, we found that salinities were actually higher in the sites receiving the most sediment although the areas were less flooded. Some higher marsh species (*S. patens* and *D. spicata*) were found in the SAR-II, -III and -IV areas, but these sites exhibited no change in relative dominance over the two years following sediment addition (1993 and 1994). However, species composition may still change over time, and we have continued to monitor this area to follow any changes in composition that do occur. The low marsh species, *S. alterniflora*, readily reestablished after sediment introduction in SAR-II, -III, and -IV sites. Its reestablishment in SAR-III and -IV sites was primarily from seed, as resident stands of *S. alterniflora* were buried and killed during sediment addition. However, revegetation of SAR-II was mostly from the vegetative regrowth of underlying plants that were not buried by enough sediment to cause plant mortality (I.A. Mendelssohn, personal observation). It seems likely that *S. alterniflora* dominates in these areas simply because it pre-empted the space into which other species might invade.

Both manipulative field experiments and smaller-scale case studies support our findings of enhanced plant response to sediment subsidy. Probably the most rigorous experimental investigation of sediment addition to salt marshes was done by Reimold et al. (1978). Although these investigators were primarily interested in determining if dredged material could be disposed of in *S. alterniflora*-dominated salt marshes in Georgia, USA without causing detrimental impacts to the marsh flora and fauna, their research has direct application to the use of dredged material for marsh rehabilitation. They added one of three sediment types (coarse sand, clay, and a mixture of sand and clay) at seven burial depths (0, 8, 15, 23, 30, 61, and 90 cm) in 0.9 m diameter plots. As we found in the SAR-III and -IV marsh sites, *S. alterniflora* was not able to survive the highest sediment addition levels (61 and 90 cm). However, *S. alterniflora* biomass was enhanced at sediment additions of 8, 15, and 23 cm burials with a significant reduction at 30 cm. Interestingly, seedling establishment in the 61 and 90 cm burial treatments resulted in considerable new *S. alterniflora* biomass production, just as observed in SAR-III and -IV sites in the present study. In fact, the highest live *S. alterniflora* biomass of all the sediment burial treatments occurred in the 61 cm burial treatment (Reimold et al., 1978).

The stimulatory effect of sediment additions to low productivity *S. alterniflora* salt marshes in coastal Louisiana, USA was also documented by DeLaune et al. (1990). They found that as little as 4–6 cm of added sediment doubled biomass production, primarily from new stem production. The addition of 8–10 cm of new sediment did not further increase production. Within the last decade, the use of thin-layer deposition of dredged materials, by spray-dredging technology, to restore marsh elevations of degraded wetlands has received some attention (Cahoon and Cowan, 1988; Wilber, 1992, 1993). Ford et al. (1999) demonstrated that the spray-deposition of only 2.3 cm of sediment had a significant enhancing effect on plant cover of a *S. alterniflora* marsh in southeast Louisiana, USA. Wilber (1992, 1993) also demonstrated that thin-layer deposition of dredged material can restore marsh elevation, however, the thickness of the added sediment is critical both in determining the initial impact to the existing vegetation and the longer term growth response (Reimold et al., 1978, this study). The aforementioned citations, however, support our findings that sediment subsidies can stimulate plant production and ameliorate the impacts of sea level rise-induced marsh submergence.

5. Conclusion

The addition of sediment to this submerging and deteriorating salt marsh caused a rapid, positive response to the vegetation. Thus, with respect by the sediment subsidy hypothesis, we have shown that sediment subsidies can ameliorate the negative impacts of accelerated sea level rise on salt marsh vigor. Sediment enrichment improved salt marsh plant growth by increasing soil aeration, mineral matter content and available nutrients. Areas receiving intermediate and high amounts of sediment (15–30 and 30–60 cm, respectively, after 2 years) showed increased plant production. Moreover, the highest elevation-SAR's are likely to be the most sustainable because of a reduced, if not reversed, sediment aggradation deficit. However, the optimum clay and silt content of the fill soil should be more like that of the SAR-II and -III sites, which have a higher capacity to hold nutrients. Thus, it appears that sediment enrichment could play a positive role in the management of sediment starved marshes including deteriorating coastal wetlands where relative sea level rise exceeds vertical elevation accretion. The use of hydraulically dredged sediment with a high fluid to solids ratio allows the sediment to be dispersed over relatively long distances across sediment deficient marshes. Fluid-sediment-enrichment is a modification of the traditional beneficial use of dredged sediment, which attempts to maximize the proportion of solids to fluid in the discharge, to create marshes in areas of shallow open water. The real value of fluid-sediment-enrichment is for the ecological enhancement of existing wetlands that are stressed due to imbalances between marsh elevation and water level, resulting in excessive wetland submergence. Consequently, this methodology could also have important implications for coastal wetlands worldwide if the rate of sea level rise increases with global climate change.

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